NATIONAL WATER QUALITY MANAGEMENT STRATEGY

PAPER No. 4

Australian and New Zealand Guidelines for Fresh and Marine Water Quality

Volume 3

Primary Industries — Rationale and Background Information

(Irrigation and general water uses, stock drinking water, aquaculture and human consumers of aquatic foods)

(Chapter 9)

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9.1 Introduction

Both the quality and the quantity of water resources are critical issues for agriculture and aquaculture in Australia and New Zealand. Water quality is also of major importance for the protection of human consumers of food products. In keeping with the principles of ecologically sustainable development, these guidelines have been developed to take consideration of not only productivity issues but also the possible adverse impacts of these primary industries on downstream water quality.

Productivity in the Australian agricultural sector has increased significantly during the 1990s, with the highest growth rates occurring in specialist broadacre cropping industries. In comparison, livestock industries have been relatively static over this period (Wilson & Johnson 1997). In 1996–97, the gross value of agricultural commodities produced in Australia was approximately $28 000 million, with a significant proportion contributing to export earnings (ABS 1999).

The value of aquaculture production in Australia has been growing at over 10% per annum since the late 1980s, with an estimated farm gate value in excess of $464.6 million in 1994–95 (O'Sullivan & Kiley 1996). The industry is also expanding rapidly in New Zealand. The intimate association between the cultured organisms and their water environment makes water quality of paramount importance in achieving high production rates and profitability.

Agriculture is a major consumer of water resources in Australia and New Zealand, predominantly for use in irrigation and livestock watering. The industry relies on the use of both surface and groundwater resources, since rainfall in most regions is inadequate for industry requirements. Where appropriate, the guidelines provided for agricultural water use (irrigation, livestock and general water use) are applicable to both surface and groundwater quality.

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9.2 Water quality for irrigation and general use

The water quality guidelines recommended for agricultural use (irrigation, livestock drinking water and general on-farm use) have been derived using information from the previous guidelines (ANZECC 1992), extensive literature reviews, contemporary research data and inputs from public comment. Although the focus has been primarily on Australia and New Zealand, guidelines used in other countries have been considered and evaluated, particularly in regard to certain toxicant levels, where limited local data are available. Issues concerning particular methodologies used to develop specific guidelines are discussed further in the relevant Sections.

Water quality guidelines developed overseas were also reviewed, including the South African Water Quality Guidelines (DWAF 1996a,b) and the Canadian Water Quality Guidelines (CCREM 1987). Three major agricultural databases, CAB, AGRICOLA and AGRIS, were searched for current scientific information on most water quality issues for irrigation and livestock use, covering the period from 1985 to 2000. The search on issues relating to metals, metalloids and nutrients in irrigation water covered a shorter period, 1990 to 2000. Searches for livestock drinking water quality guidelines encompassed each individual parameter for primary domestic animals (cattle, sheep, goats, pigs and poultry), as well as other animals such as horses, emus and ostriches.

Methodologies used to develop specific guidelines are discussed further in relevant Sections. The primary emphasis in revising guidelines was on sustainability in agricultural practice (DEST State of the Environment Advisory Council 1996), which aims to ensure that:

- the supply of necessary inputs is sustainable;
- the quality of natural resources is not degraded;
- the environment is not irreversibly harmed;
- the welfare and options of future generations are not jeopardised by the production and consumption activities of the present generation;
- yields and product quality are maintained or improved.

9.2.1 General considerations for assessment of irrigation water quality

In assessing the suitability of waters for irrigation use, water quality characteristics that affect agricultural production, catchment condition, and downstream water quality need to be evaluated. With concerns about the decreasing quality of surface and groundwaters, and an increasing interest in wastewater and on-farm reuse, emphasis is placed on more comprehensive and flexible guidelines for irrigation water quality. Table 9.2.1 summarises the key issues concerning irrigation effects on soil, plants and water resources, and highlights factors taken into consideration in this review.

The irrigation of agricultural land, while directly influenced by irrigation water quality, is also affected by a number of other parameters which must be considered when planning a sustainable irrigation management program.

Table 9.2.1 Key issues concerning irrigation water quality effects on soil, plants and water resources

	Key issues
Soil	Root zone salinity
	Soil structural stability
	Build-up of contaminants in soil
	Effects on soil biota
	Release of contaminants from soil to crops and pastures
Plants	Yield
	Product quality
	Salt tolerance
	Specific ion tolerance
	Foliar injury
	Uptake of toxicants in produce for human consumption
	Contamination by pathogens
Water resources	Deep drainage and leaching below root zone
	Movement of salts, nutrients and contaminants to groundwaters and surface waters
Other important factors	Quantity and seasonality of rainfall
	Soil properties
	Crop and pasture species and management options
	Land type
	Groundwater depth and quality

9.2.1.1 Catchment water balance

The water balance of a catchment is an important factor in considering irrigation water quality. Assessment of water movement through the catchment gives an indication of potential contaminant transport, sources, sinks and concentrations, and allows more effective management decisions to be made by the individual landholder. Consideration should be given to an overall water cycle management strategy, with an emphasis on maintaining both on-site and downstream water quality.

9.2.1.2 Soil characteristics

Crop yields under irrigation are influenced by the physical and chemical characteristics of soils, for example, fertility, texture, structure, clay percentage, water-holding capacity, cation exchange capacity (CEC), exchangeable sodium percentage (ESP), leaching fraction (LF), pH, organic matter and trace elements. The suitability of a water for irrigation use therefore depends to some extent on its interaction with the soil environment. These guidelines have attempted to take soil characteristics into consideration to give a more accurate estimate of irrigation requirements and sustainable levels of application.

9.2.1.3 Crop tolerance

The level of tolerance to various toxic substances varies between different crop species. Toxicity problems occur if certain constituents in the soil or water are taken up or absorbed by the plant and accumulate to concentrations high enough to cause crop damage, reduced

yields or product quality (Ayers & Westcot 1985). Guidelines for crop tolerance to various toxicants are provided including salinity and certain inorganic and organic contaminants.

9.2.1.4 Climatic conditions

The effects on irrigation management of rainfall, temperature and evapotranspiration must be considered in relation to the quality of water and its application rate, and the existing soil environment.

The amount of rainfall in a particular region can influence soil structure and solute transport mechanisms within the soil profile. The leaching flux, soil water content, trace element and contaminant concentration are all affected by water application rates and may result in reduced crop yield if not managed in conjunction with seasonal rainfall.

High temperatures and dry conditions, common in the semi-arid irrigation areas of Australia, may lead to increased evapotranspiration rates and can result in the concentration of ions and potential contaminants from irrigation waters in the upper soil profile. This may adversely affect crop and pasture species through accumulation of salt and toxicants in the root zone, leading to decreased productivity and loss of vegetative cover.

9.2.1.5 Subsurface drainage

The provision of adequate drainage is an important component of irrigation management. With the addition of salts and contaminants in irrigation water, and the selective use of the water by plants, concentrations in the root zone need to be managed by altering the degree of leaching in the soil profile.

9.2.2 Biological parameters

9.2.2.1 Algae

No trigger value for algae in irrigation waters is recommended; however, excessive algal growth may indicate nutrient pollution of the water supply.

Description

Algae are chlorophyll-containing plants that exist as simple, uni-cellular or multi-cellular organisms in most surface water sources. Excessive algal growth may occur where there is a combination of 'favourable' environmental conditions, namely suitable flow regime, temperature, an abundance of nutrients and adequate sunlight.

Effect on agriculture

The main problem associated with excessive algal growth in irrigation waters is the blockage of distribution and irrigation equipment. This can result in reduced or uneven flow throughout the irrigation system, which may reduce crop yield and increase overall maintenance costs.

Excessive algal growth in water storages and irrigation ditches commonly occurs as a result of nutrient pollution which may arise from both point and non-point agricultural sources. Eutrophication (the process whereby a waterbody is enriched with nutrients such as nitrogen, phosphorus and organic carbon) may be accelerated and result in excessive levels of algal growth. Of particular concern are blue-green algae (which are actually a form of bacteria known as cyanobacteria), discussed separately below.

Unlike cyanobacteria, algae generally do not release toxins. However, they do have the potential to deoxygenate waterbodies under suitable conditions, resulting in fish kills and stagnation of water, decreasing water quality and making it unsuitable for irrigation purposes.

Excessive algal growth in irrigation water does not appear to affect the health of most irrigated crops, however limited research has been conducted in this area. Waters containing high levels of algae may not be suitable for use on crops which are required to maintain a high level of aesthetic appearance (e.g. unprocessed fruit and vegetables) or those which will be directly used for human consumption (Cooper et al. 1996).

9.2.2.2 Cyanobacteria (blue-green algae)

No trigger values for cyanobacteria in irrigation waters are recommended at this time.

Description

Cyanobacteria (blue-green algae) are naturally occurring micro-organisms that closely resemble algae in morphology, habitat and photosynthetic ability. Some cyanobacteria can produce toxins. In Australia, the genera of concern include *Mycrocystis*, *Anabaena*, *Nodularia* and *Cylindrospermopsis* (NHMRC & ARMCANZ 1996). The latter is normally associated with tropical and sub-tropical regions (Queensland Water Quality Task Force 1992).

Toxic blooms of cyanobacteria can consist of more than one species of cyanobacteria (Van Halderen et al. 1995) and are most likely to occur when wind conditions are mild, water temperature is warm (15–30°C), pH is neutral to alkaline (6–9), hydraulic flows are low (or a reservoir is stratified) and there is an abundance of available nutrients (Carmichael 1994).

Problems associated with cyanobacteria arise when toxins are produced in excessive amounts during these blooms. Cyanobacterial cells (and algal cells) can also cause problems through clogging of filters, sprays and other equipment.

Effect on suitability of waters for irrigation

Concerns regarding the effects of elevated levels of cyanobacteria and associated toxins in irrigation waters used on agricultural produce, crops and pastures have recently been highlighted (Cooper et al. 1996). Uncertainty about the risks associated with low level toxin consumption have raised questions in relation to the use of irrigation waters potentially contaminated with toxic cyanobacteria.

Moreover, many toxins are extremely persistent in the environment, often being resistant to chemical or bacterial degradation (Cooper et al. 1996). This can cause concern where contaminated waters used for irrigation come in direct contact with crops and pastures, creating a potential health risk to human consumers of affected produce and to grazing livestock. In particular, spray irrigation of leafy vegetables such as lettuce and cabbages may represent a risk for accumulation of toxic residues (Jones et al. 1993). Dried cyanobacterial cells can remain toxic for several months on vegetative surfaces (Jones et al. 1995).

If a bloom of toxic cyanobacteria is suspected, a sample should be sent for analysis to identify the species present and if necessary, the level of toxicity. An alternative source of irrigation water should be used in the interim to minimise risk. Professional advice should be sought before any treatment method is implemented to ensure that the most effective measures are taken.

A major constraint to the management and control of cyanobacterial blooms is the risk of releasing toxins into surrounding waters. It is recommended that treated water sources not be used for irrigation for at least ten days. However, in some cases a longer withholding period may be needed; for example, mycrocystins have been known to survive for longer than three weeks (Jones & Orr 1994).

ARMCANZ and the NHMRC have established a working group as part of the National Algal Management Strategy to examine the issue of guidelines for cyanobacteria and cyanobacterial toxins in surface waters (including waters used for drinking, recreation and irrigation).

9.2.2.3 Human and animal pathogens

Trigger values for thermotolerant coliforms in irrigation waters are provided in table 9.2.2.

Table 9.2.2 Trigger values for thermotolerant coliforms in irrigation waters used for food and non-food crops[a]

Intended use	Level of thermotolerant coliforms[b]
Raw human food crops in direct contact with irrigation water (e.g. via sprays, irrigation of salad vegetables)	<10 cfu[c] / 100 mL
Raw human food crops not in direct contact with irrigation water (edible product separated from contact with water, e.g. by peel, use of trickle irrigation); or crops sold to consumers cooked or processed	<1000 cfu / 100 mL
Pasture and fodder for dairy animals (without withholding period)	<100 cfu / 100 mL
Pasture and fodder for dairy animals (with withholding period of 5 days)	<1000 cfu / 100 mL
Pasture and fodder (for grazing animals except pigs and dairy animals, ie cattle, sheep and goats)	<1000 cfu / 100 mL
Silviculture, turf, cotton, etc (restricted public access)	<10 000 cfu / 100 mL

a Adapted from ARMCANZ, ANZECC & NHMRC (2000)

b Median values (refer to discussion on derivation of guidelines below)

c cfu = colony forming units

Description

The presence of human and animal pathogens in irrigation waters is becoming an important issue in agricultural water quality management, particularly with the overall trend towards decreasing water quality and the increasing reuse of municipal and agricultural wastewaters for irrigation of crops and pastures. Potential pathogen contamination of natural waters is also of increasing concern, emphasising the need to take a holistic approach to water quality management in catchments so that the quality of water is maintained for downstream users.

Limited information is currently available on the behaviour of human and animal pathogens that may be present in irrigation waters and their expected survival rate under varying environmental conditions. However, it is generally recognised that a number of factors can influence pathogen levels (WHO 1981) including:

- present human and animal health status;
- water quality;
- soil characteristics;
- temperature, humidity, precipitation, solar radiation;

- the nature of agriculture and animal husbandry;
- method of transportation of irrigation water;
- irrigation method; and
- treatment and storage for pathogen die-off.

Water-borne pathogens of concern to human and animal health in Australia comprise a range of micro-organisms including bacteria, viruses, protozoa and helminths (DEST State of the Environment Advisory Council 1996). Many of these are known to exist in agricultural wastewaters and some can withstand conventional methods of treatment. Pathogens can be transmitted to human and animal consumers via irrigation water through direct contact of the water with the surface of edible produce. It is generally considered that there is little likelihood that pathogens are translocated internally through crop plants to affect edible portions not directly exposed to irrigation water (USEPA 1992). Pathogens transported via aerosols in spray irrigation may also present an infection risk to individuals downwind.

Bacteria are the group of pathogens most sensitive to environmental conditions. Pretreatment of irrigation water using standard disinfection techniques will normally reduce bacterial populations substantially. Exposure to drying, extremes in pH, solar irradiation and competition from soil bacteria after irrigation also greatly reduce populations (Crane & Moore 1986). Some bacteria are known to survive for prolonged periods on plant surfaces if protected from these factors, e.g. in split or cracked vegetative surfaces (Bell & Bole 1976).

Viruses consist of a strand of genetic material with a protein coat. They act by invading the host cell, subsequently modifying its behaviour to produce more viral particles (Metcalf & Eddy 1991). Pathogenic viruses occur in natural waters largely as a result of contamination with sewage and animal excreta (NHMRC & ARMCANZ 1996), and should not normally be present in irrigation waters in large numbers.

Problems can occur if viruses are present in irrigation waters used on crops directly for human consumption, as they have been known to persist on vegetation for several weeks or months. Bagdasaryan (1964) found that enteroviruses survived on vegetables that were kept in a household refrigerator (6–10°C) for 10 days or longer. Virus retention and survival on vegetables has been shown to depend on the type of vegetable material (fruit or leaves) and on the type of virus (Ward et al. 1981). Survival is also influenced by temperature, solar radiation, wind, rainfall, humidity, concentration in water and irrigation method. There is some evidence that viruses may persist for longer in soil than on aerial vegetable surfaces (Gerba et al. 1978).

Protozoa are single-celled micro-organisms without cell walls and include amoebas, flagellates and ciliates. This group is responsible for the majority of dysentery and diarrhoea related illnesses in humans and includes two pathogenic organisms of particular concern, *Giardia* and *Cryptosporidium*. Both pathogens are able to survive conventional wastewater treatment (Rose & Gerba 1991). Results of a Californian study indicated that wild pig populations may act as a reserve of these two protozoa in the environment (Atwill et al. 1997).

To protect themselves from adverse environmental conditions, protozoa often form cysts that can be transmitted through irrigation of contaminated water. This enables the survival of some species for extended periods of time on crops and pastures. Irrigated vegetables and fruit have been implicated in the transmission of several protozoan infections to humans (Froese & Kindzierski 1998).

A number of species of parasitic helminths are endemic in some parts of Australia (ARMCANZ, ANZECC & NHMRC 2000) and eggs of a variety of helminths can be transmitted via irrigation water to crops or pastures.

Effects on human health

Human pathogens that could potentially be found in irrigation waters and wastewaters and their associated health risks are listed in table 9.2.3 (Metcalf & Eddy 1991). Issues concerning livestock health are discussed separately (see Section 9.3.3.2). Protection of human and animal health from disease associated with pathogens in irrigation water is based on providing barriers to disease transmission to minimise exposure (NHMRC & ARMCANZ 1996, ARMCANZ, ANZECC & NHMRC 2000).

Table 9.2.3 Pathogens found in irrigation waters and wastewaters that may adversely affect human health[a]

Organism	Disease	Remarks
Bacteria		
Escherichia coli	Gastroenteritis	Diarrhoea
Legionella pneumophila	Legionellosis	Acute respiratory illness
Leptospira	Leptospirosis	Jaundice, fever
Salmonella sp.	Salmonellolis	Food poisoning
Salmonella typhi	Typhoid fever	Diarrhoea, fever
Shigella	Shigellosis	Bacillary dysentery
Vibrio cholerae	Cholera	Diarrhoea, dehydration
Yersinia enterolitica	Yersinosis	Diarrhoea
Helminths		
Ascaris lumbricoides	Ascariasis	Roundworm infestation
Enterobius vericularis	Enterobiasis	Pinworm
Fasciola hepatica	Fascioliasis	Sheep liver fluke
Hymnolepis nana	Hymenolepiasis	Dwarf tapeworm
Taenia saginata	Taeniasis	Beef tapeworm
Taenia solium	Taeniasis	Pork tapeworm
Trichuris trichiura	Trichuriasis	Whipworm
Protozoa		
Balantidium coli	Balantidiasis	Diarrhoea, dysentery
Cryptosporidium	Cryptosporidiosis	Diarrhoea
Entamoeba hystolica	Amebiasis	Amoebic dysentery
Giardia lamblia	Giardiasis	Diarrhoea, nausea
Viruses		
Adenovirus	Respiratory disease	
Enteroviruses	Gastroenteritis, meningitis	
Hepatitis A	Infectious hepatitis	Jaundice, fever
Norwalk agent	Gastroenteritis	Vomiting
Reovirus	Gastroenteritis	
Rotavirus	Gastroenteritis	

a From Metcalf & Eddy (1991)

Derivation of guidelines

Expanding interest worldwide in the use of reclaimed wastewaters for irrigation of crops and pastures has generated much of the recent activity in developing guidelines for their safe use for this and other purposes. Although the present guidelines concern naturally occurring waters rather than reclaimed waters, the underlying issues regarding risks to human and animal health are the same.

It is generally not feasible nor warranted to test irrigation water for the presence of the wide range of water-borne microbial pathogens that may affect human and animal health. In

practice, water supplies are more commonly tested for the presence of thermotolerant coliforms (also known as faecal coliforms), to give a general indication of faecal contamination and thus the possible presence of microbial pathogens. However, note that in tropical and sub-tropical areas thermotolerant coliforms may on some occasions include micro-organisms of environmental rather than faecal origins (NHMRC & ARMCANZ 1996). Moreover, the test does not specifically indicate whether pathogenic organisms are present or not; and coliform bacteria are not considered to be reliable indicators of protozoa (Craun et al. 1997). In view of these limitations, there is increasing interest in applying risk assessment methodologies to complement monitoring programs and enhance existing guidelines (ARMCANZ, ANZECC & NHMRC 2000).

In Australia and New Zealand, the management and use of reclaimed water from sewerage systems forms an important component of the National Water Quality Management Strategy. Guidelines for pathogen levels in irrigation water have been proposed in the ARMCANZ, ANZECC & NHMRC document, *Guidelines for sewerage systems — use of reclaimed water*. After consideration of the scientific literature and the issues associated with developing guidelines for pathogens (WHO 1989, USEPA 1992, Hespanhol & Prost 1994), the ARMCANZ, ANZECC & NHMRC (2000) guidelines for pathogens in irrigation water have been adopted for use in the present water quality guidelines.

It is recommended that a median value of thermotolerant coliforms be used, based on a number of readings generated over time from a regular monitoring program. Investigations of likely causes are warranted when 20% of results exceed four times the median guideline level (ARMCANZ, ANZECC & NHMRC 2000).

For helminths, a trigger value of ≤1 helminth egg per litre is proposed for the protection of crop consumers in areas where helminth infections are known to be endemic. A lower value of 0.5 eggs per litre may be required to protect farm workers and their families directly exposed to the water (Blumenthal et al. 1996). Insufficient information is available to set guidelines for protozoa and viruses in irrigation water.

The ARMCANZ, ANZECC & NHMRC (2000) guidelines for pathogens in irrigation water were proposed after consideration of the methodologies and information used in developing guidelines by the World Health Organisation (WHO 1989) and the United States Environmental Protection Agency (USEPA 1992), together with local considerations. This is consistent with WHO recommendations that the WHO (1989) guidelines be adapted according to local conditions and socio-economic factors (Hespanhol & Prost 1994). The ARMCANZ, ANZECC & NHMRC (2000) guidelines are based on:

- the best available scientific evidence;
- worldwide practice in reclaimed water use;
- a consensus of local practice demonstrated to be safe.

The WHO (1989) guidelines recommend upper limits of 1000 faecal coliform cells per 100 mL, one or less helminthic egg per litre and one or less protozoan parasite cyst per litre for crops likely to be eaten uncooked. Proposed guidelines for faecal coliforms in the USA are considerably more conservative and include: no detectable faecal coliforms per 100 mL for surface or spray irrigation of food crops not commercially processed (including crops eaten raw); and ≤200 faecal coliforms per 100 mL for irrigation of commercially processed food crops, surface irrigation of orchards and vineyards, and irrigation of pasture, fodder, fibre and seed crops (USEPA 1992).

9.2.2.4 Plant pathogens

No trigger values for plant pathogens in irrigation waters are recommended at this time. As a general precaution, disinfestation treatment is advisable for water that contains plant pathogens and is to be used for irrigating potentially susceptible plants.

Description

Agricultural crops and pastures can be affected by various plant pathogens transmitted through a number of different pathways including irrigation water. Although limited research has been conducted into acceptable levels of plant pathogens in irrigation water used for agricultural purposes, it is believed that the risk of transmission through this method is low under most circumstances (Hagan et al. 1967, CCREM 1987). However, plant pathogens in irrigation water used for intensive agricultural and horticultural industries (particularly where wastewaters are reused), can potentially lead to crop damage and economic loss.

Although variations exist in the range of environmental conditions suitable for different plant pathogens, most require atmospheric conditions with high humidity and the presence of free water for a prolonged period of time. In general, free water must usually be present for at least 6 to 12 hours to allow infection to occur (Menzies 1967). Infection can then be further transmitted by splashing water, which can loosen spores from infected soil and plant surfaces and spread them to other plants in the vicinity (Hagan et al. 1967).

A great deal of work needs to be done before guidelines can be developed, particularly concerning the efficacy of water-borne plant pathogens on a wide range of crops.

Effect on irrigation water quality

Plant pathogens of potential concern to irrigation water quality commonly include viruses, fungi and bacteria (K Bodman, pers comm). Examples of some plant pathogens that may be present in recycled irrigation water are given in table 9.2.4 (Dutky 1995).

Table 9.2.4 Examples of plant pathogens found in irrigation water[a]

Viruses	Tomato mosaic virus
	Cucumber green mottle virus
	Pelargonium flower break virus
	Carnation mottle virus
Fungi	*Phytophthora cryptogea*
	Phytophthora nicotianae
	Plasmopara lactucae-radicis
	Pythium aphanidermatum
	Pythium dissotocum
	Pythium intermedium
	Pythium myriotylum
	Fusarium oxysporum
Bacteria	*Pseudomonas solanacearum*
	Xanthomonas campestris

a After Dutky (1995)

Some species of nematodes causing plant damage are also believed to be potentially transmitted in irrigation waters, although limited research has been conducted in this area. Root rot is a major pathogenic problem, which can be caused by a variety of pathogens (usually fungal) that live freely in water, such as *Phytophthora*, *Pythium* and *Olpidium* sp (Rolfe et al. 1994). *Phytophora* species are responsible for significant economic losses in horticultural, ornamental and pasture crops in Australia (Cahill 1993), with an estimated direct loss from these pathogens in 1991–92 of at least $223 million (James et al. 1996).

Plants can exhibit a number of symptoms in response to pathogenic infection, including: over development of plant tissue (e.g. galls, swellings and leaf curls), underdevelopment of plant tissue (e.g. stunting, lack of chlorophyll and incomplete development of organs), and death of plant tissue (University of Nebraska 1997).

9.2.3 Salinity and sodicity

To assess the salinity and sodicity of water for irrigation use, a number of interactive factors must be considered. As outlined in this Section, these include: irrigation water quality; soil properties; plant salt tolerance; climate; landscape (including geological and hydrological features); and water and soil management.

9.2.3.1 Description

Salinity is the presence of soluble salts in or on soils, or in waters. High levels of soluble salts in soils may result in reduced plant productivity or the elimination of crops and native vegetation. Elements forming salts derive from the weathering of the earth's crust and are transported and cycled through rainfall and the movement of water. When the hydrologic balance of a landscape is altered through natural processes or human induced disturbances, a new hydrologic equilibrium is established with a subsequent translocation of salt to the soil or water environment.

Under general irrigation practice, the addition of water can result in the physical rise of the watertable underlying the land under irrigation, creating potential waterlogging and shallow watertables, if soil conditions are appropriate. Off-site degradation of surface or groundwaters may also occur. If elevated levels of salt are present in irrigation waters and/or the soil profile, accumulation of salts can lead to reduced crop yield and land degradation.

This process of salt accumulation is referred to as salinisation and is a major concern in the degradation of agricultural lands in Australia. In the Murray–Darling Basin in 1987, it was estimated that 96 000 hectares of the irrigated land was salt-affected and 560 000 hectares had water tables within two metres of the land surface. In addition, data from the Salinity Audit report in 1999 indicates that 116 000 hectares of non-irrigated land in South Australia and 840 000 hectares in the Victorian Section of the Murray–Darling Basin is likely to be salt-affected by the year 2050 (MDBC 1999). In Western Australia, 1.8 million hectares were estimated to be affected by salinity in 1998 and this could double again before equilibrium is reached. Salinisation problems related to both dryland and irrigation practices have now been observed in all Australian States and Territories. In New Zealand, irrigation is associated with changes in land use from dryland stock systems to much more intensive horticultural crop and animal systems. Since most irrigation is on recent soils on very deep alluvial gravels without a watertable, salinisation is currently not widespread.

Sodicity is a condition that degrades soil properties by making the soil more dispersible and erodible, restricting water entry and reducing hydraulic conductivity (the ability of the soil to

conduct water). These factors limit leaching so that salt accumulates over long periods of time, giving rise to saline subsoils. Furthermore, a soil with increased dispersibility becomes more susceptible to erosion by water and wind.

A high proportion of sodium in soil can result in dispersion and, once dry, soils may become dense, cloddy and structureless, destroying natural particle aggregation. Because the relative proportions of exchangeable cations in a given soil are determined by the relative concentration of cations in the soil solution, the composition of irrigation water can influence soil sodicity (Rengasamy & Olsson 1995).

9.2.3.2 Factors affecting irrigation salinity

The extent and effect of irrigation water salinity on land under irrigation is dependent on a variety of interactive factors:

- irrigation water quality
- soil properties
- plant salt tolerance
- climate
- landscape
- irrigation management practices.

These are discussed separately within this Section to allow a better understanding of the processes involved. The methodology presented here attempts to assess a realistic irrigation environment more accurately, through increased emphasis on the role of soil properties in sustainable irrigation. Figure 9.2.1 illustrates the interactions of various processes in relation to salinity and sodicity.

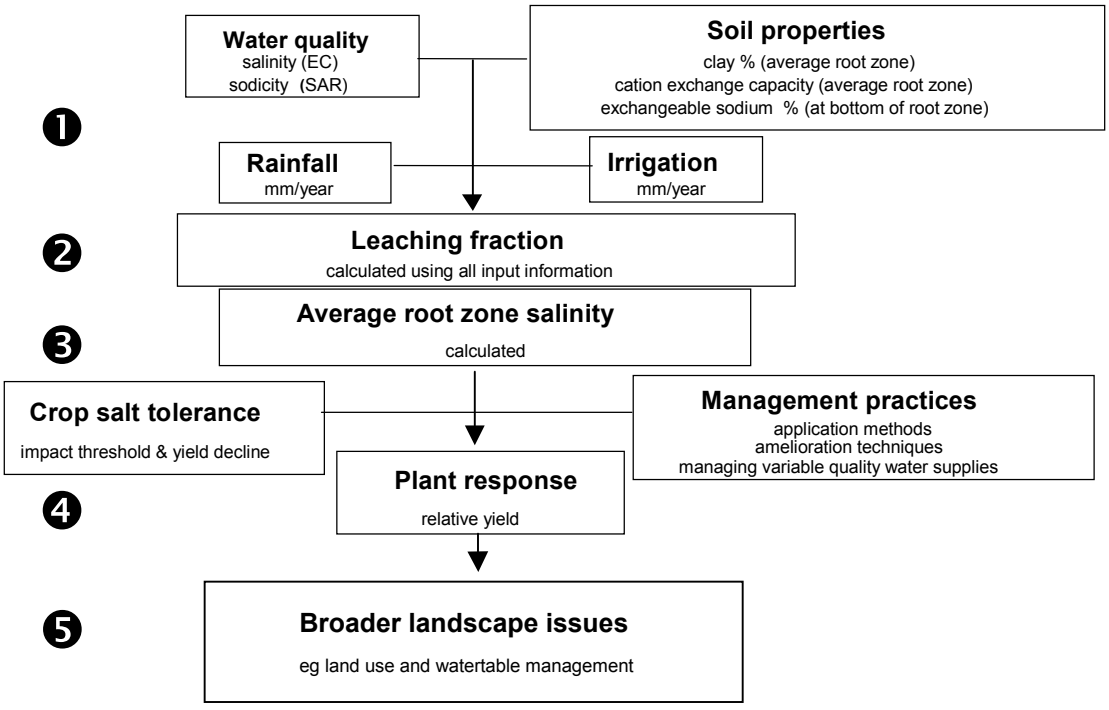

Figure 9.2.1 Flow diagram for evaluating salinity and sodicity impacts of irrigation water quality

There are five key steps to determining the suitability of irrigation water with respect to salinity and sodicity (fig 9.2.1).

Step 1 Identify the soil properties, water quality, climate (rainfall) and management (irrigation application rates) practices for the site in question.

Step 2 Estimate the leaching fraction under the new/changed irrigation regime utilising approaches outlined in this Section.

Step 3 Estimate the new average root zone salinity as outlined in this Section. Average root zone salinity is considered the key limitation to plant growth in response to salinity and sodicity levels in irrigation water. However, poor soil structure can also reduce plant yields by limiting aeration, water infiltration and root growth. The likelihood of soil structural problems induced by irrigation can be predicted from trigger values derived in this Section.

Step 4 Estimate relative plant yield (although note that the impact of salinity and sodicity can be modified by management practices as discussed later in this Section).

Step 5 Consider salinity and sodicity problems within the framework of broader catchment issues such as regional watertables, groundwater pollution and surface water quality. Watertable salinity develops in response to excess water and salts accumulating in sensitive parts of the landscape. Excess water can percolate to the groundwater as a result of changing climatic patterns (e.g. frequency and duration of rainfall events), land use or land management (including irrigation). Before an irrigation scheme is developed, the planning process should include investigation of the regional hydrogeology to avoid development of watertable salinity. The guidelines given here concentrate on localised impacts of irrigation, but broader salinity issues should not be ignored.

The details of this methodology follow, with some worked examples provided later in Section 9.2.3.3. Software *SALF PREDICT* is provided on CD-ROM to estimate the parameters necessary for a detailed assessment of irrigation water quality in relation to soil properties, rainfall, water quality and plant salt tolerance. The software is based on summer rainfall areas and needs to be used with some caution in winter rainfall areas. It incorporates many of the detailed algorithms included in this Section. Copies of the software may also be obtained from the Queensland Department of Natural Resources.

Irrigation water quality

Salinity assessment

To assess irrigation water quality with respect to salinity, the salt content or electrical conductivity of the water must be known. Electrical conductivity (EC) measures the ability of water to conduct an electric current, which is carried by various ions in solution such as chloride, sodium, sulfate, nitrate, carbonate, bicarbonate, calcium and magnesium. Electrical conductivity is commonly used as an estimate of the concentration of total dissolved salts (TDS) and is measured in decisiemens per metre (dS/m) or microsiemens per centimetre (µS/cm). Units of dS/m are used throughout these guidelines for irrigation water quality. One dS/m is equivalent to one thousand µS/cm. Because different conversion factors are used to convert EC to TDS, it is recommended that only directly analysed EC data be used.

A preliminary water salinity rating can be assigned to irrigation waters based on EC (table 9.2.5). These ratings provide only a general guide and are not intended to be used on their own to define the suitability of irrigation water. As emphasised throughout this

Section, other factors such as soil characteristics, climate, plant species and irrigation management must be considered.

Table 9.2.5 Irrigation water salinity ratings based on electrical conductivity[a]

EC (dS/m)[b]	Water salinity rating	Plant suitability
<0.65	Very low	Sensitive crops
0.65–1.3	Low	Moderately sensitive crops
1.3–2.9	Medium	Moderately tolerant crops
2.9–5.2	High	Tolerant crops
5.2–8.1	Very high	Very tolerant crops
>8.1	Extreme	Generally too saline

a Adapted from DNR (1997); b 1dS/m = 1000 μS/cm

The primary purpose of measuring the EC of irrigation water (EC_{iw}) is to calculate the average root zone salinity (EC_{se}), one of the critical measurements used in salinity assessment and the evaluation of plant salt tolerance (see later discussion on soil properties).

Sodicity assessment

Sodicity is the presence of a high proportion of sodium (Na^+) ions relative to other cations in soil (in exchangeable and/or soluble form) or water. The presence of Na^+ salts in soil, which can lead to soil salinity, can also act as a coagulant or flocculant of soil particles. However, Na^+ as an exchangeable cation acts as a dispersant.

Elevated levels of Na^+ in irrigation water can lead to sodicity problems in the soil profile under irrigation. An estimation of sodicity levels in irrigation water can be predicted using the sodium adsorption ratio (SAR). This is calculated using the following equation, where ionic concentrations are in $mmole_c/L$:

$$SAR = \frac{Na^+}{\left[\frac{\left(Ca^{2+} + Mg^{2+}\right)}{2}\right]^{0.5}} \qquad (9.1)$$

The SAR value can be used to predict permissible sodicity levels in irrigation water to maintain soil structural stability. Clay mineralogy data (usually expressed as CCR in $mmole_c/kg$) is related to soil texture or clay content for the soil under irrigation, as shown in table 9.2.6, to determine a soil stability response to SAR. A sustainable SAR value can then be approximated. As the salt concentration in irrigation water can act as a flocculant, both EC and SAR need to be considered in the final assessment of water quality suitability.

Soil properties

Soil properties are major factors affecting irrigation salinity assessment. Soil salinity and sodicity can be predicted using empirical relationships between readily measured soil properties and leaching (adjusted for changes resulting from irrigation water salinity and sodicity), taking into consideration rainfall effects. Estimation of equilibrium soil salinity and sodicity values can then be calculated, assuming a steady state mass balance approach. This is the essence of steps 1–3 of figure 9.2.1.

Table 9.2.6 Guide to permissible SAR of irrigation water for maintaining a stable soil surface under high rainfall[a,b]

Clay content (%)	Soil texture	Permissible irrigation water SAR				
		Clay mineralogy expressed as CCR ($mmole_c$ /kg)[c]				
		<0.35	0.35–0.55	0.55–0.75	0.75–0.95	>0.95
<15	Sand, sandy loam	>20	>20	>20	>20	>20
15–25	Loam, silty loam	20	11	10	10	8
25–35	Clay loam	13	11	8	5	6
35–45	Light clay	11	8	5	5	5
45–55	Medium clay	10	5	5	5	5
55–65	Medium-heavy clay	5	5	5	4	4
65–75	Heavy clay	–	4	4	4	4
75–85	Heavy clay	–	–	4	5	5

a From DNR (1997); b These values are based on the prediction of the leaching fraction model under a high rainfall of 2000 mm/year, to estimate the soil ESP and hence SAR that should prevent surface soil dispersion; c CCR: cation exchange capacity/clay ratio

Soil salinity assessment

Leaching fraction (LF) and EC are two critical measurements concerning soil salinity assessment. These parameters form the basis of predicting soil root zone salinity (EC_{se}) and plant response, from which a sustainable irrigation management strategy can be determined. Methodologies are outlined in the following Sections; a more detailed discussion can be found in Shaw (1994).

Leaching fraction

Leaching fraction is defined as the proportion of applied water (irrigation + rainfall) that drains below the root zone in the soil profile (see fig 9.2.2), expressed as a percentage.

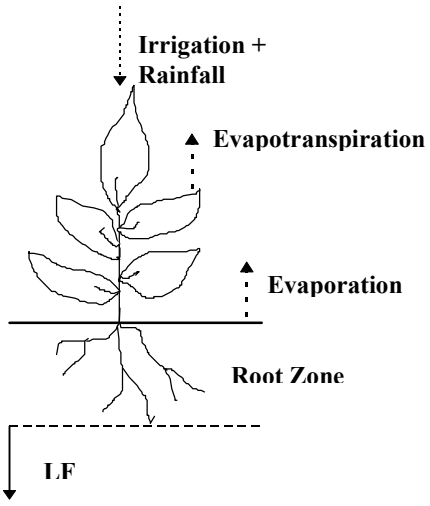

Figure 9.2.2 Schematic representation of leaching fraction

Prediction of LF is fundamental to irrigation salinity assessment, in particular to the estimation of root zone salinity and consequent effects on plant production. A simple empirical approach estimates LF based on a steady-state salt mass balance, which assumes that an equilibrium occurs between the inputs and the outputs of salt after a given period, where the change in salt storage becomes zero.

Leaching fraction is estimated most easily by calculating the ratio of inputs (rainfall + irrigation) to outputs. On the assumption that the water draining at the bottom of the root zone is equivalent to the soil matrix salinity (in most cases), then LF can be estimated as follows:

$$LF = \frac{EC_i}{EC_d} = \frac{D_d}{D_i} \tag{9.2}$$

where:

EC_i = electrical conductivity of water entering soil in dS/m (rainfall + irrigation) *(known)*

EC_d = electrical conductivity of drainage water below the root zone in dS/m *(measurable)*

D_i = depth of water applied to the soil profile in mm/year (rainfall + irrigation) *(known or estimated)*

D_d = depth of water draining below the root zone in mm/year *(predicted)*

EC_i can be calculated (on an annual basis) using the following relationship:

$$EC_i = \frac{(EC_r \text{ x } D_r) + (EC_{iw} \text{ x } D_{iw})}{(D_r + D_{iw})} \tag{9.3}$$

where:

EC_i = electrical conductivity of water entering soil in dS/m (rainfall + irrigation)

EC_r = electrical conductivity of rainfall, taken to be 0.03 dS/m (unless measured locally)

EC_{iw} = electrical conductivity of irrigation water in dS/m

D_r = rainfall depth in mm/year

D_{iw} = depth of irrigation water applied to the soil profile in mm/year

Note in the above that 1 dS/m = 1000 μs/cm.

LF prediction models

Irrigation water salinity can be predicted using several approaches as outlined in table 9.2.7.

Table 9.2.7 Summary of methods and data required for estimating leaching fraction for different conditions[a]

Condition/model	Field data	Method
Prior to irrigation Shaw & Thorburn (1985) Shaw (1996)	(a) EC and amount of irrigation water, rainfall, $EC_{1:5}$ at bottom of root zone, maximum field water content	Leaching fraction predicted from soil properties and water application.
	(b) Clay and CEC from 0 to 0.9 m, ESP at 0.9 m, annual rainfall and irrigation and EC of irrigation.	Adjustments are necessary for winter rainfall areas.
Long-term irrigation (steady state: 5–10 years) USSL (1954)	EC and amount of irrigation water, rainfall, $EC_{1:5}$ at bottom of root zone, maximum field water content	$EC_{1:5}$ converted to EC_s and leaching fraction calculated.
Short term irrigation (non-steady state) Rose et al. (1979) Thorburn et al. (1987)	Cl of irrigation water, $Cl_{1:5}$ profiles taken at two times, depth of rainfall and irrigation, maximum field water content	Leaching fraction calculated utilising the SODICS model.
Changed irrigation water salinity Shaw & Thorburn (1985), Shaw (1996)	As for (a) and (b) above, plus quantity and EC of past and future irrigation water, annual rainfall	Leaching fraction predicted from soil properties and water salinity.

a Discussed further in the following sections

These methods are based on empirical relationships between readily measured soil properties and soil leaching. Soil leaching is adjusted for changes resulting from irrigation water salinity and sodicity. Rainfall is explicitly incorporated in its effect on water composition and soil leaching behaviour from which equilibrium soil salinity and sodicity values are estimated, assuming a steady-state mass balance approach.

- Prior to irrigation

To assess the suitability of land for irrigation, it is necessary to predict the LF value that will occur under irrigation. Shaw and Thorburn (1985) and Shaw (1996) developed a method for directly predicting the LF at the bottom of the root zone that would occur under irrigation.

The soil properties of dominant influence on soil leaching are clay content, clay mineralogy (CCR) (expressed as CEC/clay ratio, $mmole_c/kg$ of clay), and the exchangeable sodium percentage (ESP), which is the exchangeable sodium content of soil expressed as a percentage of the cation exchange capacity (CEC). As a result of the relationship between soil properties, ESP and rainfall are specified for different soil groups across a wide range of rainfalls. Leaching fraction under irrigation can then be calculated by substituting the depth of irrigation plus rainfall D_{i+r}, for D_r. Because a change in electrolyte concentration will result in a change in leaching for a given soil ESP, an adjustment of the predicted leaching fraction is made where the irrigation water has a high SAR, the ESP at the bottom of the root zone is also adjusted.

LF_r (expressed as a percentage) is the predicted LF under rain-fed conditions and is calculated from the general equation for each soil group (as given in table 9.2.8) using the general form:

$$LF_r = \frac{EC_r}{EC_d} \tag{9.4}$$

where EC_d can be approximated by 2.2 x EC_{se} (the EC of the soil saturation extract). EC_{se} can be predicted utilising information on soil properties (Shaw 1996) giving the following equation:

$$LF_r = \frac{EC_r}{2.2 \times 10^{\left[a + b \log\left(\frac{0.03 \times rainfall}{ESP}\right)\right]}} \tag{9.5}$$

ESP of the soil under irrigation can be calculated following the procedure discussed later in the Section on soil sodicity. The coefficients ‘a’ and ‘b’ are provided in table 9.2.8. The value 2.2 represents the generalised relationship between water content of the soil saturation extract and maximum water content.

In the case of differing salt contents of irrigation waters, the following equation can be used:

$$LF_i = LF_r \left(\frac{EC_i}{EC_r}\right) \tag{9.6}$$

where:

LF_i = predicted leaching fraction under irrigation (expressed as a percentage)

LF_r = predicted leaching fraction under rainfall (expressed as a percentage)

EC_i = weighted EC of input water for irrigation (i) and rainfall (r) (in dS/m)

EC_r = 0.03 dS/m

Table 9.2.8 Parameters used in equation 9.5 to estimate LF under irrigation[a]

Clay content range	Parameter	CCR <0.35	CCR 0.35–0.55	CCR 0.55–0.75	CCR 0.75–0.95	CCR >0.95
(%)		($mmole_c$/kg)	($mmole_c$/kg)	($mmole_c$/kg)	($mmole_c$/kg)	($mmole_c$/kg)
5–15	a	-0.653	-0.240	-0.124	-0.115	-0.559
	b	-0.098	-0.521	-0.562	-0.506	-0.067
15–25	a	-0.011	0.330	0.440	0.479	0.295
	b	-0.593	-0.857	-0.934	-1.195	-0.671
25–35	a	0.147	0.411	0.633	0.772	0.457
	b	-0.672	-0.936	-1.032	-0.980	-0.750
35–45	a	0.438	0.706	0.827	0.831	0.663
	b	-1.036	-1.141	-1.087	-0.962	-0.897
45–55	a	0.602	0.831	0.802	0.794	0.570
	b	-1.161	-1.047	-0.971	-1.105	-0.807
55–65	a	0.802	0.812	0.870	0.783	0.613
	b	-0.888	-1.317	-1.006	-0.888	-0.588
65–75	a		0.722	0.663	0.684	0.394
	b		-0.826	-0.840	-1.109	-0.583
75–85	a			0.660	0.690	0.248
	b			-0.751	-0.872	-0.777

a From Shaw (1996)

On the basis of experience with heavy textured soils in the Lockyer Valley using variable salinity irrigation waters, and because the soil responses to salt vary with physico-chemical properties, a non-linear adjustment was developed, where the adjustment decreases with the increasing salinity of the applied water. The non-linear adjustment for salt concentration is used to predict leaching fraction for irrigating with different salinity waters.

Thus the EC ratio component of equation 9.6 is adjusted as follows:

$$LF_f = LF_i\left[2.65\left(\frac{EC_i}{EC_r}\right)^{0.5} - 1.35\right] \tag{9.7}$$

where:

LF_f = prediction of LF in the future after allowing for irrigation water quality and depth (expressed as a percentage)

EC_i = electrical conductivity of water entering soil in dS/m (rainfall + irrigation)

EC_r = electrical conductivity of rainfall = 0.03 dS/m

- Long-term irrigation (steady state)

Where soils have been under irrigation for some years, steady-state conditions should exist and the following equation is valid:

$$LF = \frac{EC_i}{EC_s} \tag{9.8}$$

where:

EC_i = electrical conductivity of water entering soil in dS/m (rainfall + irrigation).

EC_s is the equivalent of EC_d of equation 9.2 and is determined from soil EC_{se} or $EC_{1:5}$ measurements (taken at the bottom of the root zone). However, the $EC_{1:5}$ value will have to be converted to EC_s from the ratio of dilution as outlined in the following Section on calculation of EC_{se}. EC_i can be calculated (on an annual basis) using equation 9.3.

- Short-term irrigation

As irrigation changes the salt balance, soil salinity will change (increase or decrease) after the commencement of irrigation until a new equilibrium (steady state) is attained. Until this is reached, EC_s will not give an accurate indication of LF.

As an alternative, the change in soil salinity which occurs between two sampling times can be used, as illustrated by Rose et al. (1979). This model is most suited to slowly permeable soils with lengthy periods required to reach equilibrium. The data required are soil salinity profiles (preferably chloride) at two sampling times, the amount and salinity (chloride) of irrigation water used, and the maximum field water content of the soil.

Maximum water content can be measured in the field after an extended wet period, or is easily predicted from the equations of Shaw and Yule (1978) or Littleboy (1997) for most slowly permeable soils (Thorburn & Gardner 1986). The equation of Rose et al. (1979) is:

$$S_2 = S_1 + \left[\left(\frac{D_i S_i}{D_d I} - S_i\right)\lambda - S_i\right]\left[1 - \exp\left(-D_d \frac{1}{Z\theta}\right)^t\right] \qquad (9.9)$$

where S_1 and S_2 are the mean root zone salinities determined at two different times, t is the time between determinations, z is the depth of root zone, θ is the volumetric water content to which drainage will occur and λ is a factor to account for soil salinity profile shape.

The value of D_d is the only unknown in the equation and can be calculated from the model. It can be used to calculate LF and give the average root zone salinity value that will occur at that site at steady state. The model can also indicate the time period when steady state conditions will be reached, and how much salinity will increase (or possibly decrease) until that time. If the EC root zone value at steady state is too great for the crop to be grown, irrigation management practice will have to be modified.

- Changed irrigation water salinity

Shaw and Thorburn (1985) found that the change in LF between a rainfall situation and irrigation was directly related to the ratio of the weighted salinity of the irrigation water and the rainfall in the future situation, and the rainfall salinity itself. This can also be applied when changing to irrigation water of different salinity.

The relationship is:

$$LF_f = LF_p \left(\frac{EC_i}{EC_r}\right) \qquad (9.10)$$

where:

LF_f = prediction of LF in the future (expressed as a percentage)

LF_p = past LF value (expressed as a percentage)

EC_i is calculated from equation 9.3 where it represents future electrical conductivity of water entering soil in dS/m (irrigation + rainfall).

Electrical conductivity of soil

Measurement of soil salinity has traditionally been based on EC and chloride concentration determined through laboratory testing using 1:5 soil:water suspension procedure (Rayment & Higginson 1992). While this is a convenient laboratory measure of the salt content of a soil, measurements of EC at other water contents are more useful, namely EC_{se} for plant response and EC_s for salt movement.

EC_{se} (in dS/m) is defined as the electrical conductivity of the soil saturation extract, while EC_s (in dS/m) is the electrical conductivity of the soil solution at maximum field water content (note that 1 dS/m is equivalent to 1000 µs/cm). Maximum field water content is the maximum measured water content of the soil in the field, two to three days following wetting. It is expressed on a mass basis (g/100 g) and is considerably lower than the common estimate of laboratory 'field capacity' using ground samples (Gardner & Coughlan 1982).

The soil:water ratio of 1:5 was established in response to difficulties that arise when using the traditional saturation extract mixing method with heavy textured Australian soils and is a convenient laboratory and field technique. However, it is not directly related to soil behaviour and plant response, as the ratio is far more dilute than is normally found under field conditions and it is fixed irrespective of soil texture. Analysis of $EC_{1:5}$ tends to underestimate the electrical conductivity of sandy soils compared with clay soils.

Plants respond to salinity at water contents equal to or drier than saturation. The EC_{se} is the most dilute soil solution concentration that plants could be expected to encounter and has been successfully used to relate plant response to soil salinity across a wide range of soil textures. This soil water content, a well accepted standard (USSL 1954), is commonly used as it is the lowest reproducible soil water content for which enough extract can be readily removed for analysis. It also consistently relates to field soil water contents and soil textures (Rhoades 1983).

Salt movement in soils becomes limited once the soil water content is drier than maximum field water content. The salinity at this water content, EC_s, which represents the salt content at the point where soil profile drainage has effectively ceased, is used in leaching fraction estimations and in solute movement studies and modelling. Table 9.2.9 shows the relative dilutions with respect to field water contents for the three measures of EC.

Table 9.2.9 Relative dilution above maximum field water content for three measures of soil salinity, $EC_{1:5}$, EC_{se} and EC_s[a]

Measure	Dilution above field water content
$EC_{1:5}$	5 to >40 times
EC_{se}	2 to 3 times
EC_s	1 time solution

a From DNR (1997)

- Calculation of EC_{se}

There are two methods that can be used to calculate EC_{se}. The first method is based on the EC_{iw} value obtained from the analysis of irrigation water. This method provides an approximate estimate of EC_{se} using predicted leaching fraction (LF) of the soil under irrigation.

1. Converting EC_{iw} to EC_{se}

The equation to calculate EC_{se} for the average root zone using this method is:

$$EC_{se} = \frac{EC_{i\,w}}{2.2 \text{ x } LF_{av}} \tag{9.11}$$

where:

EC_{se} = average root zone salinity (in dS/m) (Note that 1 dS/m = 1000 µS/cm)

EC_{iw} = electrical conductivity of irrigation water (in dS/m) and

LF_{av} = $(0.976\ LF + 0.282)^{0.625}$; expressed as a percentage (9.12)

where:

LF is calculated from the appropriate model as previously discussed.

This is based on the relationships of Rhoades (1982) and Shaw et al. (1987).

The EC_{se} value can then be used to match plant species to a particular irrigation situation as described in the following discussion on plant salt tolerance and table 9.2.10.

2. Converting $EC_{1:5}$ to EC_{se}

As discussed previously, $EC_{1:5}$ is commonly used for routine salinity appraisal and is a convenient method for estimating soil salt content. This value can be used to more accurately predict average soil root zone salinity, EC_{se}, using a model developed by Shaw (1994). The derivation of this EC conversion model is based on the conservation of mass equation at equilibrium. A given mass of dissolved salt in a system at two water contents is represented by:

$$Q_{se}EC_{se} = Q_{1:5}EC_{1:5} \tag{9.13}$$

This equation can be then rearranged as:

$$EC_{se} = EC_{1:5}\left(\frac{Q_{1:5}}{Q_{se}}\right) \tag{9.14}$$

where:

Q_{se} = water content equivalent to soil saturation or saturation percentage (SP)

EC_{se} = electrical conductivity of salt solution at the water content Q_{se} in dS/m

$Q_{1:5}$ = water content at equivalent 1:5 soil water suspension

$EC_{1:5}$ = electrical conductivity of salt solution at 1:5 soil water dilution

The saturation percentage of a soil is equivalent to saturation water content.

A number of constraints exist with practical applications of a simple water content ratio conversion. These are discussed in detail in Shaw (1994).

- Saturation water content is not unique, varying with methodology and in conversions. Using the above method, saturation water content would have to be predicted from other soil properties such as air dried soil moisture and clay contents.
- Soils contain salts of varying solubility. Calcium sulfate (gypsum), sodium carbonate and bicarbonate, and calcium carbonate are more soluble in dilute solutions, and their solubility depends on the composition of other salts present (e.g. gypsum is more soluble

if sodium chloride is present and less soluble if calcium chloride is present). Hence the composition of salts is important in a 1:5 soil:water suspension.

- In some cases where clay remains in suspension, the charge carried by the clay contributes to $EC_{1:5}$. This is not taken into consideration in EC_{se}, which is measured on extracts without any clay contribution.
- The increase in dilution ratio results in ion exchange with a preference for monovalent ions such as sodium on the exchange complex. This creates a sink for calcium, resulting in slightly enhanced solubility of calcium salts at greater dilutions.
- As a solution becomes more concentrated, dissolved ions pair together forming neutral ion pairs such as calcium sulfate. Since these ion pairs do not conduct an electrical current, the EC at high concentrations of salts that form ion pairs is reduced. Thus the direct conversion of $EC_{1:5}$ to EC_{se} may overestimate EC_{se} at high salinity levels.

Therefore, to accurately estimate EC_{se} from $EC_{1:5}$, the above factors must be taken into consideration. This can be done by adding a power term *b* to the water content ratio term which takes into account the solubility effects of different salt concentrations and compositions and the effect of suspended clays.

The equation then becomes:

$$EC_{se} = EC_{1:5}\left[\frac{Q_{1:5}}{Q_{se}}\right]^{b} \qquad (9.15)$$

where $Q_{1:5}$ is the water content of the 1:5 mixture and Q_{se} is the saturated soil water content.

$EC_{1:5}$ can be estimated for a series of relationships with soil properties including air dry moisture content (ADMC) as shown below. EC_{se} can also be estimated, an example is shown below in equation 9.16. The *b* coefficient is derived from the ratio of the non-chloride and chloride salts (based on the chloride analysis of a 1:5 soil: water extract and related to EC using the equation of USSL (1954) and McIntyre (1980).

Based on these relationships, Shaw (1996) developed the following equation to more accurately predict EC_{se}:

$$EC_{se} = EC_{1:5}\left(\frac{500 + 6ADMC}{6.57\,ADMC + 30.34}\right)\left[\frac{1}{1.024 + 0.232\ln\left(\frac{EC_{1:5}}{10^{0.92\log(56.42Cl\%)-0.865}}\right)}\right] \qquad (9.16)$$

where:

ADMC = air dry moisture content, defined as the water content between air dry 40°C and 105°C expressed as a percentage of the oven dry soil weight (g/100 g).

The EC_{se} value is then used to select the appropriate plant species to match soil conditions.

- Calculation of EC_S

EC_s is approximately two times the EC_{se} value for most soils, therefore the following equation is applicable:

$$EC_s = 2.2 \text{ x } EC_{se} \qquad (9.17)$$

Soil sodicity assessment

Two common methods of measuring soil sodicity are:

- exchangeable sodium percentage (ESP), being the proportion of sodium adsorbed onto the clay mineral surfaces as a proportion of the total cation exchange capacity (CEC, the ability of soil particles to adsorb cations); and
- sodium adsorption ratio (SAR), being the relative concentration of sodium to calcium and magnesium in the soil solution.

Exchangeable sodium percentage (ESP)

ESP is determined by routine CEC and exchangeable cation methods as outlined by Bruce and Rayment (1982) and Rayment and Higginson (1992). It is traditionally calculated using the following equation:

$$\text{ESP} = \frac{\text{Na x 100}}{\text{CEC}} \tag{9.18}$$

where:

Na = ionic concentration of Na^+ in $mmole_c$ /100 g

CEC = cation exchange capacity of the soil in $mmole_c$ /100 g

In the absence of CEC data, the sum of the exchangeable cations sodium (Na), calcium (Ca), magnesium (Mg) and potassium (K) can be used as an approximation of CEC, except:

- in acid soils, unless exchange acidity has been determined (Rayment & Higginson 1992) where an overestimate of ESP will occur from summation of cations;
- in alkaline soils where Tucker's solution at pH 8.4 (Rayment & Higginson 1992) has not been used to extract cations, sparingly soluble Ca salts will give inflated Ca and hence an underestimate of ESP.

In some variable charge soils (usually acid soils), the CEC measured by the above method may be an overestimate due to pH-dependent charge, and an underestimate of ESP may occur.

The SAR of soil solution or irrigation water can be used to predict soil sodicity response to irrigation or changes in ESP.

Predicting changes in ESP

Sodium in waters and in the soil solution is usually expressed as SAR because of its close relationship with the ESP of the soil. The proportions of Ca, Mg and Na ions on the soil exchange are not identical to the proportions in the soil solution because the divalent cations (Ca and Mg) are preferentially adsorbed onto the clay exchange surfaces. ESP can be calculated from SAR using the following relationship (USSL 1954), which has been found to provide practical predictions in many situations, including Australian soils under irrigation (Skene 1965).

$$\text{ESP} = \frac{100(-0.0126 + 0.01475\text{SAR})}{1 + (-0.0126 + 0.01475\text{SAR})} \tag{9.19}$$

The reverse equation for obtaining SAR from ESP based on the regression of the original USSL (1954) data is as follows; the equation is valid for ESP values between 0 and 50.

$$\text{SAR} = 0.6906\,\text{ESP}^{1.128} \quad (R^2 = 0.888) \tag{9.20}$$

While changes in the soil salt content under irrigation are reasonably rapid for the surface 0.1 m (occurring in a matter of months), changes in cation exchange composition in the subsoil may take many years to come to equilibrium. The rate of change is proportional to the quantity of salts added. For example, an application of 530 mm/yr of an irrigation water with an EC of approximately 5 dS/m to a clay soil with a CEC of 50 $mmole_c$/100 g would contribute an additional 6 percent of cations to the exchange complex in the top 0.6 m of soil each year.

Predicting changes in SAR

The SAR of an irrigation water provides an indication of the effect the water is likely to have on a soil. A number of factors influence the relationship between ESP and SAR. In particular, the proportion of bicarbonate and calcium ions can result in the precipitation of calcium carbonate, removing calcium from the system. Also, with depth in the root zone, the soil solution is concentrated by root water extraction, resulting in precipitation of the less soluble salts. However, the partial pressure of carbon dioxide is higher in the root zone due to root activity, with the result that carbonate salts can remain in solution.

Additionally, the amount of deep drainage (or leaching) has an important effect in changing the concentration of salts in the root zone. Suarez (1981) developed a model for the SAR of the drainage water at the bottom of the root zone. This point was chosen because it would theoretically reflect the highest SAR reached in the soil profile.

$$SAR_d = \frac{\frac{Na_{iw}}{LF}}{\left(\frac{Mg_{iw}}{LF} + Ca_d\right)^{0.5}} \qquad (9.21)$$

where

SAR_d = SAR of drainage water at the bottom of the root zone

LF = leaching fraction at the bottom of the root zone

Na_{iw} = Na concentration in the irrigation water (in $mmole_c$ /L)

Mg_{iw} = Mg concentration in the irrigation water (in $mmole_c$ /L)

Ca_d = Ca concentration in the drainage water (in $mmole_c$/L)

Ca_d is predicted from the ionic strength, HCO_3:Ca ratio, and partial pressure of CO_2. Ca_d values can be calculated from data given by Suarez (1981).

An alternative approximate prediction of the effect of sodic irrigation water on the SAR in the root zone is provided by Miyamoto (1980):

$$SAR_d = SAR_{iw}\left(\frac{1}{LF}\right)^{0.5} \qquad (9.22)$$

where:

SAR_d = SAR of the deep drainage water at the bottom of the root zone

SAR_{iw} = SAR of the irrigation water

To estimate the ESP at the bottom of the root zone following a change in irrigation water SAR, the LF is predicted for the existing soil as per the methods in table 9.2.7. Once a LF is

determined, equation 9.22 can be used in conjunction with equation 9.19 to estimate a new soil ESP and this new ESP can then be used in the methods outlined in table 9.2.7.

Assessing soil structural stability using SAR and EC of irrigation water

In most cases, rainfall can leach accumulated salts below the root zone, thus salt accumulation from irrigation can usually be managed. However, more serious consequences result from using waters with high SAR. High sodium levels affect soil behaviour by increasing soil dispersibility, reducing water entry, making cultivation and good seed beds more difficult to attain, and reducing soil profile water availability. These issues are particularly important after rainfall, where accumulated salts are washed out of the soil surface. The soils then disperse because of the higher ESP levels.

Some general relationships can be established for many soils which indicate the combination of irrigation water EC and SAR where these dispersion problems are most likely to occur (see fig 9.2.3).

Water compositions that occur to the right of the equilibrium lines are considered satisfactory for use, provided the SAR is not so high that severe dispersion of the surface soil water will occur following rainfall. For example, if an irrigation water of EC 4 dS/m and an SAR of 8 is used for irrigation, the soil will be stable. Water quality that falls to the left of the solid line is likely to induce degradation of soil structure and corrective management will be required (e.g. application of lime or gypsum). Water that falls between the lines is of marginal quality and should be treated with caution.

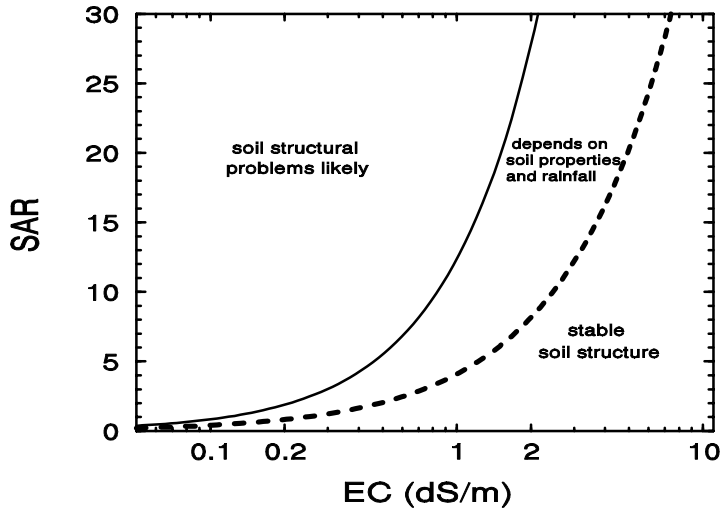

Figure 9.2.3 Relationship between SAR and EC of irrigation water for prediction of soil structural stability (adapted from DNR 1997; note that 1 dS/m = 1000 μS/cm)

Plant salt tolerance

Plant salt tolerance can be defined as the ability of plants to survive and produce economic yields under adverse conditions caused by salinity. In the case of ornamental species, the ability to survive and maintain aesthetic appearance may be more important than yield. Criteria that are commonly used to assess the suitability of a plant for a particular salinity situation are:

- salinity (the effect of the salt concentration on the plant, largely osmotic in nature);

- specific ion toxicity (the toxic effects on plants of high concentrations of specific ions, particularly sodium, chloride and other metal ions);
- nutritional disorders (due to excessive concentrations of some ions).

Plants respond to salinity in the root zone. Two measures of root zone salinity are commonly used: average and water uptake weighted. Because plants respond to the integration of atmospheric and soil conditions, average root zone salinity provides a conservative measure of soil salinity conditions for estimating plant response. Several studies (Devitt et al. 1984, Rhoades 1982, Bernstein & Francois 1973) have shown average root zone salinity to provide an appropriate estimate of root zone salinity for determining plant response to salinity.

Many Australian soils have increasing salinity and reduced soil porosity, hydraulic conductivity and water storage capacity at depth. Thus a measure of root zone salinity weighted for actual water uptake pattern of plants in the root zone could provide a more realistic estimate of plant response, since water uptake by roots is not uniform throughout the root zone. The shape of the water uptake pattern with depth varies considerably with frequency of rainfall and/or irrigation.

Shockley (1955) found that 40% of soil water extraction by plants occurred within the top quarter of the root zone depth, 30% in the second quarter depth, 20% in the third quarter depth and 10% in the fourth quarter. This relationship has been widely used; however, under conditions of frequent irrigation it was found that higher proportions of soil water extraction occurred in the top 25% of the root zone (Shaw & Yule 1978).

Water uptake weighted root zone salinity, while providing a better representation of root zone salinity where subsoils are saline, may not be sufficiently conservative to account for plant response during dry periods where subsoil water is critical for plant survival. Average root zone salinity is probably a better estimate under these conditions. For areas with shallow watertables where salt accumulation occurs in the surface layers, water uptake weighted salinity is probably a better estimate.

Plant response

Most agriculturally important crops respond to total salinity as an osmotic effect. Some woody horticultural species are also susceptible to concentrations of specific ions. When these concentrations reach toxic levels, effects are noticeable in the leaves, particularly the leaf margins. Symptoms include necrotic spots, leaf bronzing and in highly toxic cases, defoliation. This is discussed further in Maas (1986). The ions most often associated with this are sodium, chloride and boron (see Sections 9.2.4.3, 9.2.4.2 and 9.2.5.6).

The figures given for plant salt tolerance in relation to EC_{se} will depend on the intended use of the plants. Maas and Hoffman (1977) reviewed worldwide literature published on plant salt tolerance and normalised the data into a uniform framework to allow data to be evaluated and used consistently. They concluded that the normal response of plants to salinity is to have no yield reduction up to threshold level, beyond which there is an approximate linear decrease in yield with increasing soil salinity. Groupings were made based on the response of relative yield of a wide range of species to salinity into five salt tolerance categories (see fig 9.2.4).

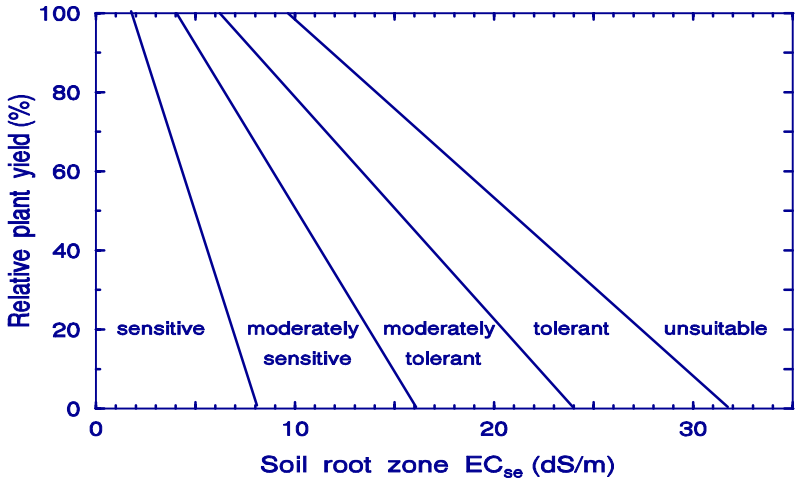

Figure 9.2.4 Relative crop yield in relation to soil salinity (EC_{se}) for plant salt tolerance groupings of Maas and Hoffman (1977). Note that 1 dS/m = 1000 µS/cm.

By incorporating irrigation water salinity (EC_{iw}) with the soil properties calculated previously (LF and EC_{se}), an approximation of the suitability of water quality to a particular irrigation situation can be assessed. This interrelationship is shown in figure 9.2.5. This figure (modified from Rhoades 1982) illustrates that leaching fraction and thus root zone salinity, has a profound influence on what plants can be grown.

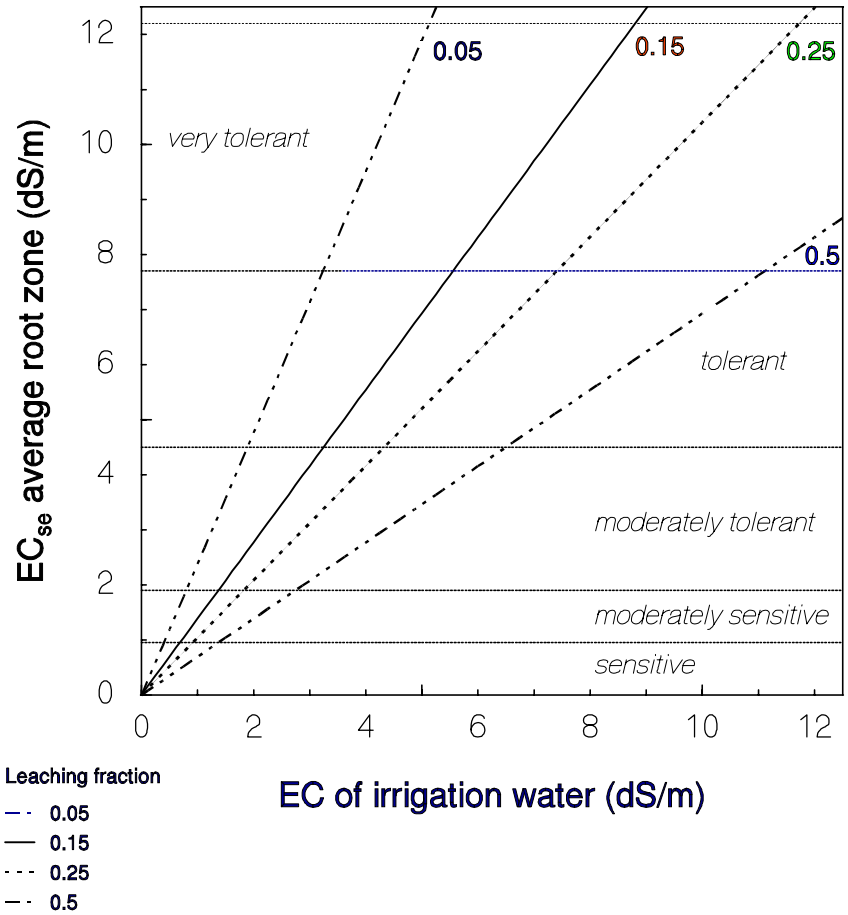

Figure 9.2.5 Interrelationships between irrigation water salinity, root zone salinity, leaching fraction and plant salt tolerance (modified from Rhoades 1982; note that 1 dS/m = 1000 µS/cm)

Table 9.2.10 is a compilation of plant salt tolerance data, including threshold salinity values and rate of yield decline with increasing salinity. The data are correct for uniformly salinised soils in which the dominant anion is chloride. This table provides a guideline of plant suitability based on average root zone salinity (EC_{se}), which was calculated previously and takes into consideration a range of factors including irrigation water quality. Information in this table is derived from data currently available in the literature, but preference should be given to locally derived data where available.

To determine actual yield response from the table, the following relationship is used:

$$Y_r = 100 - B\,(EC_{se} - A) \qquad (9.23)$$

where:

Y_r = relative yield

B = the percentage productivity decrease per dS/m increase above the threshold value (from table 9.2.10) and

A = the salinity threshold.

EC_{se} values are also provided in the table.

To calculate the EC_{se} at 90 percent yield, the equation is rearranged as:

$$EC_{se_{90\%}} = A + \frac{10}{B} \qquad (9.24)$$

and can be applied to 75 percent or 50 percent yield values as shown in table 9.2.10.

Factors affecting the expression of salinity

Historically, where major shallow groundwater systems were or are still present, or are developing, soils show considerable salt accumulation in the upper layers. In the absence of a shallow watertable (generally within the top two metres of the soil surface), salts accumulate at the bottom of the active plant root zone or at the depth of effective soil wetting. Because of the annual and longer cycles in rainfall variability, these pulses of salt accumulation can occur over a reasonable soil depth. The degree of salt accumulation in a soil depends on the degree of leaching (equivalent to the soil permeability), the presence of vegetation (evapotranspiration) and the amount of rainfall plus irrigation.

Low permeability soils tend also to be sodic in the root zone, probably derived from the presence of shallow sodic watertables in the past. Soils have limited salt accumulation in high rainfall situations (<2000 mm/year), where there is sufficient leaching to remove accumulated salts out of the soil profile.

Table 9.2.10 Plant salt tolerance data, in alphabetical order by common name, within broad plant groups[a]

Common name	Scientific name	Salinity threshold (EC_{se} dS/m)[b]	Productivity decrease per dS/m increase (%)	Soil Salinity EC_{se} (dS/m) at 90% yield	75% yield	50% yield	Reference[c]
Grains							
Barley, grain	*Hordeum vulgare*	8.0	5.0	10.0	13.0	18.0	1
Corn, grain, sweet	*Zea mays*	1.7	12.0	2.5	3.8	5.9	1
Cotton	*Gossypium hirsutum*	7.7	5.2	9.6	12.5	17.3	1
Cowpea (seed)	*Vigna unguiculata*	1.6	9.0	2.7	4.4	7.2	9
Cowpea, Caloona	*Vigna unguiculata* var *Caloona*	2.0	10.8	2.9	4.3	6.6	3
Flax/Linseed	*Vinum usitatissimum*	1.7	12.0	2.5	3.8	5.9	1
Oats	*Avena sativa*	5.0	20.0	5.5	6.3	7.5	9
Peanut	*Arachis hypogala*	3.2	29.4	3.5	4.1	4.9	2
Phasey bean, Murray	*Macroptilium lathyroides*	0.8	7.9	2.1	4.0	7.1	3
Rice, paddy	*Oryza sativa*	3.0	12.2	3.8	5.1	7.1	1
Safflower	*Carthamus tinctorius*	6.5					6
Sorghum	*Sorghum bicolor*	6.8	15.9	7.4	8.4	9.9	4
Sorghum, crooble	*Sorghum almum*	8.3	11.2	9.2	10.5	12.8	3
Soybean	*Glycine max*	5.0	20.0	5.5	6.3	7.5	1
Sugarcane	*Saccharum officinarum*	1.7	5.9	3.4	5.9	10.2	1
Sunflower	*Helianthus annuus*	5.5	25.0	5.9	6.5	7.5	9
Wheat	*Triticum aestivum*	6.0	7.1	7.4	9.5	13.0	1
Wheat, durum	*Triticum turgidum*	5.7	5.4	7.6	10.3	15.0	4
Fruits							
Almond	*Prunus dulcis*	1.5	18.0	2.1	2.9	4.3	1
Apple	*Malus sylvestris*	1.0	18.0	1.6	2.4	3.8	1
Apricot	*Prunus armeniaca*	1.6	23.0	2.0	2.7	3.8	1
Avocado	*Persea americana*	1.3	21.0	1.8	2.5	3.7	7
Blackberry	*Rubus* spp	1.5	22.2	2.0	2.6	3.8	1
Boysenberry	*Rubus ursinus*	1.5	22.2	2.0	2.6	3.8	1
Date	*Phoenix dactylifera*	4.0	3.4	6.9	11.4	18.7	1
Fig	*Ficus carica*	4.2					6
Grape	*Vitis* spp	1.5	9.5	2.6	4.1	6.8	1

a From DNR (1997); b 1 dS/m = 1000 µS/cm

c References: 1 Maas & Hoffman (1977); 2 Ayers & Westcot (1976); 3 Russell (1976); 4 Maas (1986); 5 West & Francois (1982); 6 Bresler et al. (1982); 7 Ayers (1977); 8 Heuer et al. (1986); 9 Shaw et al. (1987)

Table 9.2.10 continued

Common name	Scientific name	Salinity threshold (EC_{se} dS/m)[b]	Productivity decrease per dS/m increase (%)	Soil Salinity EC_{se} (dS/m) at 90% yield	75% yield	50% yield	Reference[c]
Grapefruit	*Citrus paradisi*	1.8	16.1	2.4	3.4	4.9	1
Guava, pineapple	*Feijoa sellowiana*	1.2					6
Lemon	*Citrus limon*	1.0					6
Natal plum	*Carissa grandiflora*	6.0					6
Olive	*Olea europaea*	4.0					6
Orange	*Citrus sinensis*	1.7	15.9	2.3	3.3	4.8	1
Peach	*Prunus persica*	3.2	18.8	3.7	4.5	5.9	1
Pear	*Pyrus* spp	1.0					6
Plum	*Prunus domestica*	1.5	18.2	2.0	2.9	4.2	1
Prune	*Prunus domestsica*	1.0					6
Pomegranate	*Punica granatum*	4.0					6
Raspberry	*Rubus ideaeus*	1.0					6
Rockmelon	*Cucumis melo*	2.2	7.3	3.6	5.6	9.0	7
Strawberry	*Fragaria*	1.0	33.3	1.3	1.8	2.5	1
Heavy vegetables							
Beet, garden	*Beta vulgaris*	4.0	9.0	5.1	6.8	9.6	1
Beet, sugar	*Beta vulgaris*	7.0	5.9	8.7	11.2	15.5	1
Onion	*Allium cepa*	1.2	16.1	1.8	2.8	4.3	1
Potato	*Solanum tuberosum*	1.7	12.0	2.5	3.8	5.9	1
Sweet potato	*Ipomoea batatas*	1.5	11.0	2.4	3.8	6.0	7
		1.5	11.1	2.4	3.8	6.0	1
Ornamentals							
Aborvitae	*Thuja orientalus*	2.0					6
Algerian ivy	*Hedera camariensis*	1.0					6
Bambatsi	*Panicum coloratum*	1.5	3.2	4.6	9.3	17.1	3
Bottlebrush	*Callistemon viminalis*	1.5					6
Bougainvillea	*Bougainvillea spectabilis*	8.5					6
Boxwood	*Buxus microphylla* var *Japonica*	1.7	10.8	2.6	4.0	6.3	1
Chinese holly	*Ilex cornuta*	1.0					6
Dracaena	*Dracaena endivisa*	4.0	9.1	5.1	6.7	9.5	1
Euonymus	*Euonymus japonica* var *Grandiflora*	7.0					6

a From DNR (1997); b 1 dS/m = 1000 µS/cm;

c References: 1 Maas & Hoffman (1977); 2 Ayers & Westcot (1976); 3 Russell (1976); 4 Maas (1986); 5 West & Francois (1982); 6 Bresler et al. (1982); 7 Ayers (1977); 8 Heuer et al. (1986); 9 Shaw et al. (1987)

Table 9.2.10 continued

Common name	Scientific name	Salinity threshold (EC_{se} dS/m)[b]	Productivity decrease per dS/m increase (%)	Soil Salinity EC_{se} (dS/m) at 90% yield	75% yield	50% yield	Reference[c]
Heavenly bamboo	*Handina domestica*	1.0					6
Hibiscus	*Hibiscus rosa-sinensis* cv. *Brilliante*	1.0					6
Juniper	*Juniperus chinensis*	1.5	9.5	2.6	4.1	6.8	1
Lantana	*Lantana camera*	1.8					1
Oleander	*Nerium oleander*	2.0	21.0	2.5	3.2	4.4	1
Pittosporum	*Pittosporum tobira*	1.0					6
Privet	*Ligustrum lucidum*	2.0	9.1	3.1	4.7	7.5	1
Pyracantha	*Pyracantha braperi*	2.0	9.1	3.1	4.7	7.5	1
Rose	*Rosa* spp	1.0					6
Star jasmine	*Trachelosperumum jasminoides*	1.6					6
Viburnum	*Viburnum* spp	1.4	13.2	2.2	3.3	5.2	1
Xylosma	*Xylosma senticosa*	1.5	13.3	2.3	3.4	5.3	1
Pastures							
Barley, forage	*Hordeum vulgare*	6.0	7.0	7.4	9.6	13.1	1
Barley, hay	*Hordeum vulgare*	6.0	7.1	7.4	9.5	13.0	2
Barrel medic, Cyprus	*Medicago truncatula*	3.0	14.6	3.7	4.7	6.4	3
Barrel medic, Jemalong	*Medicago truncatula*	1.0	7.7	2.3	4.2	7.5	3
Buffel grass, Gayndah	*Cenchrus ciliaris* var *Gayndah*	5.5	10.3	6.5	7.9	10.4	3
Buffel grass, Nunbank	*Cenchrus ciliaris* var *Nunbank*	6.0	6.8	7.5	9.7	13.4	3
Clover, alsike, ladino, red	*Trifolium* spp	1.5	12.0	2.3	3.6	5.7	1
Clover, berseem	*Trifolium alexandrinum*	2.0	10.3	3.0	4.4	6.9	3
Clover, berseem (USA)		1.5	5.8	3.2	5.8	10.1	1
Clover, rose (Kondinin)	*Trifolium hirtum*	1.0	8.9	2.1	3.8	6.6	3
Clover, strawberry (Palestine)	*Trifolium fragiferum*	1.6	10.3	2.6	4.0	6.5	3
Clover, white (New Zealand)	*Trifolium reperis*	1.0	9.6	2.0	3.6	6.2	3
Clover, white (Safari)	*Trifolium semipilosum*	1.5	12.1	2.3	3.6	5.6	3
Corn, forage	*Zea mays*	1.8	7.4	3.2	5.2	8.6	1
Couch grass	*Cynodon dactylon*	6.9	6.4	8.5	10.8	14.7	1
Cowpea (vegetative)	*Vigna unguiculata*	1.3	14.3	2.0	3.0	4.8	1
Desmodium, green leaf	*Desmodiuim intortum*	2.1	14.9	2.8	3.8	5.5	3
Desmodium, silverleaf	*Desmodium uncinatum*	1.0	22.7	1.4	2.1	3.2	3

a From DNR (1997)

b 1 dS/m = 1000 μS/cm

c References: 1 Maas & Hoffman (1977); 2 Ayers & Westcot (1976); 3 Russell (1976); 4 Maas (1986); 5 West & Francois (1982); 6 Bresler et al. (1982); 7 Ayers (1977); 8 Heuer et al. (1986); 9 Shaw et al. (1987)

Table 9.2.10 continued

Common name	Scientific name	Salinity threshold (EC_{se} dS/m)[b]	Productivity decrease per dS/m increase (%)	Soil Salinity EC_{se} (dS/m) at 90% yield	75% yield	50% yield	Reference[c]
Dodonea	*Dodonea viscosa*	1.0	7.8	2.3	4.2	7.4	1
Dolichos Rongai	*Lablab purpureus*	1.0	15.6	1.6	2.6	4.2	3
Fescue	*Festuca clatior*	3.9	5.3	5.8	8.6	13.3	1
Glycine tinaroo	*Glycine ughtii*	1.8	9.9	2.8	4.3	6.9	3
Green panic, Petri	*Panicum maximum*	3.0	6.9	4.4	6.6	10.2	3
Kikuyu grass, Whittet	*Pennisetum clandestinum*	3.0	3.0	6.3	11.3	19.7	3
Liechhardt	*Macrotyloma uniflorum*	3.0	15.6	3.6	4.6	6.2	3
Lotononis, Miles	*Lotononis bainesii*	1.0	12.2	1.8	3.1	5.1	3
Lovegrass	*Eragrostis* spp	2.0	8.5	3.2	4.9	7.9	1
Lucerne, Hunter River	*Medicago sativa*	2.0	6.0	3.7	6.2	10.3	3
Lucerne, Hunter R. (temperate)		1.5	6.9	2.9	5.1	8.7	3
Lucerne (USA)	*Medicago sativa*	2.0	7.3	3.4	5.4	8.8	1
Meadow foxtail	*Alopecurus pratensis*	1.5	9.7	2.5	4.1	6.7	1
Orchard grass	*Dactylis glomerata*	1.5	6.2	3.1	5.5	9.6	1
Pangola grass	*Digitaria decumbens (pentzii)*	2.0	4.0	4.5	8.3	14.5	3
Paspalum	*Paspalum dilatatum*	1.8	9.0	2.9	4.6	7.4	3
Phalaris	*Phalaris tuberosa (aquatica)*	4.2					6
Rhodes grass, Pioneer	*Chloris gayana*	7.0	3.2	10.1	14.8	22.6	3
Sesbania	*Sesbania exaltata*	2.3	7.0	3.7	5.9	9.4	1
Setaria, Nandi	*Setaria speculata* var *sericea*	2.4	12.2	3.2	4.5	6.5	3
Siratro	*Macroptilium atropurpureum*	2.0	7.9	3.3	5.2	8.3	3
Snail medic	*Medicago scutellata*	1.5	12.9	2.3	3.4	5.4	3
Strand medic	*Medicago littoralis*	1.5	11.6	2.4	3.7	5.8	3
Sudan grass	*Sorghum sudanense*	2.8	4.3	5.1	8.6	14.4	1
Townsville stylo	*Stylosanthes humilis*	2.4	20.4	2.9	3.6	4.9	3
Trefoil, big	*Lotus uliginosus*	3.0	11.1	3.9	5.3	7.5	1
Trefoil, birdsfoot	*Lotus corniculatus tenuifolium*	5.0	10.0	6.0	7.5	10.0	1
Urochloa	*Urochloa mosambicensis*	8.5	12.4	9.3	10.5	12.5	3
Wheatgrass, crested	*Agropyron desertorum*	3.5	4.0	6.0	9.8	16.0	1
Wheatgrass, fairway	*Agropyron cristatum*	7.5	6.9	8.9	11.1	14.7	1
Wheatgrass, tall	*Agropyron elongatum*	7.5	4.2	9.9	13.5	19.4	1

a From DNR (1997)
b 1 dS/m = 1000 μS/cm
c References: 1 Maas & Hoffman (1977); 2 Ayers & Westcot (1976); 3 Russell (1976); 4 Maas (1986); 5 West & Francois (1982); 6 Bresler et al. (1982); 7 Ayers (1977); 8 Heuer et al. (1986); 9 Shaw et al. (1987)

Table 9.2.10 continued

Common name	Scientific name	Salinity threshold (EC_{se} dS/m)[b]	Productivity decrease per dS/m increase (%)	Soil Salinity EC_{se} (dS/m) at 90% yield	75% yield	50% yield	Reference[c]
Vegetables							
Bean	*Phaseolus vulgaris*	1.0	18.9	1.5	2.3	3.6	1
Broadbean	*Vicia faba*	1.6	9.6	2.6	4.2	6.8	1
Broccoli	*Brassica oleracea*	2.8	9.1	3.9	5.5	8.3	1
Cabbage	*Brassica oleracea (*var *Capitata)*	1.8	9.7	2.8	4.4	7.0	1
Carrot	*Daucus carota*	1.0	14.1	1.7	2.8	4.5	1
Cauliflower	*Brassica oleracea*	2.5					6
Celery	*Apium graveolens*	1.8	6.2	3.4	5.8	9.9	4
Cucumber	*Cucumis sativus*	2.5	13.0	3.3	4.4	6.3	1
Eggplant	*Solanum melongena*	1.1	6.9	2.5	4.7	8.3	8
Kale	*Brassica campestris*	6.5					6
Lettuce	*Latuca sativa*	1.3	13.0	2.1	3.2	5.1	1
Pea	*Pisum sativum L.*	2.5					6
Pepper	*Capsicum annum*	1.5	14.1	2.2	3.3	5.0	9
Rosemary	*Rosmarinus lockwoodii*	4.5					6
Spinach	*Spinacia oleracea*	2.0	7.6	3.3	5.3	8.6	1
Squash	*Cucurbita maxima*	2.5					6
Squash, scallop	*Cucurbita pepo melopepo*	3.2	16.0	3.8	4.8	6.3	4
Tomato	*Lycopersicon esculentum*	2.3	18.9	2.8	3.6	4.9	1
Turnip	*Brassica rapu*	0.9	9.0	2.0	3.7	6.5	4
Zucchini	*Cucurbita peop melopepo*	4.7	9.4	5.8	7.4	10.0	4

a From DNR (1997)

b 1 dS/m = 1000 μS/cm

c References: 1 Maas & Hoffman (1977); 2 Ayers & Westcot (1976); 3 Russell (1976); 4 Maas (1986); 5 West & Francois (1982); 6 Bresler et al. (1982); 7 Ayers (1977); 8 Heuer et al. (1986); 9 Shaw et al. (1987)

Climate

The climate of a region contributes to salinity in the soil under irrigation. The two key processes of influence are evapotranspiration and rainfall.

The evapotranspiration rate is most important when the soil is wet since that is when water movement will be at a maximum (Shaw 1996). The relative rates of soil water movement downwards through the soil matrix and of evapotranspiration are the important criteria. The rate of evapotranspiration is driven by net radiation (the energy source) and is modified by the rate of removal of water from the evaporating surface (the demand), determined by vapour pressure deficit.

Seasonal patterns of rainfall have been shown to make a difference to natural soil salt levels (Shaw 1996). Yaalon (1983) showed that winter rainfall regions, with the same annual rainfall as a summer rainfall region, have greater soil leaching and recharge to groundwater, than an equivalent summer dominant rainfall. Shaw et al. (1987) examined the relative distribution of the incidence of dryland salting in Australia in relation to climate and rainfall pattern. There was a consistent relationship between the degree of winter rainfall and the area affected by dryland salinity which reflects a greater opportunity for recharge to the groundwater.

Climate influences the energy balance of a given region, which determines the difference between summer and winter rainfall hydrology, and additional parameters need to be included if the two rainfall environments are used together. Milly (1994) hypothesised that the average annual water balance was controlled by rainfall as input, by potential evapotranspiration as demand, and by soil water storage as the buffer in the system. He summarised the three rainfall/evapotranspiration regimes in terms of energy and rainfall, as:

- energy limiting (rainfall > evapotranspiration)
- rainfall limiting (evapotranspiration = rainfall)
- rainfall nonlimiting (evapotranspiration < rainfall)

Thus, a summer rainfall environment would be expected to give a similar response at high rainfall, when rainfall is not limiting, to a winter rainfall environment where energy is limiting. Different situations will occur at intermediate annual rainfalls.

Landscape

Geological features and past patterns of weathering make some landforms more hydrologically sensitive and susceptible to salting than others. The important feature of sensitive landforms is the presence of some restriction to groundwater flow that causes the watertable to rise to near the soil surface, resulting in a discharge area with evaporative concentration of salts. Hydrologically sensitive landscapes often show evidence of past seepages or shallow watertables. If development in these types of landscapes changes the hydrological balance, salting may occur as a result.

Watertable salting commonly occurs upslope of landscape features that restrict or inhibit groundwater movement or that provide preferential flow paths to the ground surface. For instance:

- Geological features, such as faults or dykes, create barriers to water flow so that water pools upslope of these barriers.

- Heavy soils at the base of slopes or clays deposited at the confluence of streams slow the movement of water through the soil or sediments, so that the groundwater pools at this point and the watertable rises.
- When water flowing through relatively permeable rock types or sediments encounters less permeable underlying materials, the water flows along the line of the strata.
- Where rock bars or other barriers constrict the outlet of a catchment, the rate of groundwater flow is reduced and water pools upslope of this point. Human-constructed barriers to water flow, such as roads or dams, have a similar effect.

During the period of landscape and soil formation, salinity processes caused salt to accumulate in areas where drainage was poor or where watertables were close to the soil surface. As more recent climates have been drier than past climates and watertables deeper, these historic salt loads are now generally at some depth in undisturbed landscapes.

When the hydrologic balance of a landscape is changed through natural processes or human activities so that a new and wetter hydrologic equilibrium is established, rising watertables can move salt from these historic salt loads closer to the soil surface. In areas sensitive to hydrologic change, watertable salting can occur when human activities disturb the hydrologic balance by increasing water inputs to the catchment or by introducing barriers to water movement within the catchment.

There is a marked association between land clearing and outbreaks of watertable salting in hydrologically restricted catchments, although there can be long time intervals (20 to 50 years or more) between clearing and salting. This delay depends on the degree of hydrologic change (due to clearing, irrigation, climatic variation) and the storage and outflow capacities of the catchment. Finely balanced catchment systems with low storage and subsurface outflow capacities will experience salting in perhaps a few years compared with a number of years in systems with greater capacities. When native vegetation is cleared and an area is developed for agriculture, grazing pressure and cropping practices can reduce the vegetative cover at times such that the vegetation cannot adequately use the available water provided by rainfall. Also, most crop and pasture species are more shallow-rooted than native species. During these periods, extra water moves below the root zone to the groundwater, increasing the likelihood of watertable rise.

Managing salinity at the catchment scale

Management decisions are rarely straightforward due to the range of factors and complexity of interactions that contribute to salinity and determine management priorities:

- The expression of salinity in landscapes results from complex interactions between land use and management, landscape hydrology, geomorphology, historic salt loads, and socio-economic and environmental factors.
- Because of the slow hydrologic response in many landscapes, there is often a long lead time between the expense and effort of implementing a management strategy and the subsequent enjoyment of the results.
- In some situations, the cost of implementing management strategies or controls can be greater than the value of on-site benefits or cost of off-site effects (although there is difficulty in assessing the full ‘cost’ of off-site effects).

- Property boundaries rarely encompass whole catchments, and additional problems can occur when areas where the salinity problem is 'caused' and 'expressed' are controlled by different landholders.

The first step in developing an integrated, sustainable management strategy is to thoroughly investigate the processes and local factors contributing to salinity. Causal factors which have not been investigated and identified cannot be addressed comprehensively and effectively. Four potential approaches to management of salinity through aiming to achieve a hydrologic balance between recharge and discharge areas are:

- manage the existing situation;
- reduce recharge;
- intercept water in the transmission area;
- increase water use in the discharge area.

Each of these approaches is listed in table 9.2.11, together with features of situations most suited to each management approach and desirable management practices. This table is intended only to provide an indication of the most viable management options for a situation at hand when management is initially being considered.

Table 9.2.11 Suitable situations and desirable management practices for each of the major salinity management approaches. Desirable management practices for implementing each strategy are listed approximately in order of likely effect.[a]

Management approach	Situations most suitable for the management approach	Desirable management practices
Manage existing situation	• Landform features: basalt, catena, alluvial valley, stratigraphic, dykes, confluence of streams • Affected land not of high value or productivity • Controlling recharge areas too costly, or recharge areas much more productive than affected discharge areas • Vegetation currently surviving on most of the affected area • Existing vegetation can be enhanced and/or fenced to control grazing • Seepage on the affected area is fair quality water • Erosion not a problem, or erosion can be stabilised with vegetation • Downstream water quality not significantly affected by salting in the affected area • Salt load in the discharge area is moderately high • Watertable intercepts the soil surface seasonally or periodically	• Set a high priority on maintaining vegetative cover • Fence off affected areas and manage grazing pressures • Enhance amount of salt-tolerant vegetation in the worst affected areas • Stabilise area against erosion, but do not prevent seasonal flooding where this would normally occur • Improve surface drainage • Plant trees or other perennial deep-rooted vegetation that can handle salt and waterlogging
Reduce recharge	• The catena landform feature • Recharge area clearly identifiable and available for treatment • Area experiences a winter rainfall pattern • Shallow-rooted pastures are main vegetative cover in the recharge area	• Avoid summer fallow in summer rainfall areas, and use double or opportunity cropping if possible • Introduce deeper rooted or perennial species into the pasture mix • Incorporate agroforestry into management

Management approach	Situations most suitable for the management approach	Desirable management practices
	• Current cropping practices could be made more water-use efficient • Rainfall periods not aligned with periods of high water use by crops • Recharge rates high • Land value or productive value of the discharge area greater than that of the recharge areas • Soil in the discharge area likely to be productive after the area is reclaimed — that is, groundwater in the discharge area not particularly sodic and soil structure not severely affected	• Revegetate stock routes, along fence lines and geomorphic boundaries • If leakage from ponded areas is significant, reduce size of these areas
Intercept water in the transmission area	• Landform features: basalt, catena, colluvia of former land surfaces, valley restrictions, dykes, confluence of streams • Transmission area relatively well defined • Recharge area large and not well defined	• Depending on water quality and depth to groundwater • Pump with pumps or windmills from single or linked tubewells. (A total minimum flow of around 2 to 3 L/s is needed for this option to be viable.)
	• Groundwater is of acceptable quality • Good aquifers identifiable in the transmission area • Aquifers suitable for pumping or accessible by tree roots • Pumped water can be discharged into streams, evaporated, or used for irrigation • Discharge area is under upward hydraulic pressure resulting from a confining clay layer and thus much more difficult to manage • Both recharge and discharge areas have high land values • Large quantities of water involved • Major salt loads occur in the discharge area	• If water is good quality, intercept groundwater and use to irrigate adjacent areas or to water stock • Plant dense vegetation belts, using high water use species, in areas where these plants can access the groundwater • Construct subsurface drainage (for off-site disposal) if water is of acceptable quality
Increase water use in discharge area	• Landform features: colluvia of former land surfaces, valley restriction, dykes, geologic faulting • Recharge area diffuse and extensive • Recharge areas distant from the discharge area, or not under the control of the discharge area landholder • Discharge area extensive • High economic value of the recharge areas, regardless of the comparative value of the affected discharge areas • Transmission area diffuse • Finite salt loads exist in the discharge area • Groundwater of generally acceptable quality, or groundwater saline and using evaporative basins to evaporate the excess water is cost-effective • Waterlogging is an issue	• Revegetate the area with perennial, high water use, salt-tolerant vegetation • Plant halophytic species in high salinity areas • Pump with pumps or windmills from single or linked tubewells. (A total minimum flow of around 2 to 3 L/s is needed for this option to be viable.) • Construct subsurface and surface drainage • Pump into evaporation basins • If water is good quality, pump to irrigate adjacent areas

a Adapted from DNR (1997)

In most situations, a combination of these four approaches may be needed to formulate the best salinity management strategy for local conditions and the available resources. Decision support tools such as property management models and cost-benefit analyses will assist in developing a balance between different levels of control in each of the recharge, transmission and discharge areas. The relative size of recharge and discharge areas will determine, to some extent, which strategies may be appropriate.

Irrigation management for salinity control

More specific management options for the prevention or amelioration of salinity in irrigation areas are listed below.

High watertables

Efficient water management is required to prevent rises in watertable levels, especially in surface water irrigation systems. Watertables should be kept below 1.2 m. Various methods of high watertable prevention and control are available including:

- planning to identify restrictions to drainage in the landscape and delineate appropriate controls of watertables;
- reducing accessions to watertables by surface levelling and selection of water application systems according to soil permeability;
- appropriate lining of channels or use of pipes for on-farm distribution to minimise seepage from channels;
- incorporating drainage where it is both economically and environmentally sustainable.

Saline and sodic irrigation waters

Accurate irrigation water quality assessment is the best preventive measure to reduce salinity and sodicity effects, since water is matched to the soil properties and crops. However, a number of management alternatives are available to minimise the effects of marginal-quality irrigation waters on soils and crops. These management options include changing the frequency, duration and method of irrigation; judicious timing of leaching irrigations; mixing of irrigation water supplies; and cultural practices, including soil amendments. These are described in detail by Ayers and Westcot (1976).

9.2.3.3 Worked examples

Worked examples are given in this Section to provide a practical guide to salinity management in a number of situations using soil and water quality data. Note that EC is expressed as dS/m (1 dS/m = 1000 μS/cm).

Scenario 1

A farmer has been irrigating for 10 years from a local bore and is interested in the range of crops most suitable for his water quality and soil type.

Available data:

Soil

$EC_{1:5}$ at 0.9 m = 0.9 dS/m

Air dry moisture content = ADMC = 5%

Saturation percentage = SP = 60%

Water

Depth of irrigation = D_i = 600 mm

EC of irrigation = EC_{iw} = 1.7 dS/m

Depth of rainfall = D_r = 650 mm

EC of rainfall = EC_r = 0.03 dS/m

Convert $EC_{1:5}$ to EC_s

$$EC_s = 2.2 \text{ x } EC_{1:5} \text{ x } \left[\frac{(500 + 6 \text{ x ADMC})}{SP}\right] \qquad \text{(derived from Eqns 9.14 and 9.17)}$$

EC_s = 2.2 x 0.9 x 8.83 = 17.5 dS/m

Calculate EC_i (weighted EC of input water)

$$EC_i = \frac{\left[(D_{iw} \text{ x } EC_{iw}) + (D_r \text{ x } EC_r)\right]}{D_i} \qquad \text{(from Eqn 9.3)}$$

$$EC_i = \frac{\left[(600 \text{ x } 1.7) + (650 \text{ x } 0.03)\right]}{1250} = 0.83 \text{ dS/m}$$

Calculate leaching fraction (bottom of root zone)

$$LF = \frac{EC_i}{EC_s} = 0.05 = 5\% \qquad \text{(from Eqn 9.8)}$$

Calculate leaching fraction (average root zone)

$$LF_{av} = (0.976 \times LF + 0.022)^{0.625} = 0.19 = 19\% \qquad \text{(from Eqn 9.12)}$$

Predicted root zone salinity

$$EC_{se} = \frac{EC_i}{(2.2 \times LF_{av})} = 2.0 \text{ dS/m} \qquad \text{(from Eqn 9.11)}$$

The predicted EC_{se} can then be compared with plant salt tolerance data provided in table 9.2.10 to determine likely crop response to this irrigation regime.

Scenario 2

A farmer has installed a new bore as an alternative source for irrigation water supply and is interested in possible limitations to its use.

Available data:

Soil

Average clay % to 0.9 m = 55%

Average CEC to 0.9 m = 45 $mmole_c$/100 g

ESP at 0.9 m = 4.5%

Water

SAR = 5

EC_i = 1.7 dS/m

Depth irrigation = D_i = 700 mm

Depth rainfall = D_r = 550 mm

Total depth = $D_t = D_i + D_r$

Calculate leaching fraction under rainfall using:

$$LF_r = \frac{EC_r}{2.2 \times 10^{\left[a + b\log\left(\frac{0.03 \times \text{rainfall}}{ESP}\right)\right]}} \qquad \text{(from Eqn 9.5)}$$

where:

a = 0.794

b = –1.105

LF = 0.009 = 1%

The coefficients a and b are obtained from table 9.2.8.

Calculate leaching fraction under new irrigation water quality:

$$LF_f = LF_i\left[2.65\left(\frac{EC_i}{EC_r}\right)^{0.5} - 1.35\right] \qquad \text{(from Eqn 9.7)}$$

This relationship accounts for an increase in LF due to increased ionic concentration of the soil solution.

LF_f = 0.121 = 12.1%

Calculate leaching fraction (average root zone):

$LF_{av} = (0.976 \text{ x } LF + 0.022)^{0.625} = 0.29 = 29\%$ (from Eqn 9.12)

Predicted root zone salinity:

$$EC_{se} = \frac{EC_i}{(2.2 \times LF_{av})} = 1.51 \text{ dS/m} \qquad \text{(from Eqn 9.11)}$$

The predicted EC_{se} can then be compared with plant salt tolerance data provided in table 9.2.10 to determine likely crop response to this irrigation regime.

9.2.3.4 Alternative approaches to deriving guideline values

In the past, to overcome the complexity of the many interactive factors that determine leaching, guidelines were developed based on water composition alone, under assumed

'average' conditions of use. The result has been guidelines that are too conservative, particularly for 'above average' conditions of use.

Historically, water quality guidelines for irrigation have been developed for specific regions where local soils, environmental conditions and management practices have been influential in framing the suitability limits. Since the influence of local conditions is often poorly defined, generally conservative water quality guidelines have been developed which cannot be satisfactorily extrapolated to different regions.

Inherent in the philosophy of previous guidelines, is the control of soil leaching through the quantity of water applied. This is satisfactory for permeable soils; however, for slowly permeable soils (with 'steady-state' infiltration rates), leaching is predominantly controlled by soil properties rather than by irrigation water management.

The methodology outlined in these guidelines is based on a new approach which assesses irrigation water salinity and sodicity using a combination of environmental parameters including irrigation water quality, rainfall, soil characteristics and plant salt tolerance. A large number of water quality assessment schemes have been devised throughout the world and applied to irrigation salinity management situations in Australia with varying degrees of success. Overseas approaches were reviewed by Shaw (1996); a brief review of the most influential schemes is provided below.

United States of America irrigation water quality assessment schemes

The United States of America has provided various irrigation water quality assessment schemes that form the basis of guidelines still in use today. Christiansen et al. (1977) give an overview of these schemes in Shaw (1986), summarised below.

Schofield (1936)

One of the earliest published schemes in the USA was that of Schofield (1936) who gave a good review of the factors affecting the suitability of waters for irrigation. He deduced water quality criteria based on field observation of their effects and reported these for a given soil, a given climate and a given group of crops. He considered waters as doubtful for irrigation with EC values >2 dS/m and sodium percentage >60%. This scheme was regional and later theoretical studies indicated the use of SAR as a more theoretically sound basis for sodium hazard assessment.

United States Salinity Laboratory Staff (1954)

The United States Salinity Handbook 60 (USSL 1954) has probably received the widest recognition of any water quality scheme in existence. It was a very thorough and definitive statement for its time. The criteria were based on waters and soils from the western United States with low rainfall and the major criticisms of this scheme are its very conservative guidelines. The water quality divisions were based on the frequency distribution of the waters in the regions studied and their leaching rates in the particular soils studied, rather than on a definite crop tolerance basis.

Doneen (1954; 1966)

Doneen was concerned with the solubility of the less soluble carbonates and sulfates when considering the salinity evaluation of a water, since these will precipitate out of solution first. He defined an 'effective salinity' which is total salt content minus calcium carbonate ($CaCO_3$), magnesium carbonate ($MgCO_3$) and calcium sulfate ($CaSO_4$).

It has not received very wide acceptance and Lloyd Doneen himself stated that the values could be reasonably exceeded where there is adequate leaching. In 1966 he published a slight modification where he considered potential salinity as equal to ½ SO_4 + Cl (concentrations in meq/L) which made calculation easier. He also developed an empirical permeability index from which he derived various class intervals based on relative permeabilities for the few soils tested. He concluded that at best a classification scheme can be used only as a general guide and local conditions to some extent determine its usefulness.

Adjusted SAR

In the mid-1960s to 1970s, an adjustment to the normal sodium adsorption ratio (SAR) calculation was considered, to account for the precipitation of carbonates. This was based on the work of Bower et al. (1965, 1968) and modified by Rhoades (1972). This was found to be an overestimate for many situations, as it considered $MgCO_3$ to precipitate with $CaCO_3$, which does not occur. Where it is still used by Jim Rhoades, an empirical correction factor of 0.5 is used. Miyamoto (1980) and Suarez (1981) have provided more theoretically sound and still readily useable relationships.

University of California Committee on Irrigation Water Quality Standards

In 1959, the State Department of Water Resources requested the University of California to provide a classification system for water quality suitable for planning purposes in the California Water Plan. Although no scheme was forthcoming, a conference was held in 1963 to discuss outcomes of the study and the general consensus was reasonably close to the USSL (1954) but not as conservative.

In 1972–73, the State of California commissioned a study on water resources including management strategies to maintain groundwater quality. They engaged private consultants and State agencies who worked in conjunction with a 'Committee of Consultants' set up by the University of California and the US Salinity Laboratory. As a result of this study, and the delay in publishing a revision of Handbook 60, Ayers and Westcot (1976) was published with the involvement of Doneen and the Food and Agriculture Organisation (FAO) of the United Nations.

Ayers and Westcot (1976)

Ayers and Westcot (1976) designed practical water quality guidelines that could be used in the field in developing countries and contains a detailed Section on management of water quality problems. There is a very strong similarity in the EC classes with all the earlier USA schemes from Schofield onwards and a heavy reliance on adjusted SAR and clay mineral response which has since been proven to be invalid. The limits for EC are conservative and the philosophy of the guidelines is that severe problems will result if EC is >3 dS/m, unless recommended management procedures are undertaken.

Other overseas schemes

Bernstein (1967)

Bernstein (1967) attempted to derive a more quantitative water quality scheme by incorporating a soil leaching term based on evapotranspiration rate and infiltration rate. For slowly permeable soils, drainage rate below the root zone and evapotranspiration rate are used. For most assessment situations, this information is not available and errors in measurement, particularly for cracking clay soils, are high making the implementation of the scheme less flexible.

Williams and Gostrik (1981)

The authors outline a water quality classification for England which accounts for the seasonal irrigation requirement for crops with varying salt tolerance. Because irrigation is supplemental at very low rates (50–200 mm/year) in contrast to Australia (400–900 m/year), the basis and use is much more limited in scope. Chloride is used as the basis for salinity assessment.

Australian schemes

Queensland Department of Primary Industries

In Queensland, Brunnich (1927) provided the earliest known water quality classification. He considered waters with total salt content up to 1430 mg/L (approximate EC 2.2 dS/m) as suitable for irrigation. Sodium carbonate (residual alkali) between 4 and 8 meq/L was also acceptable. The salinity level is low and the residual alkali level very high in relation to other schemes.

In the early 1960s, von Steiglitz (1961) compiled a set of water quality criteria based on chloride content which was widely adopted by the Queensland Department of Primary Industries until 1984. This was replaced by Gill (1984) who outlined an amended scheme more soundly based on plant salt tolerance groupings for 90% optimum yield [as outlined by Shaw & Hughes (1981) and Shaw and Dowling (1985)], because chloride use was found to be a problem in high bicarbonate and sulfate waters where it underestimated salinity hazard.

VIRASC (1980) and Hart (1974)

VIRASC (1969, 1980) give water quality guidelines based largely on a system very similar to USSL (1954) but with earlier sodicity data from Wilcox (1958). Guidelines given by Hart (1974) are based on VIRASC (1969) and USSL (1954) but have included the more recent work of Ayers and Westcot (1976) as an Appendix.

Rhoades (1983)

Rhoades (1983) developed a water quality suitability model based on an earlier approach of Rhoades and Merrill (1976). The basis of the model is:

- prediction of salinity, sodicity and concentration of toxic solutes in the soil water within a simulated crop root zone under irrigation with a specified water composition and a specified leaching fraction;
- evaluation of the effects of predicted salinity on crop yield and the effects of predicted surface soil sodicity on soil permeability.

The method uses simplified calculations to derive equilibrium soil salinity and sodicity levels and is suitable when soil leaching fraction is known and can be varied with irrigation water management. No account is taken of changes in soil leaching with increased electrolyte or sodicity under irrigation, which is particularly important for clay soils.

Cass and Sumner (1982 a,b,c)

Cass and Sumner developed a model from the earlier work of Cass (1980) who incorporated soil and climatic factors in a water quality assessment method based on the model of Bernstein (1967) for slowly permeable soils. Cass (1980) used a measured or estimated soil drainage rate to calculate soil salinity and sodicity. The difficulty in making a realistic estimate of the true soil drainage rate, and thus the magnitude of the drainage term in relation to evapotranspiration, prevented any quantitative prediction of soil salinity.

Cass and Sumner (1982a) developed an empirical 'sodium stability model' to evaluate soil hydraulic conductivity reduction and aggregate stability with varying electrolyte and sodicity levels. Their approach normalises the slope of the traditional hydraulic conductivity SAR and electrolyte concentration relationships and from this derives a stability index based on the properties of the irrigation water and the predicted soil solution composition from the model of Oster and Rhoades (1975). Crop yield is determined from the predicted soil solution composition related to the data of Maas and Hoffman (1977) through a yield index.

9.2.4 Major ions of concern for irrigation water quality

9.2.4.1 Bicarbonate

No trigger value is recommended for bicarbonate in irrigation waters.

Description

The bicarbonate (HCO_3^-) ion is one of the major contributors to alkalinity in irrigation waters and soil. It is formed through the reaction of carbon dioxide with various components in the water source (or, in the case of groundwater, the soil or geological strata through which it percolates).

An example of the chemical reactions involved in bicarbonate formation is given below:

$$CO_2 + CaCO_3 + H_2O \leftrightarrow Ca^{2+} + 2\ HCO_3^- \quad (9.25)$$

Effects on agriculture

Elevated levels of bicarbonate in irrigation waters can adversely affect irrigation equipment, soil structure and crop foliage. In arid and semi-arid regions of Australia, irrigation water containing elevated concentrations of bicarbonate is frequently used. Prolonged use of such irrigation water can lead to a high concentration of bicarbonate in the soil water due to evapotranspiration, and there is an increasing tendency for calcium and magnesium to precipitate as insoluble carbonates. Over time, this reduction of calcium and magnesium concentration can result in an increased sodium adsorption ratio (SAR), which may impact adversely on soil structure (discussed in Section 9.2.3).

Crops such as ornamentals, fruit and flowers which are marketed on the basis of aesthetic value, can be affected by white scale formation on visible surfaces. This occurs as a result of spray irrigation, when a white precipitate of carbonates is desposited following evaporation of residual water droplets on the plant. The process continues to occur with further build-up of material due to the low solubility of these carbonate compounds, which do not redissolve when wetted but tend to accumulate.

White scale accumulation can occur with relatively low concentrations of bicarbonate and appears to be more prevalent in periods of low humidity and high evaporation (Gill 1986). High pH, which can occur when excessive amounts of bicarbonate are present in irrigation waters, can also be detrimental to plant growth by limiting uptake of certain ions.

9.2.4.2 Chloride

There are two distinct issues concerning chloride concentrations in irrigation waters: relating to the risk of (1) foliar injury to crops; and (2) increased uptake by plants of cadmium from soil.

1 Foliar injury

Trigger values for prevention of foliar injury due to chloride in irrigation water from sprinkler application are provided in table 9.2.12.

The chlorides of sodium, potassium, calcium and magnesium are highly soluble in water. Chloride behaves similarly to sodium with similar foliar symptoms. Yield declines previously attributed to chloride levels in waters have more recently been found to be closely related to sodium levels or electrical conductivity. High levels of chloride in the soil solution will lead to yield decline due to an osmotic effect, hence threshold values for salinity (expressed as EC) should be used as a guide to water quality (Section 9.2.3).

Chloride in irrigation water can also reduce the quality of tobacco leaf. Chloride concentrations >40 mg/L are considered unsuitable for irrigation of this crop and some reduction in quality may occur with waters containing chloride concentrations in the range 25–40 mg/L (Gill 1986).

Table 9.2.12 Chloride concentrations in irrigation water (mg/L) causing foliar injury in crops of varying sensitivity[a]

Sensitive <175	Moderately sensitive 175–350	Moderately tolerant 350–700	Tolerant >700
Almond	Pepper	Barley	Cauliflower
Apricot	Potato	Maize	Cotton
Citrus	Tomato	Cucumber	Sugar beet
Plum		Lucerne	Sunflower
Grape		Safflower	
		Sesame	
		Sorghum	

a After Maas (1990)

2 Interaction between chloride in irrigation water and cadmium in soil

Trigger values for assessing chloride levels in irrigation water with respect to increased cadmium uptake by crops are provided in table 9.2.13.

Table 9.2.13 Risks of increasing cadmium concentrations in crops due to chloride in irrigation waters[a]

Irrigation water chloride concentration (mg/L)	Risk of increasing crop cadmium concentrations
0–350	Low
350–750	Medium
>750	High

a McLaughlin et al. (1999)

Chloride (Cl) forms a series of complexes with cadmium (Cd) depending on solution chloride concentration (Hahne & Kroontje 1973):

$$Cd^{2+} + Cl^- \rightarrow CdCl^+ \quad \log K = 2.0 \tag{9.26}$$

$$CdCl^+ + Cl^- \rightarrow CdCl_2^{\,0} \quad \log K = 2.6 \tag{9.27}$$

$$CdCl_2^{\,0} + Cl^- \rightarrow CdCl_3^{\,-} \quad \log K = 2.4 \tag{9.28}$$

$$CdCl_3^{\,-} + Cl^- \rightarrow CdCl_4^{\,2-} \quad \log K = 2.5 \tag{9.29}$$

Thus, as solution chloride concentrations increase above approximately 400 mg/L, $CdCl^+$ will be more abundant in solution than Cd^{2+}. Such chloride concentrations are common in irrigation waters, and soil solutions may contain much higher chloride concentrations due to evapotranspiration, so that $CdCl_n^{2-n}$ complexes dominate cadmium solution chemistry in saline irrigated soils in Australia (McLaughlin et al. 1997). Due to the increased mobility of cadmium in the soil-plant system conferred by chloride, particularly at the root surface (Smolders & McLaughlin 1996), cadmium concentrations in crops are increased.

From Australian data, it has recently been clearly demonstrated that for commercial crops, addition of chloride in irrigation waters significantly increases crop Cd concentrations (McLaughlin et al. 1994). If high chloride concentrations are present in the irrigation water, it is recommended that produce irrigated with the water is tested for cadmium concentration in the edible portions (e.g. potato tubers, leafy vegetables, cereal grains, etc).

9.2.4.3 Sodium

Trigger values for prevention of foliar injury due to sodium in irrigation water following sprinkler application are provided in table 9.2.14. Values for specific toxicity effects are provided in table 9.2.15.

Table 9.2.14 Sodium concentration (mg/L) causing foliar injury in crops of varying sensitivity[a]

Sensitive <115	Moderately sensitive 115–230	Moderately tolerant 230–460	Tolerant >460
Almond	Pepper	Barley	Cauliflower
Apricot	Potato	Maize	Cotton
Citrus	Tomato	Cucumber	Sugar beet
Plum		Lucerne	Sunflower
Grape		Safflower	
		Sesame	
		Sorghum	

a After Maas (1990)

Table 9.2.15 Effect of sodium expressed as sodium adsorption ratio (SAR) on crop yield and quality under non-saline conditions[a]

Tolerance to SAR and range at which affected	Crop	Growth response under field conditions
Extremely sensitive SAR = 2–8	Avocado Deciduous Fruits Nuts Citrus	Leaf tip burn, leaf scorch
Sensitive SAR = 8–18	Beans	Stunted growth
Medium SAR = 18–46	Clover Oats Tall fescue Rice Dallis grass	Stunted growth, possible sodium toxicity, possible calcium or magnesium deficiency
High SAR = 46–102	Wheat Cotton Lucerne Barley Beets Rhodes grass	Stunted growth

a After Pearson (1960); SAR Sodium Adsorption Ratio (see Section 9.2.3)

In minute quantities, sodium is beneficial to the growth of some plants. At higher concentrations it is toxic to many plants. High levels of sodium can cause three effects on plant growth: (1) excess sodium accumulates in leaves, causing leaf burn and possibly defoliation; (2) development of poor soil physical conditions which limit growth (see Section 9.2.3); and (3) calcium and magnesium deficiency through reduced availability and imbalance with respect to sodium.

9.2.5 Heavy metals and metalloids

9.2.5.1 Scope

Revision of the irrigation water quality guidelines assessed the following criteria for each heavy metal and metalloid:

- existing Australian, New Zealand and international soil quality criteria and metal/metalloid guidelines;
- plant phytotoxicity;
- minimisation of toxic metal uptake into food crops (food quality);
- impact on farm infrastructure (e.g. bio-clogging of irrigation lines due to iron or manganese);
- off site impacts;
- impact on soil biota (ecotoxicity).

Although the potential toxicity of metals and metalloids to the soil biota (micro and macro flora and fauna) is an issue receiving international attention, and ecotoxicity is generally observed at lower soil concentrations than phytotoxicity (Will & Suter 1994b), research in this area is in its infancy (Brookes 1995). While the guidelines have considered the potential environmental impacts of inorganic contaminants in irrigation water on soil biota, insufficient information is available at present to be able to set water quality guideline values based on ecotoxicity to soil biota.

The metal guideline values for irrigation water use address the specific targets and environmental quality criteria listed above, and the potential for the transport of contaminants off-site. When compared to other Sections of the Water Quality Guidelines, guideline values are different. For instance, the trigger value for cadmium in aquatic systems ranges from 0.013 to 0.13 μg/L, whereas for irrigation water it ranges from 0.01–0.05 mg/L. This difference is partly due to the sensitivity of the target organisms, that is, native aquatic species in aquatic systems versus plants growing in soil in agricultural systems. However, the main difference between the irrigation water quality approach and the aquatic systems approach, is the attenuation of the potential adverse effects of metals when irrigation water is added to soils. The irrigation water guidelines work from a conservative and protective soil metal concentration (in line with existing soil metal guidelines), back to irrigation water concentrations. This approach is therefore more consistent than current guidelines.

9.2.5.2 Methodology for development of guideline values

Sources of irrigation water

The guidelines for water quality with regard to inorganic contaminants have been developed with a range of different irrigation sources in mind. These include groundwater, rivers, farm

dams, treated secondary sewage effluent and treated industrial effluents. Water quality from these different sources will be highly variable. It has also been assumed that the concentrations of many of the minor elements such as lithium, selenium, uranium, vanadium, etc, will be negligible even in industrial effluents. However, some irrigation waters may have high concentrations of these elements resulting for instance, from natural geochemical enrichment.

Irrigation water use

These guidelines assume that irrigation water is applied to soils and that soils may reduce contaminant bioavailability by binding contaminants and reducing the solution phase concentration. The values in these guidelines may not be suitable for plants grown in soil-less media (hydroponics or similar methods).

Toxicity of contaminants in irrigation waters to crops

There are two main ways in which the presence of inorganic contaminants in irrigation waters may have a negative impact on crops:

- contaminants may be directly phytotoxic to crops during periods of irrigation; and
- prolonged irrigation will lead to the build-up of inorganic contaminants in the soil surface layer and there is the potential for contaminants to reach concentrations in soil that are toxic to crops or cause a reduction in crop quality, through plant root uptake.

Calculation of irrigation loading rates and time periods

In order to develop these guidelines the following set of assumptions were used to calculate the contaminant loading rates resulting from irrigation:

- annual application of irrigation water is 1000 mm;
- inorganic contaminants are retained in the top 150 mm of the soil profile;
- irrigation will continue on an annual basis for a maximum of 100 years;
- soil bulk density is 1300 kg/m^3.

This set of assumptions is internationally recognised as a basis for developing irrigation water quality guidelines and has been used in the development of Canadian (CCREM 1987), UN Food and Agriculture Organisation (Pescod 1992), United States (USEPA 1992) and South African (DWAF 1996a) irrigation water quality guidelines.

Theoretical basis to guideline value development

Many factors can modify contaminant behaviour and toxicity in the soil environment, such as soil texture, soil and irrigation water pH, soil and irrigation water salinity, soil organic matter content. Thus, fine textured soils (i.e. clay soils) can withstand much higher loadings of contaminants before toxicity symptoms are evident in plants or biota. Similarly, for the same loading of cationic metal (e.g. cadmium, zinc), acidic soils have greater potential for toxicity to be manifest than alkaline soils. Thus a single trigger value must be treated with caution, as effects of contaminants on plants and organisms are therefore soil (condition) specific.

Potential for contamination of groundwater is also an issue that is highly soil (condition) specific. If areas subject to high levels of leaching, or soils with known by-pass or preferential flow are receiving significant irrigation inputs, a site-specific risk assessment is strongly suggested. This should include determining the partition coefficient (K_d) for metals on that particular soil type, leaching fraction and volume of preferential/by-pass flow. The

trigger values for metal contaminants in irrigation water may then be revised if groundwater contamination is considered a potential risk.

The trigger values suggested here have been developed with regard to soil threshold values (where available) in the literature that aim to prevent potential adverse effects of inorganic contaminants on plants and organisms, coupled with the assumptions regarding irrigation loads given below.

The proposed trigger values have been developed to be compatible with international guidelines for irrigation water quality; Australian, New Zealand and international guidelines for maximum contaminant concentrations in soils (McLaughlin et al. 2000); and recent draft New South Wales EPA soil phytotoxicity investigation levels (NSWEPA 1998). Two trigger values have been produced for irrigation water quality, and a separate limit has been proposed for a maximum soil contaminant loading, where existing soil threshold values are available.

Guideline values for irrigation water quality are defined as:

- **Long-term trigger value (LTV)**. The LTV is the maximum concentration (mg/L) of contaminant in the irrigation water which can be tolerated assuming 100 years of irrigation, based on the irrigation loading assumptions previously mentioned.
- **Short-term trigger value (STV)**. The STV is the maximum concentration (mg/L) of contaminant in the irrigation water which can be tolerated for a shorter period of time (20 years) assuming the same maximum annual irrigation loading to soil as LTV.

The STV and LTV values have been developed to minimise the build-up of contaminants in surface soils during the period of irrigation, but also to prevent the direct toxicity of contaminants in irrigation waters to standing crops. Where STV and LTV have been set at the same value, the primary concern is the direct toxicity of irrigation water to the standing crop (e.g. for lithium and citrus crops), rather than a risk of contaminant accumulation in soil and plant uptake.

The guideline value for contaminant concentration in soil is defined as the:

- **Cumulative contaminant loading limit (CCL)**. The CCL is the maximum contaminant loading in soil defined in gravimetric units (kg/ha) and indicates the cumulative amount of contaminant added, above which site specific risk assessment is recommended if irrigation and contaminant addition is continued.

The CCL is calculated based on background concentrations of contaminants in Australian agricultural soils, mixing of the contaminant in the top 0.15 m of soil, a soil bulk density of 1300 kg/m^3, and Australian guideline values for contaminant concentrations in agricultural soils treated with sewage biosolids (NSWEPA 1995a, SAEPA 1996). Once the CCL has been reached, it is recommended that a soil sampling and analysis program be initiated on the irrigated area, and an environmental impact assessment of continued contaminant addition be prepared. As the background concentrations of contaminants in soil may vary with soil type, and contaminant behaviour is dependent on soil texture, pH, salinity, etc, it should be noted that CCLs may be overly protective in some situations and less protective in others. The CCL is designed to be used in soils with no known history of contamination from other sources. Where contamination of the soil is suspected prior to commencement of irrigation, background levels of contaminants in the soil should be determined and the CCL adjusted accordingly.

The CCL for contaminants has been calculated as follows:

$$CCL = \frac{(MAC - BC) \text{ x } Depth \text{ x } BD}{10^2} \ (kg / ha) \qquad (9.30)$$

where:

MAC = maximum allowable soil concentration of a contaminant (mg/kg)

BC = assumed background concentration (mg/kg)

Depth = soil depth (0.15 m)

BD = soil bulk density (kg/m^3)

Example calculation of CCL for zinc

In soils, zinc is often applied as a crop micronutrient, but at high concentrations can be both phytotoxic and toxic to the soil flora and fauna (see reviews by Will & Suter 1994a, b and Scott-Fordsmand & Pederson 1995). Recommended maximum zinc concentrations in soil, above which adverse effects on either plants or microorganisms are likely, vary from 100–200 mg/kg (Will & Suter 1994a,b, Scott-Fordsmand & Pederson 1995). In Australia, maximum allowable concentrations (MACs) for zinc in agricultural soils receiving sewage biosolids have been set at 200 mg/kg (NSWEPA 1995a) or 250 mg/kg (SAEPA 1996). Background concentrations of zinc in Australian soils are not well documented, but from data in Olszowy et al. (1995) the arithmetic mean zinc concentrations in a range of uncontaminated rural soils was 21 mg/kg. Tiller (1983) quoted a mean value of 34 mg/kg for zinc in 459 broadacre agricultural soils. In an unpublished survey of metal concentrations in Australian horticultural soils (CSIRO, unpublished data), the arithmetic mean zinc concentration was 48 mg/kg. Data from these studies as well as the survey of Barry (1997) are summarised in table 9.2.16. The median background zinc concentration in soil of 39 mg/kg was derived from these four studies and used with the lower of the current soil zinc MACs for biosolid disposal of 200 mg/kg (table 9.2.17), to derive the CCL. Median values for contaminants were used to derive the background concentrations used for calculating CCL in these guidelines due to the log normal distribution of many of the datasets.

Zinc CCL is calculated as follows:

$$\frac{(200 - 39) \text{ x } 0.15 \text{ x } 1300}{10^2} = 300 \text{kg/ha Zn} \qquad (9.31)$$

Using only an analysis of the irrigation water and the irrigation water application frequency and amounts, the user can calculate the CCL (assuming soils have no history of metal contamination). Once the CCL has been reached for a particular site, it does not necessarily follow that irrigation must cease. When the CCL has been exceeded, it is recommended that a site environmental impact assessment is instigated, which would include analysis of soil contaminant concentrations on the irrigated site.

From table 9.2.17, the LTV and STV for zinc are 2 mg/L and 5 mg/L, respectively. The LTV and STV would allow zinc concentrations in soil to reach over 1000 and 2500 mg/kg, respectively, using the previously stated assumptions. These soil zinc concentrations are likely to lead to severe adverse effects on both plant and soil biota in high risk soils (e.g. sandy, acidic soils). The CCL has been introduced to avoid this undesirable situation, but still allow assessment of environmental risks due to contaminants in irrigation water using only a water analysis, rather than a specialised soil sampling and analysis program.

Agricultural irrigation water LTV, STV and soil CCL guidelines for a range of metals and metalloids are summarised in table 9.2.17.

Table 9.2.16 Datasets used to derive suggested upper background values for uncontaminated Australian soils

	Olszowy et al. (1995)			Barry (1997)			CSIRO unpub			Tiller (1983)		Spouncer & Mowat (1991a–d)		
	Number of samples = 120			Number of samples = 91			Number of samples = 350			Number of samples not stated		Number of samples = 209		
	Depth = 0–150 mm			Depth = 0–100 mm			Depth = 0–150 mm			Depth not stated		Depth = 1–100 mm		
Metal	**Range mg/kg**	**Mean mg/kg**	**Median mg/kg**	**Range mg/kg**	**Mean mg/kg**	**Median mg/kg**	**Range mg/kg**	**Mean mg/kg**	**Median mg/kg**	**Range mg/kg**	**Mean mg/kg**	**Range mg/kg**	**Mean mg/kg**	**Median mg/kg**
Al														
As	5–53	7	5	1–20	3	3								
Be														
B[a]												0.09–8.0	0.87	0.61
Cd				0.016–2.0	0.195	0.125	0.01–0.78	0.15	0.11					
Cr	5–56	8	5	<9–573	132	65	2.5–673	115	41					
Co				<6–165	37	26	0.4–147	16	6	<2–170	11			
Cu	3–412	16	9	<8–148	43	38	0.4–200	22	13	<1–190	22			
Fe														
Pb	5–56	14	14	5–81	27	24	2–60.5	14	12					
Li														
Mn	4–7357	814	201							4–5100	780			
Hg				<0.006–0.15	0.042	0.035								
Mo				0.2–5.2	1.34	1.01				<1–20	3.2			
Ni	5–38	6	5	<10–439	88	27	1–517	56	18					
Se				<0.05–3.2	0.37	0.28								
U														
V	5–121	21	12											
Zn	5–92	21	10	<12–263	76	73	1–219	48	32	<2–180	34			

a Soil boron determined as hot 0.01M $CaCl_2$ extractable

Table 9.2.17 Summary of agricultural irrigation water long-term trigger value (LTV), short-term trigger value (STV) and soil cumulative contaminant loading limit (CCL) guidelines for heavy metals and metalloids

Metal	Suggested upper background values[a]	NSWEPA (1995b) Biosolid guidelines food production	New Zealand DoH (1992) Biosolid guidelines soil metal limits	NSWEPA (1998) Draft phytotoxicity investigation levels	Calculated soil CCL	Suggested soil CCL	LTV in irrigation water	STV in irrigation water
	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(kg/ha)	(kg/ha)	(mg/L)	(mg/L)
Al	–	–	–	–	–	ND	5	20
As	10	20	10	20	20	20	0.1	2.0
Be	–	–	–	–	–	ND	0.1	0.5
B	1.0[b]	–	–	–	–	ND	0.5	Refer to Table 9.2.18
Cd	0.12	1	3	3	2	2	0.01	0.05
Cr (VI)	–	–	–	10	–	ND	0.1	1
Co	27	–	–	–	–	ND	0.05	0.1
Cu	28	100	140	100	140	140	0.2	5
F	–	–	–	–	–	ND	1	42
Fe	–	–	–	–	–	ND	0.2	10
Pb	18	150	300	600	257	260	2	5
Li	–	–	–		–	ND	2.5 (0.075 citrus crops)	2.5 (0.075 citrus crops)
Mn	201	–		–	–	ND	0.2	10
Hg	0.03	1	1	–	1.89	2	0.002	0.002
Mo	1	–		–	–	ND	0.01	0.05
Ni	17	60	35	60	84	85	0.2	2
Se	0.5	5	–	–	9	10	0.02	0.05
U	–	–		–	–	ND	0.01	0.1
V	–	–		–	–	ND	0.1	0.5
Zn	39	200	300	200	313	300	2	5

a Median values from Australian soil surveys (non-contaminated sites): Olszowy et al. (1995), Barry (1997); CSIRO Land and Water (unpublished data); Tiller (1983) — (table 9.2.16)

b $CaCl_2$ extractable boron, based on South Australian surveys of Murray Mallee, Eyre Peninsula and Upper SE [Spouncer & Mowat. CSIRO Technical Bulletin (1991a–d)]

ND = Not determined, insufficient background data to calculate CCL

9.2.5.3 Aluminium

It is recommended that the concentration of aluminium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		5.0 mg/L
Short-term trigger value in irrigation water	**Short-term use**	20 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Aluminium metal does not occur naturally, but aluminium is found in abundance in the geosphere (81 g/kg in the earth's crust) in complexes with oxygen, fluorine, and silicone. Aluminium compounds are very stable, due to aluminium's ionic radius of 57 pm, high oxidation potential (1.66 V) and oxidation state (+3). All soils contain aluminium compounds, mostly in aluminino-silicate minerals, although aluminium may be present in ion-exchangeable form in acidic soils (Scott-Fordsmand & Peterson 1995).

Crop yield and quality considerations

Toxicity of aluminium to field crops is an important cause of reduced productivity on acid soils, because the soluble aluminium concentration 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, and the irrigation water STV and LTV guidelines have been developed to minimise the risk of phytotoxicity. A CCL for aluminium has not been determined, as it is inappropriate to set a CCL for a major soil constituent.

9.2.5.4 Arsenic

It is recommended that the concentration of arsenic in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.1 mg/L
Short-term trigger value in irrigation water	**Short-term use**	2.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water		20 kg/ha

Arsenic can exist in anionic (negatively charged) and cationic (positively charged) form, the charge dictating the behaviour of the element in soil. Chemically, arsenic behaves in a similar way to phosphorus, and therefore arsenic compounds compete with their phosphorus analogues. In the 3^+ ionic form, arsenic compounds (arsenite) are water soluble as a cation, as a negative oxy-ion, as the hydroxide, and as the negative sulpharsenite ion. As_2O_3 (arsenic trioxide, white arsenic) exists in various forms. Representatives of the 5^+ oxidation state are As_2O_5, (arsenic pentoxide), As_2S_5 (arsenic pentasulphide), As_2Se_5 (arsenic pentaselenide) and arsenates.

Crop yield and quality considerations

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 5^+) and arsenite (arsenic 3^+) 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). Typical concentrations of arsenic in uncontaminated freshwaters are <1 µg/L (DWAF 1996a). The median arsenic concentration in uncontaminated Australian soils is 5 mg/kg (table 9.2.16). 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). Will and Suter (1994a) derived a solution concentration phytotoxicity benchmark for arsenic of 0.001 mg/L. These studies indicate that as well as phytotoxicity resulting from elevated soil arsenic concentrations, there is the potential for direct phytotoxicity to crops of arsenic in irrigation waters. The LTV and STV guidelines for irrigation waters have therefore been derived to protect crops from the direct phytotoxic effects of arsenic in irrigation waters. Existing environmental guidelines for arsenic (NSWEPA 1995a) and suitable soil background data have allowed the derivation of a CCL limit for arsenic in soils.

9.2.5.5 Beryllium

It is recommended that the concentration of beryllium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.1 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.5 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Beryllium is commonly found in silicate and oxide minerals, predominantly as beryl, a beryllium aluminium silicate. The silicate and carbonate forms are insoluble in water and are generally bound tightly to sediments.

Crop yield and quality considerations

Beryllium is toxic to both animals and plants. There are no primary research data describing the toxicity of beryllium to plants grown in soil. However, in solution culture studies the lowest beryllium concentration at which reductions in germination and vegetative growth were noted was 0.5 mg/L (Will & Suter 1994a). Romney and Childress (1965) reported that 2 mg/L in nutrient solutions reduced the growth of various plant species. Toxicity is likely to be greater in acid soils (Williams & LeRiche 1968). The translocation of beryllium from the roots of the plants to the foliage does not occur readily (Gough et al. 1979). As there are no data on background concentrations of beryllium in Australian soils at present, and concentrations in unpolluted waters are expected to be in the µg/L range (DWAF 1996a), it is not possible at this stage to determine a soil CCL for beryllium.

9.2.5.6 Boron

It is recommended that the concentration of boron in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.5 mg/L
Short-term trigger value in irrigation water	**Short-term use**	Refer to values in table 9.2.18. Dependent on crop type.
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Boron is present in the environment as borates and borosilicate minerals, such as borax associated with salt deposits in saline lakes, borate and aluminium borosilicate. Boron is commonly associated with saline hydrogeological conditions.

Crop yield and quality considerations

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 9.2.18. Boron is generally sorbed onto soil surfaces at alkaline pH values. High boron concentrations in soils have been shown to cause plant toxicity in northern Victoria (Sauer 1958). Total concentrations of boron in soils of 10 mg/kg have been shown to cause no adverse effects on plants (Will & Suter 1994a). However, unlike the other elements described in this guideline, limits for boron in soil have been set using concentrations determined by hot 0.01M $CaCl_2$ extraction, as this relates to the plant available fraction in soil. In a survey of South Australian agricultural soils the median concentration of $CaCl_2$ extractable boron was 0.61 mg/kg (table 9.2.16), placing these soils just above the limit for very sensitive crops. In general, maximum concentrations of boron tolerated by plants in irrigation water without reduction in yield or vegetative growth are approximately equal to soil water boron concentrations listed in table 9.2.18 (ANZECC 1992).

Table 9.2.18 Relative tolerance of agricultural crops to boron[a]

Tolerance	Concentration of boron in soil water (mg/L)	Crop
Very sensitive	<0.5	Blackberry, lemon
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, lime beans
Moderately sensitive	1.0–2.0	Capsicum, pea, carrot, radish, potato, cucumber
Moderately tolerant	2.0–4.0	Lettuce, cabbage, celery, turnip, 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

a From Westcot & Ayers (1984), cited by ANZECC (1992)

Therefore, for crop protection from boron toxicity it is recommended that the values listed in table 9.2.18 are used to determine the STV in irrigation waters. The LTV has been set to protect the most sensitive species. In general toxic concentrations of boron are associated with irrigation waters derived from groundwater or secondary wastewater (Ayers & Westcot

1976). It is recommended that additional water quality monitoring of boron should be undertaken if these sources of irrigation water are used. Insufficient data are available at present to allow the determination of a CCL for boron.

9.2.5.7 Cadmium

It is recommended that the concentration of cadmium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.01 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.05 mg/L
Cumulative contaminant loading in soil receiving irrigation water		2 kg/ha

Cadmium in its pure form is a relatively soft, silver-white, lustrous and ductile metal. It is readily soluble in nitric acid, but only slowly soluble in hydrochloric and sulphuric acid and insoluble in basic solutions. Salts of cadmium with strong acids are readily soluble in water, whereas cadmium sulphide, carbonate, fluoride and hydroxide are less soluble. In the presence of organic material, cadmium has a high affinity for thiol and hydroxyl groups, for example, proteins, enzymes, and other essential compounds (Scott-Fordsmand & Pederson 1995).

Crop yield and quality considerations

Cadmium is toxic to both animals and plants at 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 carcinogenity also may be a possibility (Merian 1984). Uncontaminated soils in Australia generally contain around 0.05–0.10 mg/kg cadmium (McLaughlin et al. 1996), but fertilised agricultural soils may contain higher concentrations due to addition of phosphate fertiliser containing cadmium as an impurity, or additions of manures, composts or biosolids (McLaughlin et al. 1996). In rural areas, inputs of cadmium via atmospheric deposition may also contribute to elevated concentrations of cadmium in the soil (Merry & Tiller 1991). Although it is not required for metabolism, cadmium is readily taken up by plants and uptake increases with soil acidity, soil salinity and the total content of cadmium in the soil system (Herms & Brümmer 1984, McLaughlin et al. 1994). Chloride concentration in irrigation water is important in controlling cadmium uptake by plants and should also be considered (see Section 9.2.4.2, table 9.2.13).

As cadmium is similar to zinc (an essential element for plant growth), it can readily interfere with metabolic processes within the plant by blocking zinc binding sites. The absorption of cadmium by the plant can be minimised by ensuring soils are not acidic or saline, and ensuring a good supply of zinc, manganese and copper (Cataldo et al. 1983, Oliver et al. 1994). Cadmium in nutrient solutions is phytotoxic to a range of plants at levels ranging from 0.1 mg/L to 1 mg/L (Will & Suter 1994a), but human and animal health concerns from ingestion of cadmium-contaminated crops are triggered at sub-phytotoxic concentrations. The LTV and STV have therefore been set to prevent the uptake of cadmium into crops that may pose a threat to animal and human health. Given the existence of a reasonable data set in Australia for soil background cadmium concentrations, and existing cadmium limits for agricultural soils receiving biosolids (NSWEPA 1995a), a cadmium CCL has been derived for soils receiving irrigation.

9.2.5.8 Chromium (VI)

It is recommended that the concentration of chromium (VI) in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water	0.1 mg/L
Short-term trigger value in irrigation water **Short-term use**	1.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water	Not determined

In its pure form chromium is a steel-grey, bright, brittle and very hard metal and is resistant to corrosion. It is known in all oxidation states from -2 to +6, with +3 (chromic) and +6 (chromate) being the most common in soils. Oxidation states below +3 are reducing and oxidation states above are oxidising.

Crop yield and quality considerations

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). 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 50–5000 mg/kg, depending on plant species and soil type (NRCC 1976). Because translocation of chromium within the plant does not occur readily, most of the absorbed chromium remains in the roots (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 South Australian EPA has de-regulated chromium from its biosolid land application guidelines as chromium is predominantly present in biosolids as the chromium (III) ion. However, SAEPA has placed the proviso in the current regulations that future reviews aim at determining a limit for chromium (VI) in soils (SAEPA 1996). Depending on prevailing redox conditions in soil, chromium (III) can be oxidised to the more mobile chromium (VI); manganese oxides and organic matter play an important role in this reaction as electron acceptors (McGrath 1995). However, in agricultural soils with normal Eh and pH ranges, chromium (VI) is likely to be reduced to the chromium (III) ion.

Studies with nutrient solutions indicate that there may be some direct phytotoxic effect on irrigated crops of chromium in irrigation waters. Concentrations of 1–10 mg/L in nutrient solutions reduce crop yield, depending on the tolerance of different plant species (NAS 1974), and there is limited evidence that chromium (III) and chromium (VI) in nutrient solutions are about equally available to plants (Will & Suter 1994a). It is therefore inappropriate to set a guideline based on total chromium or chromium (III) due to the lack of evidence that chromium (III) poses a significant environmental or phytotoxic threat. Guidelines are therefore set for the chromium (VI) ion in irrigation waters based on the revised South African irrigation water quality guidelines (DWAF 1996a). As there are no available data on background concentrations of chromium (VI) in Australian soils or chromium (VI) toxic thresholds for soils, it is not possible to set a CCL limit at this stage. However, it is recommended that future guidelines attempt to set chromium (VI) soil limits.

9.2.5.9 Cobalt

It is recommended that the concentration of cobalt in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.05 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.1 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Cobalt occurs as various sulfide ores in nature and is generally associated with arsenic, iron, nickel, and copper. Concentrations in unpolluted surface waters are generally in the order of <1µg/L (DWAF 1996a). The chemical properties of cobalt are similar to iron and nickel, however unlike the Fe (II) ion, Co (II) is quite stable in soils.

Crop yield and quality considerations

Cobalt is not considered to be an essential plant micronutrient, with the exception of legumes involved in symbiotic nitrogen fixation with *Rhizobia*. Cobalt in soils tends to be tightly bound to manganese oxides. However, this reaction is pH dependent and increased cobalt uptake into plants has been observed with decreasing pH (Smith & Paterson 1995). Hodgson (1960) reported a strong interaction between cobalt and most soils at neutral and alkaline pH values. The field occurrence of cobalt toxicity is rare (Hart 1974), and Vanselow (1966) showed that high concentrations of cobalt (100 mg/kg) had little effect on citrus crops, probably due to adsorption of cobalt by soil particles. While there is little evidence of cobalt toxicity due to elevated soil concentrations, evidence for potential toxicity due to cobalt in irrigation waters comes from nutrient solution studies. Will and Suter (1994a) noted that concentrations of cobalt in solution of 0.06 mg/L may reduce the vegetative growth of plants. DWAF (1996a) notes that cobalt in nutrient solution has been found to be toxic to tomatoes at a concentration of 0.1 mg/L, and that this concentration approximates a toxicity threshold for other plants. Given these data, a LTV of 0.05 mg/L in irrigation water is considered appropriate and protective for continuous use. The STV in irrigation water has been set at 0.1 mg/L in order to protect crops from direct effects of irrigation waters. Given the paucity of data relating to phytotoxic concentration thresholds of cobalt in soils and the fact that there are no regulations relating to cobalt limits in Australian soils, a cobalt CCL has not been determined at this stage.

9.2.5.10 Copper

It is recommended that the concentration of copper in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.2 mg/L
Short-term trigger value in irrigation water	**Short-term use**	5.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water		140 kg/ha

Copper, in its pure form, is a reddish coloured metal. Copper is a near noble metal, only dissolving in oxidising acids.

Crop yield and quality considerations

Copper is an important component of several plant enzymes and therefore is 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 1987). The median concentration of copper in uncontaminated Australian soil is 28 mg/kg (table 9.2.17). However copper concentrations in soils can range from 0.4–412 mg/kg (table 9.2.16). Higher concentrations can occur due to application of biosolids, copper-based fungicides (e.g. vineyards), and animal manures. 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 & Brümmer 1984, Sanders 1982), and toxicity is therefore related to the pH of the soil.

Copper toxicity from nutrient solutions has been noted at concentrations between 0.1 and 1.0 mg/L with concentrations of 0.03 mg/L causing growth reductions in one study (Will & Suter 1994a). It is therefore a possibility that elevated levels of copper in irrigation water may have a direct phytotoxic effect on plants. In order to prevent this the LTV for copper has been set at 0.2 mg/L. Given the existence of datasets for background concentrations of copper in Australian soils, and existing copper limits for agricultural soils receiving biosolids (NSWEPA 1995a), a copper CCL has been derived for soils receiving irrigation.

9.2.5.11 Fluoride

It is recommended that the concentration of fluoride in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		1.0 mg/L
Short-term trigger value in irrigation water	**Short-term use**	2.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Fluorine has a higher oxidation potential than ozone and is the most electronegative element. It reacts vigorously with most oxidisable substances at room temperature. Fluorine does not occur free in nature, but is the most reactive metalloid and binds, directly or indirectly, to form fluorides with all the elements except the inert gases. The occurrence of fluoride in the earth's crust is 0.027%.

Crop yield and quality considerations

Fluoride has been found to occur naturally in all soils. Total fluoride concentrations in soils range from trace amounts to 7000 mg/kg but are generally below 200 mg/kg (Moen et al. 1986). Freshwater usually contains less than 2 mg F/L (WHO 1970). Most irrigation water also contains less than 2 mg F/L, although this is dependent on the sources of the water. Excessive intake of fluoride can lead to dental and skeletal fluorosis, characterised by hypermineralisation of bones, causing them to become brittle. The margin between beneficial and detrimental concentrations is small.

In the majority of soils, a high proportion of added fluoride is firmly retained by the soil. In general, slightly acid soils (pH 5.5–6.5) have the greatest affinity for fluoride (Larsen & Widdowson 1971). Due to the ability of the soil to rapidly adsorb fluoride, soil retains a large portion of the fluoride added, and fluoride contamination of groundwater through irrigation with water containing high concentrations of fluoride is unlikely. There do not appear to be any data indicating phytotoxic effects of fluoride in soil. However, recent solution culture

data suggest uptake and toxicity of fluoride are dependent on the ionic species of fluoride in the solution exposed to the plant root (Stevens et al. 1997). Toxic concentrations of fluoride in solution culture ranged from 1 to >100 mg F/L depending on ionic species of fluoride present and plant species.

Regular consumption by stock of water containing fluoride concentrations greater than 2 mg/L progressively increases the risk of fluorosis (see Section 9.3.5.9). The LTV has been set on the assumption that irrigation water could potentially be phytotoxic to sensitive plants or contaminate stock drinking water. The STV has been set on the assumption that irrigation water could potentially contaminate stockwater. A CCL has not been determined for fluoride, as there are insufficient data for Australian soils to determine background concentrations and soil concentrations, which may be phytotoxic.

9.2.5.12 Iron

It is recommended that the concentration of iron in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.2 mg/L
Short-term trigger value in irrigation water	**Short-term use**	10 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

The occurrence of iron in the earth's crust is 4.7%. Iron is a silvery-white or grey, hard, ductile, malleable, somewhat magnetic metal. It is stable in dry air but readily oxidises in moist air, forming rust. In water, iron can be present as dissolved ferric iron, Fe(III), as ferrous iron, Fe(II) or as suspended iron hydroxides.

Crop yield and quality considerations

Most soils are naturally rich in iron. The soil pH and aeration determine the oxidation state and thus solubility of iron in soil. Iron is an essential micro-nutrient and plant deficiency results in chlorosis. There are insufficient data to determine a toxicity threshold of iron for plants growing in soils and there are no known direct negative effects of iron in soil (Will & Suter 1994a, DWAF 1996a). However, there have been a few reports of iron concentrations of approximately 10–50 mg Fe/L present in solution culture reducing plant growth (Will & Suter 1994a).

Iron deficiency develops mostly in alkaline soils, where iron precipitates as hydroxides and becomes unavailable to plants. Ferrous iron dissolved in irrigation water is relatively unavailable to plants as it oxidises (ferric iron) and precipitates upon aeration when applied to soil. However, under reducing conditions (waterlogging) precipitated ferric iron can be reduced to the more soluble ferrous iron. Precipitated iron in soils binds phosphorus and molybdenum (essential plant nutrients), making them unavailable to the plant.

Iron dissolved in irrigation water can cause problems when it precipitates on plant leaves or in irrigation equipment. Dissolved iron in irrigation water is relatively common in Australia (ANZECC 1992). It precipitates on aeration and concentrations less than 5 mg Fe/L may produce light-brown spotting on plants (Hart 1974, DWAF 1996a). Concentrations of iron less than 0.2 mg/L will cause only minor problems with clogging of trickle or drip irrigation systems, while concentrations above 1.5 mg Fe/L may cause severe problems (ANZECC 1992, DWAF 1996a).

In view of the potential clogging of irrigation systems (trickle or drippers), the LTV has been set to ensure minimal problems with this type of irrigation technique and to ensure minimal plant foliage damage or blemishes by iron deposits when irrigating. If trickle or dripper irrigation techniques are not used, or plants are not sprayed with irrigation water, then higher concentrations of iron will be acceptable. The STV has been set so that continual irrigation of plants will not expose them to phytotoxic concentrations of iron. A CCL for iron has not been determined, as it is inappropriate to set a CCL for a major soil constituent.

9.2.5.13 Lead

It is recommended that the concentration of lead in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		2.0 mg/L
Short-term trigger value in irrigation water	**Short-term use**	5.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water		260 kg/ha

Lead in its pure form is a bluish-white metal of bright lustre, is soft, highly malleable, ductile, and a poor conductor of electricity. It is very resistant to corrosion. Lead chloride and bromide salts are slightly soluble (1%) in cold water, whereas carbonates and hydroxide salts are almost insoluble (Adriano 1986). Lead is a natural constituent of the earth crust. It is the most abundant among the heavy metals with an atomic number >60. It is present in a series of different metals of which the most important economically are Galena (PbS), Cerussite ($PbCO_3$) and Anglesite ($PbSO_4$) (Scott-Fordsmand & Pederson 1995).

Crop yield and quality considerations

Lead is strongly retained by most soils (Elliott et al. 1986) so that soil solution lead concentrations are very low (<1 mg/L), especially in relation to other metals like cadmium, zinc and copper (Brümmer & Herms 1983). As for other cationic metals, low soil pH mobilises lead in soil allowing greater plant uptake (von Judel & Stelte 1977). Due to the strong sorption by soils, surface applications of lead, whether from atmospheric sources, inadvertent additions in fertilisers, manures or sludges, or deliberate use of lead-containing agricultural chemicals, are retained in the upper or plough layer of soil profiles (Merry et al. 1983).

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). Horses appear to be among the animals most sensitive to lead poisoning; chronic poisoning occurred after consuming grass contaminated with lead at concentrations of 5–20 mg/kg (dry weight) (Singer 1976). Phytotoxic concentrations of lead in soils have been noted at concentrations ranging from 250–500 mg/kg, while phytotoxicity of lead in solution has been observed at concentrations of 10 mg/L (Will & Suter 1994a). Given the evidence from solution culture of potential direct lead toxicity to plants, the STV and LTV have been set in order to minimise these risks. Given the existence of datasets for background concentrations of lead in Australian soils, and existing lead limits for agricultural soils receiving biosolids (NSWEPA 1995a), a lead CCL has been derived for soils receiving irrigation.

9.2.5.14 Lithium

It is recommended that the concentration of lithium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		2.5 mg/L (0.075 mg/L if used on citrus crops)
Short-term trigger value in irrigation water	**Short-term use**	2.5 mg/L (0.075 mg/L if used on citrus crops)
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Lithium generally occurs in association with aluminosilicate and aluminium fluorophosphates. Higher concentrations tend to be found in association with hot springs in arid hydrogeological conditions. Typical lithium concentrations in unpolluted freshwaters are 0.02 mg/L. A monovalent cation, lithium is easily displaced by other cations in soil solution and is relatively mobile.

Crop yield and quality considerations

No data are available on the background concentrations of lithium in Australian soils. Soil concentrations of between 2 and 50 mg Li/kg in soil have been shown to be toxic to a range of crops (Will & Suter 1994a), the lower values observed with citrus crops. Usually crops sensitive to sodium are also affected by high lithium concentrations, and lithium uptake appears to share the potassium transport carrier. Lithium has a similar (but less severe) effect on soil physical structure to sodium, however, phytotoxicity occurs at much lower concentrations than effects on soil structure (DWAF 1996a). Potential direct toxic effects of lithium in irrigation waters are suggested from results of solution culture studies. Except for citrus trees, most crops can tolerate up to 5 mg/L lithium in nutrient solution (NAS/NAE 1973). Will and Suter (1994a) suggest a phytotoxicity benchmark of 3 mg/L in solution for crops excluding citrus. 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). The STV and LTV for lithium in irrigation waters have both been set at 2.5 mg/L, based on the potential for direct irrigation water toxicity to the majority of crops. However, if irrigation is applied to citrus crops a limit of 0.075 mg/L is recommended. Due to lack of data on lithium concentrations in soils or lithium toxicity thresholds in soils, a CCL limit has not been determined.

9.2.5.15 Manganese

It is recommended that the concentration of manganese in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.2 mg/L
Short-term trigger value in irrigation water	**Short-term use**	10 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Crop yield and quality considerations

Manganese is a major constituent of soils and its solubility is controlled by pH and oxidation-reduction reactions, which control solubility and sorption reactions of manganese with soil. Manganese in soil solution is found predominantly as the Mn^{2+} ion.

Manganese concentrations in soil solution are increased under reducing conditions (waterlogging) and at low soil pH values. The staining problems associated with natural waters containing high concentrations of Mn^{2+} are due to oxidation of the Mn^{2+} to form a black hydrated oxide (MnO_2).

Manganese is essential for plant growth, as it is involved in nitrogen metabolism and in the synthesis of chlorophyll. Manganese is low in toxicity to animals and humans unless ingested in large amounts (NAS/NAE 1973). However, at high concentrations in solution, manganese may be highly toxic to plants, especially to root growth in acidic soils. In nutrient solutions, toxicity to plant roots occurs at solution concentrations as low as 0.75 mg/L (Will & Suter 1994a). Manganese may also cause clogging of irrigation equipment due to oxidation of Mn^{2+} to MnO_2. LTV and STV guidelines have therefore been set to protect against direct phytotoxicity and damage to irrigation infrastructure. A CCL for manganese has not been determined as it is inappropriate to set a CCL for a major soil constituent.

9.2.5.16 Mercury

It is recommended that the concentration of mercury in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.002 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.002 mg/L
Cumulative contaminant loading in soil receiving irrigation water		2 kg/ha

Mercury in its pure form is a silvery lustrous metal, which is liquid at room temperature and standard atmospheric pressure. Mercury dissolves several other metals forming amalgams. Considering biological activity, mercury can be separated into three main categories: metallic mercury, which has a high vapour pressure and thus vaporises under atmospheric pressure; inorganic ions (mercury may exist as Hg^{+} and Hg^{2+}, bivalent mercury readily forms complexes with organic ligands, and monovalent mercury binds less readily to organics and forms less water soluble salts); and organic mercury, which consists of mercury covalently bound to carbon.

Crop yield and quality considerations

Mercury is strongly retained by soils, especially by those high in organic matter. Median background concentrations of mercury in Australian soils are 0.03 mg/kg, derived from values in table 9.2.16. Most plants do not readily take up mercury (Hart 1982, Schachtschabel et al. 1989). Lettuce grown on contaminated soil (7 mg Hg/kg) showed only a small increase in mercury absorption (MacLean 1974a); however, carrots and mushrooms can accumulate mercury from soils. There are no reference data describing the toxicity of mercury to plants in soil (Will & Suter 1994a). Will and Suter (1994a) derived a solution phytotoxicity benchmark of 0.004 mg/L for inorganic mercury and 0.002 mg/L for organic mercury. Given that solution culture studies indicate that there may be direct phytotoxic effects to plants of mercury in irrigation waters, the LTV and STV have both been set at 0.002 mg/L total mercury. Given the existence of datasets for background concentrations of mercury in Australian soils, and existing mercury limits for agricultural soils receiving biosolids (NSWEPA 1995a), a mercury CCL has been derived for soils receiving irrigation.

9.2.5.17 Molybdenum

It is recommended that the concentration of molybdenum in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.01 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.05 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Molybdenum is an essential micro-nutrient for all living organisms, having an important role in enzyme synthesis and activity. However, excess molybdenum is toxic. Concentrations of molybdenum in unpolluted freshwaters typically range between 0.03 and 10 μg/L. Molybdenum commonly exists as an anion in waters and soils. Behaviour of molybdenum in soils is similar to other negatively charged elements which tend to be very mobile. Soil anion exchange capacity increases with decreasing soil pH, therefore under acidic conditions molybdenum is less available to plants.

Crop yield and quality considerations

The concentration of molybdenum in soils ranges from 0.1–40 mg/kg (DWAF 1996a). Median concentrations in Australian soils are 1.0 mg/kg with a range of 0.2–20 mg/kg (table 9.2.16). Plants absorb molybdenum predominantly as the MoO_4^{2-} anion from the soil solution and can concentrate it in tissue. Tissue concentrations of >100 mg/kg apparently have no adverse effects on plant growth. Accumulation of molybdenum by crops is higher in alkaline soils due to higher MoO_4^{2-} concentrations in the soil solution. Molybdenum accumulation in plant tissue may be harmful to livestock consuming contaminated feed, causing molybdenosis, which has been observed in cattle consuming legumes grown in soil solution concentrations of 0.01 mg/L of molybdenum (DWAF 1996a). High levels of molybdenum in livestock diets may also induce copper deficiency. Toxic effects of molybdenum in forage crops are considered to occur at above 5 mg/kg for cattle and 10 mg/kg for sheep (Dye 1962). There is limited evidence for the phytotoxic impacts of molybdenum in soils and irrigation water. Solution culture studies have reported toxicity to plants at concentrations as low as 0.5 mg/L (Will & Suter 1994a). Given that toxic concentrations of molybdenum may arise in herbage at soil solution concentrations apparently below those at which phytotoxicity is noted, the LTV and STV have been set at levels designed to prevent the build-up of molybdenum in soils that could raise molybdenum levels in crops above 10 mg/kg, thus protecting grazing livestock. A CCL limit has not been set for molybdenum due to lack of soil data or soil toxicity benchmarks.

9.2.5.18 Nickel

It is recommended that the concentration of nickel in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.2 mg/L
Short-term trigger value in irrigation water	**Short-term use**	2.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water		85 kg/ha

Nickel is a silvery-white metal which is hard, malleable, ductile, and a good conductor of heat and electricity. As a natural composite of the earth crust, nickel is mainly present in igneous rocks and is ubiquitous in the environment (Scott-Fordsmand & Pederson 1995).

Crop yield and quality considerations

Nickel concentrations in soils in Australia range from 5 mg/kg to 520 mg/kg with an average <100 mg/kg (CSIRO, unpublished). Soils developed from serpentine rocks contain much higher quantities of nickel (400–500 mg/kg). Soil nickel concentrations toxic to plants vary, depending on the soil conditions, particularly soil texture, organic matter content and soil pH. Nickel is sorbed strongly to most soils. Below pH6 the concentration of soluble and exchangeable nickel increases considerably (Herms & Brümmer 1984).

Although nickel is now accepted as an essential micro-nutrient for plant growth (Marschner 1995), nickel has never been found to be deficient in soils due to the ubiquitous nature of nickel in the environment. Concern for nickel phytotoxicity stems from the use of biosolids of high nickel content on soils, where concentrations in soil may reach phytotoxic levels. Soils most at risk from nickel phytotoxicity are acidic light-textured soils low in organic matter. Nickel concentrations in nutrient solutions of 0.13–2 mg/L are toxic to a number of plants (Will & Suter 1994a). The LTV and STV guidelines for nickel have therefore been set to reduce the risk of direct nickel toxicity to plants. Given the existence of datasets for background concentrations of nickel in Australian soils, and existing nickel limits for agricultural soils receiving biosolids (NSWEPA 1995a), a nickel CCL has been derived for soils receiving irrigation.

9.2.5.19 Selenium

It is recommended that the concentration of selenium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.02 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.05 mg/L
Cumulative contaminant loading in soil receiving irrigation water		10 kg/ha

Selenium is a metalloid element, found in conjunction with sulfide ores of copper, iron and zinc. Selenium is an essential human and animal micro-nutrient at low concentrations, responsible for the activity of the enzyme glutathione peroxidase. Concentrations in unpolluted surface waters are generally in the order of <10 μg/L (DWAF 1996a). Selenium occurs in soils as selenite (SeO_3^{2-}) and selenate (SeO_4^{2-}). Soil behaviour is similar to other anions such as molybdenum in that bioavailability and mobility are high. In acid soils containing iron and aluminium oxides, selenite forms low solubility complexes with the oxide fractions. In alkaline soils selenium occurs as selenate which is highly mobile.

Crop yield and quality considerations

The median background concentration of selenium in Australian soils is 0.5 mg/kg, values ranging from 0.05 to 3.2 mg/kg (table 9.2.16). The main issue regarding selenium in irrigation water is elevated concentrations of selenium in forage crops and toxicity of selenium to animals consuming high selenium fodder. Selenium concentrations of 0.03–0.1 mg/kg in forage are required by cattle to prevent deficiency. However, selenium concentrations in fodder above 5 mg/kg (Horvath 1976) are considered potentially toxic. This concentration of selenium can arise in plants grown in soils with a solution concentration of 0.05 mg/L selenium (DWAF 1996a). Plants can absorb relatively large amounts of selenium

without displaying any phytotoxicity symptoms. Will and Suter (1994a) derived a soil phytotoxic benchmark of 1 mg/kg, and a solution phytotoxic benchmark of 0.7 mg/L. Levels of selenium toxic to grazing animals can be reached in plant material long before solution concentrations become phytotoxic. On this basis the irrigation water quality LTV and STV concentrations have been set in order to prevent selenium toxicity to grazing livestock feeding on forage receiving irrigation. Given the existence of datasets for background concentrations of selenium in Australian soils, a soil CCL limit has been set for selenium based on current regulatory guidelines (NSWEPA 1995a).

9.2.5.20 Uranium

It is recommended that the concentration of uranium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.01 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.1 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Uranium is a naturally radioactive element and is a chemically reactive cation forming compounds with anions such as fluoride, phosphorus and arsenic. As with most other cations uranium binds strongly to negatively charged soil surfaces. Typical concentrations of uranium in surface soils range from 0.7–9 mg/kg, and in unpolluted surface waters concentrations are around 0.4 mg/L (DWAF 1996a).

Crop yield and quality considerations

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). Phytotoxicity as a result of elevated uranium concentrations in soils is thought to involve inhibition of enzyme systems and binding to nucleic acids. Will and Suter (1994a) note that phytotoxicity is considered to be the result of the element itself rather than any radiation associated with the isotope. Zhukov and Zudilkin (1971) reported that 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). From the limited data available, plant yield appears to remain unaffected by uranium concentrations in soil of <10 mg/kg (Will & Suter 1994a), therefore LTV and STV irrigation guidelines have been set to prevent soil uranium concentrations exceeding 10 mg/kg. It should be noted that these assumptions are based on limited data and assume that concentrations of uranium in irrigation waters will be negligible. Insufficient data are available at this stage to develop a CCL limit for uranium.

9.2.5.21 Vanadium

It is recommended that the concentration of vanadium in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		0.1 mg/L
Short-term trigger value in irrigation water	**Short-term use**	0.5 mg/L
Cumulative contaminant loading in soil receiving irrigation water		Not determined

Metallic vanadium does not occur in nature, vanadium being generally present as sulfide and calcium salts. In common with other positively charged elements vanadium is sorbed by the soil, however soluble vanadium salts are taken up by plants and animals. Of the four common oxidation states V^{4+} and V^{5+} are the most bioavailable as they remain in the soil solution phase and are not strongly sorbed to soil surfaces. Concentrations of vanadium in surface soils range from 20–250 mg/kg (Edwards et al. 1995).

Crop yield and quality considerations

In Australian soils the median concentration of vanadium from the survey of Olszowy et al. (1995) was 12 mg/kg (table 9.2.16). Concentrations in uncontaminated surface waters are generally <1 µg/L (DWAF 1996a).

Vanadium is not known to be an essential element for crop growth, however there is evidence for its involvement in symbiotic nitrogen fixation. Toxic effects are thought to be the result of interference with enzyme systems, resulting in reduced growth, and interference with the adsorption of essential elements such as calcium, copper, iron, manganese and phosphorus (Warrington 1955, Cannon 1963, Wallace et al. 1977). After plant uptake most vanadium remains in roots (Will & Suter 1994a). Depending on soil type and species of plant, vanadium concentrations of 10 mg/kg soil are thought to inhibit crop growth (DWAF 1996a). Will and Suter (1994a) set a vanadium solution concentration toxic benchmark of 0.5 mg/L. The limited information available indicates phytotoxicity at soil concentrations of >10 mg/kg and solution concentrations of 0.5 mg/L. The LTV and STV been derived on the basis that vanadium concentrations in irrigation waters will be negligible, and to prevent direct phytotoxic effects of irrigation waters on plants. Insufficient data are available to determine a soil vanadium CCL limit at this time.

9.2.5.22 Zinc

It is recommended that the concentration of zinc in irrigation waters and soils should be less than the following:

Long-term trigger value in irrigation water		2.0 mg/L
Short-term trigger value in irrigation water	**Short-term use**	5.0 mg/L
Cumulative contaminant loading in soil receiving irrigation water		300 kg/ha

Zinc is a natural composite of the earth crust, present in range of minerals, for example, sphalite (ZnS), smithsonite ($ZnCO_3$) and hemimorphite ($Zn_4(OH)_2Si_2O_7H_2O$) (Scott-Fordsmand & Pederson 1995). Zinc sulphate, nitrate and halides (except fluorides) are readily soluble in water, while zinc carbonate, oxide, phosphate and silicate are sparingly soluble or insoluble in water (CRC 1982). In the presence of organic material, zinc has a high affinity for thiol and hydroxyl groups such as in proteins, enzymes and other essential compounds.

Crop yield and quality considerations

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 0.4–6.5 mg/L (Will & Suter 1994a). Zinc toxicity in plants is evidenced by chlorosis, reduction in leaf size, necrosis of tips and distortion of foliage (Chapman 1966), although effects on symbiotic nitrogen-fixing bacteria may occur at lower soil zinc concentrations than those causing direct

phytotoxicity (Chaudri et al. 1993). Zinc is more readily available to plants in acid (pH_{CaCl_2} <6) light-textured soils (MacLean 1974b, MacLean & Dekker 1978, Hornburg & Brümmer 1989). The LTV and STV for zinc have therefore been set to minimise the potential phytotoxicity of irrigation waters due to the presence of zinc. Given the existence of datasets for background concentrations of zinc in Australian soils, a soil CCL limit has been set for zinc based on current regulatory guidelines (NSWEPA 1995a).

9.2.6 Nitrogen and phosphorus

Long-term trigger values (LTV) and short-term trigger values (STV) for nitrogen and phosphorus in irrigation water are presented in table 9.2.19. They are based on maintaining crop yield, preventing bioclogging of irrigation equipment and minimising off-site impacts. Concentrations in irrigation water should be less than the recommended trigger values.

Table 9.2.19 Agricultural irrigation water long-term trigger value (LTV) and short-term trigger value (STV) guidelines for nitrogen and phosphorus

Element	LTV in irrigation water (long-term — up to 100 yrs) (mg/L)	STV in irrigation water (short-term — up to 20 yrs) (mg/L)
Nitrogen	5	25–125 [a]
Phosphorus	0.05[b]	0.8–12 [a]

a Requires site-specific assessment
b To minimise bioclogging of irrigation equipment only

9.2.6.1 Methodology for development of guidelines

The concepts of long-term trigger value (LTV) and short-term trigger value (STV) developed for metals and metalloids have also been used to develop guidelines for phosphorus and nitrogen. The logic for setting guideline values for phosphorus and nitrogen in irrigation water is unique because of their cycling in the environment, environmental significance and the high percentages removed in harvestable portions of crops. In light of the environmental consequences of excessive nutrients in our environment, there is an imminent need for guidelines so irrigators can be environmentally responsible. Guidelines will help assessment of water quality as an overall management tool in developing nutrient budgets, not only for optimal production, but to minimise off-site effects of nitrogen and phosphorus (Parris 1998).

Nitrogen

In view of the potential for nitrogen to affect plant maturation, the LTV has been set at a concentration low enough to ensure no decrease in crop yields or quality due to excessive nitrogen concentrations during later flowering and fruiting stages (i.e. <5 mg N/L, DWAF 1996a, Ayers & Westcot 1985).

The STV for nitrogen has also been developed to ensure that groundwater and surface water nitrogen does not exceed guidelines for drinking water (NHMRC 1996). That is, total nitrogen applied to the soil in irrigation water should balance the nitrogen uptake of the harvestable portion of the crop plus the acceptable concentration in drinking water (23 mg/L nitrogen or 100 mg/L nitrate). Volatilisation, denitrification and soil immobilisation provide safety margins against nitrogen overloading (NSWEPA 1995b). Considering the range of

nitrogen concentrations removed in harvestable portions of crops (see table 9.2.20), the STV range quoted should be used as a guide only, and site-specific assessment for particular crops should be undertaken (see Section 9.2.6.2).

Phosphorus

Environmentally significant concentrations of phosphorus in water are generally considered to be greater than 0.05 mg/L (ANZECC 1992, Foy & Withers 1995). However, aquatic plant growth (including algae) is not dependent on phosphorus alone. If all other nutrients and conditions are not optimal (e.g. poor light, high turbidity, high grazing rates, poor attachment substrates), some systems will cope naturally with relatively high nutrient loads without excessive aquatic plant growth. From the viewpoint of bioclogging of irrigation equipment, the LTV has been set low enough to restrict algal growth (i.e. 0.05 mg P/L), assuming all other conditions for algal growth are adequate.

Major considerations in developing the interim site-specific STV were: the fertiliser value of phosphorus in water; phosphorus removal from irrigation sites through the harvestable portion of crops; other fertiliser inputs; and soil phosphorus sorption/retention capacities of soils. An inherent difficulty in setting an STV for phosphorus is the complexity and site specificity of the phosphorus reactions in soil. In order to minimise off-site environmental impacts of phosphorus while considering agronomic implications, it is recommended that the site-specific STV for phosphorus be refined in the future when additional information becomes available. Further research is required as there is currently limited data available to assess the movement, or potential movement, of phosphorus from soils into water bodies due to phosphorus inputs into soils through the use of fertilisers or irrigation water (Daniel et al. 1998, Kirkby et al. 1997, Nash & Murdoch 1997, Ritchie & Weaver 1993, Sharpley 1993, Stevens et al. 1999).

9.2.6.2 Nitrogen

It is recommended that the concentration of nitrogen in irrigation waters should be less than the following:

Long-term trigger value in irrigation water		5 mg/L
Short-term trigger value in irrigation water	**Short-term use**	25–125 mg/L[a]

a Requires site-specific assessment. See below.

Nitrogen (N) is an odourless gas. It constitutes about 78% of the earth's atmosphere and, fixed or combined, is also present in many mineral deposits. Nitrogen can exist as four forms in water: ammonia, ammonium, nitrite and nitrate. Ammonia (NH_3) and ammonium (NH_4^+) are reduced forms of inorganic nitrogen; the relative portion of each in water is governed by water pH and temperature. Nitrite (NO_2^-) is the inorganic intermediate, and nitrate (NO_3^-) the end product of the oxidation of organic nitrogen and ammonia. All these forms are interrelated by a series of reactions known collectively as the nitrogen cycle. In this Section, nitrogen refers to all inorganic forms of nitrogen present in water (ammonia, ammonium, nitrate and nitrite).

Effects on crop growth and off-site considerations

Nitrogen is an essential plant nutrient. Excess quantities of nitrogen can lead to leaching into ground and surface waters, altered plant morphology and stimulation of algal growth in

surface water. Nitrogen in irrigation water can also increase maintenance costs for clearing excessive vegetation growth in irrigation channels.

Nitrogen concentrations in water can be reported as total nitrogen or as nitrogen in the form that it is present in solution. Nitrogen is most commonly found or reported as organic-nitrogen, nitrate or ammonium (10 mg N/L = 45 mg NO_3^-/L or 13 mg NH_4^+/L). The most available forms to plants are nitrate or ammonium. Nitrate is the usual form found in natural waters (Ayers & Westcot 1985), while NH_4^+ is the principal form found in wastewater (DWAF 1996a). Ammonium is absorbed rapidly by soils. In contrast, nitrate is soluble, mobile and relatively stable, and is therefore more readily leached into groundwater. Because of its mobility, nitrate is the most important form of nitrogen in soils from an environmental aspect. Therefore, under the assumption that all nitrogen forms have the potential to be expressed as nitrate in soil, total nitrogen has been used for setting trigger values.

Nitrate also poses a threat to animal and human health in drinking water, and plays an active role in eutrophication (NSWEPA 1995b). No health-based guideline for drinking water has been set for ammonia (NHMRC & ARMCANZ 1996). However, high nitrate concentrations in drinking water are potentially toxic. A limit of 50 mg NO_3^-/L (11.3 mg N/L) has been adopted for potable water for infants under 3 months old, and 100 mg NO_3^-/L for those over 3 months old (NHMRC & ARMCANZ 1996). Health effects due to excessive nitrogen in water supplies include methaemoglobinemia and cancer (Follett 1989). Methaemoglobinemia occurs when nitrate is converted to nitrite in infants, where the stomach acidity can be around pH 4. Absorbed nitrite can combine with haemoglobin to form methaemoglobin, resulting in a decrease in the oxygen-carrying capacity of the blood; this problem does not arise in adults (WHO 1984). However, given the re-examination of infantile methaemoglobinemia by Avery (1999), 100 mg NO_3^-/L would probably not increase the health risk to infants.

Nitrate may also be converted to suspected carcinogenic nitrosamines in the human digestive tract (Bouwer 1990). Nitrites should be kept below 3 mg/L based on health considerations (NHMRC & ARMCANZ 1996).

Plants generally have a high nitrogen demand during the early growth stages. However, excessive concentrations during the later flowering and fruiting stages may cause yield losses. Sensitive crops, which can show some of the effects outlined above at concentrations >5 mg N/L include: apricots, grapes, sugar-beets and cotton, but there are probably others (Ayers & Westcot 1985). Most crop yields are generally unaffected until nitrogen concentrations in irrigation water exceed 30 mg/L (Ayers & Westcot 1985).

If nitrogen is the growth limiting nutrient for algae, irrigation water with 0.1–1.6 mg N/L or greater (ANZECC 1992) could lead to increased aquatic plant or micro-organism growth, leading to clogging of irrigation lines and openings (Ayers & Westcot 1985). However, because the prokaryotic blue-green algae (cyanobacteria) have the ability to fix atmospheric nitrogen, it has been considered inappropriate to set a guideline value on eutrophication potential related to nitrogen concentrations in irrigation water (ANZECC 1992). Moreover, phosphorus is considered the limiting nutrient for algae growth in many freshwater systems (Schmitz 1996).

In view of the potential for nitrogen to influence plant maturation, the LTV has been set at a concentration low enough to ensure no decrease in crop yields or quality due to excessive nitrogen concentrations during later flowering and fruiting stages (i.e. <5 mg N/L, DWAF 1996a, Ayers & Westcot 1985).

The STV range has been based on annual crop nitrogen usage and export (table 9.2.20) and to minimise the risk of ground and surface water nitrogen concentrations exceeding 23 mg N/L (i.e. generally fit for human consumption; see discussion above). The STV should be considered on a site-specific basis relative to: crop uptake; crop sensitivity to excess nitrogen concentrations; irrigation load; removal of nitrogen from the irrigated site in harvestable portions of crops; volatilisation/denitrification losses; and fertiliser nitrogen applied. An example calculation for assessing site specific use is outlined below. These calculations do not consider the concentration of soil nitrogen through plant evapotranspiration and soil leaching, or dilution on entering water bodies. These two parameters have been excluded for three reasons: 1) simplicity, 2) limited data are available at this stage to accurately assess these mechanisms, and 3) these mechanisms counterbalance each other and the net effect could be insignificant. Recent modelling suggests that groundwater contamination by nitrate can be limited with good irrigation management and selection of appropriate crops (Snow et al. 1999, Salameh Al Jamal et al. 1997, Bjorneberg et al. 1998). However, local site-specific information should be used and each case assessed individually, as nitrogen uptake and removal from irrigation sites varies considerably with the type of crop grown (see table 9.2.20).

No CCL has been determined because nitrogen is a major plant nutrient.

Calculating site-specific short-term trigger values for nitrogen

$$STV_N = N_{es} + N_{removed} + N_{gasloss} \quad (9.32)$$

where:

STV_N = short-term trigger value for nitrogen (N) in irrigation water (mg/L)

N_{es} = environmentally significant N concentration i.e. >23 mg N/L potentially toxic to humans (11 mg N/L for infants < 3 months old; questionable, see Avery, 1999) in drinking water.

$N_{removed}$ = nitrogen removed from irrigation water in harvestable portion of the plant (mg/L)

$N_{gasloss}$ = gaseous losses through volatilisation and denitrification. This figure could vary from 0 to 80% of N applied depending on the forms of N present in irrigation water and environmental considerations discussed below. If an estimate is not available it is recommend that a value of 0 be used which will provide a safety margin in most cases.

For calculation of $N_{removed}$:

$$N_{removed} = \frac{N_{harv} - N_{fert}}{I_w \times 10} \quad (9.33)$$

where:

$N_{removed}$ = net nitrogen removed from irrigation water through harvestable portion of the plant (mg/L)

N_{harv} = nitrogen removed in harvestable portion of the crop (kg/ha). Calculated by multiplying the mean N concentration in the crop to be grown (kg/Mg; table 9.2.21) by the expected yield (Mg/ha; site-specific data).

N_{fert} = nitrogen applied in fertilisers (kg/ha)

Note: Plant available soil N concentrations from an appropriate soil test should be included in N_{fert}. However, these calculations consider only N removed from the harvestable portion of the plant, and including soil N may lead to insufficient N applied to supply the total plant demands in some instances. This potential shortfall is prevented as the N_{es} can supply up to 230 kg N/ha, sufficient N for most crops (table 9.2.20). The model also assumes that the non-harvested portion of crops will be returned to the soil and N in this portion contributes to the following crop's N demands.

I_w = irrigation water height (m)

For example, if 100 kg/ha of N were applied as a fertiliser and a cabbage crop grown using 1.00 m of irrigation water:

$$N_{removed} = \frac{(3.4 \times 50) - 100}{1 \times 10}$$

= 70 mg/L

For calculation of $N_{gasloss}$:

$$N_{gasloss} = (N_{es} + N_{removed}) \times N_{dv} \quad (9.34)$$

where:

$N_{gasloss}$ = amount of N loss through denitrification and volatilisation

N_{dv} = estimated loss of N through denitrification and volatilisation (% of total applied)

N_{es} and $N_{removed}$ are defined above.

For example if N_{dv} was estimated to be 5%:

$N_{gasloss}$ = (23 + 7.0) x 0.05

= 1.5 mg/L

From the above examples:

STV_N = 23 + 7.0 + 1.5

= 31.5 mg/L

Assumptions for calculating the STV range

Irrigation height = 1.00 m

N_{es} = 23 mg/L

$N_{removed}$ = minimum (20 kg/ha) and maximum (1015 kg/ha) N removal by crops listed in table 9.2.20 (excludes stubble crops) with no N added in fertilisers.

$N_{gasloss}$ = nil

The STV range recommended for nitrogen in irrigation water (25–125 mg/L, based on interactions between groundwater protection and crop usage) is a broader range than that quoted by DWAF (1996a) which ranged from 5–30 mg/L. However, using the median nitrogen removed (94 kg/ha) with harvestable portions (table 9.2.20), the STV = 32 mg N/L. The DWAF (1996) guidelines do not allow for site-specific assessment as is recommended above. Incorporation of a site-specific assessment has allowed high concentrations of nitrogen in irrigation water in some situations while still minimising off-site impacts.

Table 9.2.20 Nitrogen and phosphorus removal (kg/ha/crop) with harvestable portions of crops from specific locations

Crop	Area of NSW (see reference 4)	Harvestable portion (Mg/ha)	Nitrogen (kg/ha)	Phosphorus (kg/ha)	Reference[a]
Cabbage		50	147	24	1
Carrots		44	100	14	1
Cauliflower		50	119	23	1
Celery		190	308	79	1
Cucumber		18	28	5	1
Green Beans		4.5	160	4	1
Lettuce		50	100	18	1
Potato		40	132	15	1
Sweet Potato		24	59	14	1
Tomato (processing)		57	79	33	1
Tomato (table)		194	361	84	1
Bean, dwarf		15	38	6	2
Broccoli		20	90	13	2
Brussels sprouts		25	163	21	2
Carrot		80	104	28	2
Cauliflower		40	112	18	2
Celery, rooted		50	125	33	2
Chinese cabbage		70	105	28	2
Cucumber, pickl		70	105	21	2
Florence fennel		40	80	12	2
Iceberg		60	78	15	2
Kale		20	120	16	2
Kohlrabi		45	126	20	2
Leek		55	138	19	2
Lettuce, head		50	90	15	2
Onion		60	108	21	2
Radicchio		25	63	10	2
Radish, small		30	60	9	2
Red beet		60	168	30	2
Red cabbage		50	110	18	2
Savoy cabbage		40	140	20	2
Spinach		30	108	15	2
White cabbage		80	160	26	2
Potatoes		31.7	105	12	3
Lettuce		25.4	51	9	3
Carrots		35.4	80	11	3
Tomatoes (glass house)		51.3	95	22	3
Tomatoes (field)		38	53	22	3
Celery		95.8	155	40	3
Cauliflowers		38	90	17	3
Cucumbers		37.6	58	10	3
Beetroot		17.7	50	9	3
Chinese cabbage		17.5	26	7	3
Onions		44	79	15	3
Barley	North West	1.7	31	7	4
	Central West	1.5	27	6	4
	South Riverina	1.7	31	7	4
Canola	Central West	1.5	69	11	4
	South West slopes	1.5	69	11	4
Faba beans	North West	1.2	49	6	4
	Riverina	2.3	94	12	4

Table 9.2.20 continued

Crop	Area of NSW (see reference 4)	Harvestable portion (Mg/ha)	Nitrogen (kg/ha)	Phosphorus (kg/ha)	Reference[a]
Grain Sorghum	North West	2.5	53	8	4
	Central West	2.5	53	8	4
	Riverina	2.8	59	8	4
Lupins	Central West	1.4	70	7	4
	South West	1.3	65	7	4
Maize	North West	5.8	93	17	4
	Central West	5.6	90	17	4
	Riverina	7.0	112	21	4
	Coastal	7.0	112	21	4
Oats	North West	1.1	19	4	4
	Central West	1.4	24	6	4
	Riverina	1.6	27	6	4
	Tablelands	1.1	19	4	4
Field Pea	Statewide	1.0	40	2	4
Soybean	North West	1.8	119	11	4
	Riverina	2.2	145	13	4
Summer grain (legumes cowpeas, mungbeans, pigeon pea)		1.0	40	2	4
Sunflower	North West	1.2	62	7	4
	Riverina	1.7	88	10	4
Triticale	Central West	2.3	46	9	4
	South West	2.1	42	8	4
Wheat	North West	1.7	37	7	4
	Central West	1.5	33	6	4
	South Riverina	1.9	42	8	4
FORAGE CROPS:					
Forage millet	North West	6.0	102	12	4
	Riverina	5.0	85	10	4
	Coast	9.0	153	18	4
Forage sorghum	North West	7.0	126	21	4
	Riverina	6.0	108	18	4
	Coast	10.0	180	30	4
Maize	North West	12.0	132	24	4
	Riverina Coast	13.0	143	26	4
Summer grain legumes	North	3.0	51	12	4
Winter cereals	Statewide	5.0	75	15	4
Winter grain legumes	Statewide	4.0	108	12	4
STUBBLES FOR HAY:					
Wheat straw	North West	1.7	9	2	4
	Central West	1.5	8	2	4
	South Riverina	1.9	10	2	4
Barley straw	North West Central	1.7	9	2	4
	West South	1.5	8	2	4
	Riverina	1.7	9	2	4
Oat straw	North West	1.1	8	1	4
	Central West	1.4	10	1	4
	South Riverina	1.6	11	2	4
	Tablelands	1.1	8	1	4
Lupin straw	Statewide	nr			4
Pea straw	Statewide	0.5	6	1	4
Triticale	Central West	2.3	12	2	4
	South West	2.1	11	2	4

Table 9.2.20 continued

Crop	Area of NSW (see reference 4)	Harvestable portion (Mg/ha)	Nitrogen (kg/ha)	Phosphorus (kg/ha)	Reference[a]
Grain sorghum	North West	3.0	36	6	4
	Central West	3.0	36	6	4
	Riverina	3.5	42	7	4
Maize	North West	7.0	63	21	4
	Central West	7.0	63	21	4
	Riverina	9.0	81	27	4
	Coastal	9.0	81	27	4
Soybean	North West	0.9	7	1	4
	Riverina	1.1	9	1	4
PASTURES FOR WHOLE OF NSW:	Active growth period	Yield (Mg/ha)			
Kikuyu	Sep – Mar	30.0	780	90	4
Phalaris	Mar – Nov	9.0	99	27	4
Perennial ryegrass	Mar – Nov	6.0	210	18	4
Fescue	Sep – May	11	264	44	4
Lucerne	All year	29	1015	116	4
White clover	Sep – Feb	20.0	740	80	4

a References: 1 Sceswell & Huett 1998 (NSW, Australian data set); 2 Fink et al. 1999 (European data set); 3 *Horticultural development* 1995 (Northern Adelaide Plains, South Australian data set; calculated predominantly from nutrient concentrations in Scesswell & Huett 1998); 4 NSW feedlot manual 1995 (NSW, Australian data set)

Nitrogen gaseous losses

Nitrogen losses are generally through either denitrification (microbial conversion of NO_3^- to N_2 or N_2O) or volatilisation (chemically NH_3 (aqueous) is converted to NH_3 (gaseous) under favourable conditions).

Monnett et al. (1995) found that nitrogen removal via denitrification from spray irrigation of reclaimed water fluctuated due to the alternating aerobic and anaerobic (anoxic) conditions caused by irrigation frequency. Gaseous losses of nitrogen averaged 5.3 and 26.2% of applied nitrogen at the 12 and 25 mm per week loading rates, respectively. Monnett et al. (1995) summarised that the denitrifying capacity of the soils was limited by both nitrogen and carbon, and that maintaining reclaimed water in the upper, more microbially active, part of the soil column through split applications, was important to nitrogen removal via denitrification.

Denitrification is enhanced by anaerobic conditions and greater nitrate concentrations in the more microbially active topsoil (Monnett et al. 1995). If forms of nitrogen are transformed during water storage or irrigation to nitrate, this form is more readily available to the plant, but also readily undergoes denitrification to a gaseous form that is lost from the plant/soil system.

Smith et al. (1996) found losses of ammonia through volatilisation following reclaimed water irrigation of pasture at Wagga Wagga, New South Wales. Ammonia flux density was strongly related to evaporation; that is, when the reclaimed water evaporated, ammonia was lost to the atmosphere. Under high evaporative conditions, a maximum of 24% of the ammoniacal-N in the reclaimed water was lost by volatilisation within 2 days of application. Growing vegetables under commercial conditions near Melbourne, Smith et al. (1983) showed that during irrigation with reclaimed water, 38–82% of the ammonia was lost by volatilisation during storage and, in addition, 25% of the remaining ammonia was lost during irrigation and from the soil's surface. The major factors which influence volatilisation of ammonia are wind speed, soil/air temperature, and pH (Freney et al. 1983, Smith et al. 1996), as they increase (pH>7) volatilisation increases.

Table 9.2.21 Mean nutrient concentrations in harvestable portions of crops[a]

Crop species	Crop moisture (%)	Mean nutrient removed N (kg/Mg FW)[b]	P (kg/Mg FW)[b]	Crop species	Crop moisture (%)	Mean nutrient removed N (kg/Mg FW)	P (kg/Mg FW)
FRUIT/BEVERAGES				HARVESTED GRAINS			
Apple	84	0.32	0.08	**Cereals:**			
Apricot	83	2.3	0.32	Barley	11	*	2.7
Avocado		1.3	0.17	Cereal rye	11	14	3.4
Babaco	94	2.1		Maize	10	13	2.3
Banana	70–80	2.2	0.52	Millet / Canary seed	11	20	3.3
Black currant	80	1.8	0.34	Oats	11	16	2.7
Blackberry	84	1.9	0.22	Rice (grain & hulls)	14	10.3	2.4
Blueberry	85	1.1	0.13	Sesame	5	34	7.2
Cantaloupe/melon	87	1.9	0.59	Sorghum	10	17	2.3
Carambola	91	1.2	0.17	Triticale	11	16	2.4
Casimiroa	80	0.14	0.2	Wheat	11	*	2.5
Cherry	80	1.5	0.21	**Grain legumes:**			
Citrus fruit		2.9	0.4	Chickpea	10	33	3.8
Coffee		46	3.4	Cowpea	10	39	6.9
Cranberry	88	0.5	0.1	Faba bean	10	38	3.6
Currants	82	2.2	0.48	Field pea	10	35	3.6
Custard apple		2.6	0.3	Lablab	11	36	10
Date	21	3.6	0.46	Lentil	10	37	3.3
Fig	83	2.2	0.28	Lupin (Sweet)	9	48	3.3
Gooseberry	87	1.3	0.35	Lupin (Albus)	9	57	3.6
Grape (table)	~80	1.3	0.27	Lupin (Sandplain)	8	51	3.8
Grape (wine berries)		1	0.26	Lupin (Yellow)	9	61	4.3
Grapefruit	89	1.1	0.21	Mung bean	9	41	7.7
Guava	83	1.2	0.26	Green Mung bean	9	42	7.2
Jackfruit				Black Mung bean	10	40	6
Kiwifruit	~84	1.5	0.21	Narbon bean	11	39	4.4
Lemon & Limes	87	1.9	0.15	Navy bean	10	39	4.5
Longan	72	1.6	0.06	Pigeon pea	10	31	7.6
Longanberry		2.8	0.24	Vetch (common)	10	42	4.2
Loquat				**Pasture legumes:**			
Lychee		2	0.4	Lucerne seed		60	6.8
Mandarin		1.6	0.16	Medic seed	10	64	6.8
Mango	90	6.5	0.75	Serradella	10		4.9
Mangosteen	85	0.8	0.2	Oilseed crops:			
Mulberry	89	3.5	0.38	Canola / Rape	8.5	35	5.1
Nectarine	86	1.4	0.22	Cotton		22	6.6
Okra				Linola	W/w	31	4.4
Olive				Linseed / Flax	8.5	25	3.8
Orange	82	1.3	0.18	Mustard	8.5	33	8.1
Passionfruit		3.3	0.4	Peanut	10	36	3.2
Pawpaw		1.3	0.3	Safflower	8.5	29	3.1
Peach/Peacharine	86	1.2	0.2	Soybean	8.5	62	5.5
Pear	85	0.24	0.03	Sunflower	8.5	30	7.8
Pepino	93	1		Other crops:			
Persimmon		1	0.22	Hops	0	54	7.4
Pineapple		0.78	0.07	Lavender	30	4.5	0.45
Plum	86	1.5	0.19	Poppy	11.5	21	5.7
Prune		5.6	0.9	Pyrethrum		17	2.2
Quince				Tobacco		39	2.5
Rambutan							
Raspberry	84	1.8	0.29				
Roselle							
Stonefruit		1.2	0.12				
Strawberry	91	1.9	0.26				
Tangelo							
Tea (pluck leaves)		40	4				
Watermelon	94	1.5	0.25				

Crop species	Crop moisture (%)	Mean nutrient removed N (kg/Mg FW)[b]	P (kg/Mg FW)[b]
HARVESTED VEGETABLES			
Artichoke (edible)	84	4.3	0.77
Asparagus	94	2.2	0.41
Beans (all types)	91	3.8	0.39
Beetroot	91	2	0.3
Broccoli (all types)	90	5.4	0.82
Brussel sprouts	88	5.9	0.86
Cabbage (all types)	92	3.4	0.6
Capsicum	92	2.2	0.31
Carrot	89	1.6	0.4
Cassava	66	2.6	0.4
Cauliflower	91	3.1	0.59
Celery	95	1.3	0.29
Chicory (roots)	80	2.2	0.61
Chilli (red)	82	2.2	1.2
Chilli (green)	81	4.5	1.2
Chives	90	2.4	0.51
Chokos			
Cucumber	96	1.4	0.26
Egg plant	93	1.8	0.25
Fennell	94	1.5	0.26
Garlic (bulbs)	61	8.2	1.7
Gherkin	93	2.2	0.38
Ginger	89	1.8	0.4
Horseradish	76	7.2	0.8
Leek	91	2	0.19
Lettuce	96	1.9	0.37
Mushroom	91	6	0.8
Okra (edible portion)	90	3.1	0.6
Onion	89	1.9	0.42
Parsley	83	5.8	0.7
Parsnip	81	3.8	0.88
Peas	75	11.2	1.33
Peas (snow)	88	4.8	
Peppers	74	5.9	0.78
Potato (tubers)	80	3	0.42
Potato (sweet)	76	2.4	0.53
Pumpkin	90	2.1	0.56
Radish	93	3.5	0.31
Rhubarb	95	1.1	0.17
Silverbeet	93	2.9	0.42
Squash	92	3.9	0.34
Spinach	93	3.2	0.3
Swede			
Sweet corn (ears)		3.9	0.56
Tomato	94	1.6	0.33
Turnip	93	1.9	0.5
Zucchini	94	2.9	0.28

Crop species	Crop moisture (%)	Mean nutrient removed N (kg/Mg FW)	P (kg/Mg FW)
LIVESTOCK FODDER			
Hay:			
Lucerne (A)		28	2
Clover or medic (B)		22*	1.7
Clover/grass (B)		21*	2
Oaten (A)		13	1.6
Pasture (B)		18*	1.8
Sorghum (C)		16	
Chopped corn (C)		12	2.4
Silage:			
Grass (B)		24*	2.8
Pasture (B)		26*	2.8
Maize (B)		12*	1.9
Oaten (B)		20*	2.5
Sorghum (C)		15	
Unspecified (C)		13	
Grain:			
Barley (D)		16	2.7
Oats (E)		15	3.2
Sorghum (D)		15	3.2
Wheat (D)		28	3.2

* Expressed on an oven-dry basis

Crop Species	Crop burnt (%)	Mean nutrient removed N (kg/Mg FW)	P (kg/Mg FW)
SUGAR CANE			
District:			
Mossman-Gordonvale	10	0.75	0.1
Babinda-Tully	12	0.75	0.11
Herbert	5	0.67	0.11
Burdekin	91	1.11	0.24
Central	17	0.9	0.15
Bundaberg	32	0.88	0.16
Maryborough–Rocky Point	59	1.15	0.17
Queensland (average of districts above)	32	0.89	0.15

a DL Reuter pers comm; interim data from project 5.4D of the National Land and Water Resources Audit currently in progress, 2000

b FW = fresh weight

9.2.6.3 Phosphorus

It is recommended that the concentration of phosphorus in irrigation waters should be less than the following:

Long-term trigger value in irrigation water (To minimise bioclogging of irrigation equipment only)	0.05 mg/L
Short-term trigger value in irrigation water	0.8–12 mg/L[a]

a Requires site-specific assessment. See below.

Phosphorus exists as three allotropic forms: white, black and red. The above chemical and physical data refer to the white form. Phosphorus does not occur free in nature and is usually found in the form of phosphates in minerals, which are more soluble than the pure form.

Effects on crop growth and off-site considerations

Phosphorus is a major nutrient required for plant growth. It is usually present in irrigation water in two forms: dissolved inorganic phosphate ions, or colloidal phosphate (bound with solid minerals and/or organics). Dissolved inorganic phosphate ions (predominantly orthophosphate) are immediately bioavailable. Colloidal phosphates may contain phosphorus which is potentially bioavailable through desorption and decomposition, or phosphorus which is so strongly bound that it not bioavailable in the short to medium term. The form of phosphate is dependent on water or soil pH. When phosphorus is added to soil it is usually strongly sorbed. Soils that sorb phosphorus strongly are all high in iron or aluminium (Barrow 1989). The total amount of phosphorus that a soil can sorb out of solution can be determined from P sorption curves. The reserve of phosphorus that the soil can release back into the soil solution is buffered by the soil.

Excessive phosphorus in irrigation water is not a direct nutritional problem to plants (Papadopoulos 1993). However, phosphorus is often the limiting nutrient preventing rapid growth of many microorganisms (e.g. algal blooms; Schmitz 1996). If all other conditions are ideal for microbial growth and phosphorus is the limiting nutrient, increased concentrations of phosphorus in irrigation water (>0.05 mg/L, Foy & Withers 1995) could lead to enhanced algal growth, causing blocking of irrigation filters, pipes and outlets when using certain irrigation methods. In some crops there is also the potential for algae contamination of produce. More favourable environmental conditions (i.e. light and warmth) in water storage facilities also have the potential to increase algal growth (Whitton 1973) if phosphorus is not limited.

Environmentally significant concentrations of phosphorus (i.e. concentrations which could cause algal blooms in water bodies) may be transported in dissolved or particulate forms (Kirkby et al. 1997, Nash & Murdoch 1997, Sharpley 1993, Stevens et al. 1999). The availability of phosphorus to be taken up by algae varies depending on the form of the phosphorus in solution. However, for these guidelines it is assumed that all phosphorus (in the long term) is potentially available and guideline values have been set using total phosphorus concentrations.

The LTV for phosphorus has been set to minimise the risk of algal blooms developing in storage facilities, and to reduce the likelihood of biofouling in irrigation equipment. This value should not be seen as a default value for phosphorus in irrigation waters if biofouling of equipment is not a potential issue. An interim STV range for phosphorus has been set, as

there is currently insufficient data available to allow accurate site-specific assessments to be calculated in all cases. It is recommended that as information becomes available further development of a site-specific STV for phosphorus be seen as a priority for future guidelines.

As phosphorus is a major plant nutrient it is inappropriate to set a CCL limit for this element.

Calculating interim site-specific short-term trigger value for phosphorus

To date, a guideline value for phosphorus concentration in irrigation water has not been set (ANZECC 1992, DWAF 1996a). In the wake of recent blue-green algal blooms in Australia and the rapidly expanding body of literature which identifies diffuse agricultural sources of phosphorus responsible for phosphorus loading in water bodies (Correll 1998, Daniel et al. 1998, Dils et al. 1999, Edwards & Withers 1998, Haygarth & Jarvis 1999, Stevens et al. 1999, Van der Molen et al. 1998, Rayment & Hamilton 1997), an attempt must be made to set a guideline value for phosphorus in irrigation water. This guideline should be developed to restrict environmentally significant concentrations of phosphorus (i.e. concentrations which could cause algal blooms) moving in to water bodies (State Government of Victoria 1995).

In developing a model for calculating guideline values for phosphorus, the major sinks of phosphorus in the soil environment, and the variable nature of its reactions in soils must be considered (Holford 1997). Yet, the model should be kept as simple as possible. To minimise off-site impacts, the model must consider phosphorus removal from irrigated soils through the harvestable portion of crops, soil phosphorus sorption/retention capacities of soils and other phosphorus fertiliser inputs in to the soil.

Such a model should also consider soil colloidal phosphorus, preferential macropore flow and surface fluxes of phosphorus. However, there are limited data presently available to quantify these fluxes of phosphorus easily (Ritchie & Weaver 1993, Sharpley 1993, Stevens et al. 1999, Kirkby et al. 1997, Nash & Murdoch 1997). Therefore, these phosphorus pools have been excluded from the interim model described below.

If soil buffering capacities, or sorption capacities, are used in such a model certain assumptions will be required. Soil solution or soil extractant phosphorus concentrations are assumed to be related to the phosphorus movement through or over soils into water bodies. However, data relating soil solution or soil extractant phosphorus concentrations to phosphorus movement through or over soils are limited (Daniel et al. 1998, Dils et al. 1999, Edwards & Withers 1998, Ulen 1998). Many of the limitations above are areas that require further research, focusing on achieving a balance between plant availability of phosphorus in soil and restriction of phosphorus leaching/moving into waterways.

The expanding wastewater reuse industry, where there are often high phosphorus loadings, is developing guidelines for water reuse that balance plant nutrient demands with nutrients applied through irrigation. This is an attempt to reduce off-site impacts of excessive applications of nutrients (*NSW feedlot manual* 1995). This industry is now also recognising that the phosphorus sorption capacities of the soil (Hu 1999) need to be considered.

Proposed changes to guideline values for other inorganic contaminants in irrigation water include two guideline values, the LTV and STV. From the viewpoint of bioclogging of irrigation equipment (e.g. filters and drippers), or decreases in product quality due to algal contamination on some crops, it is recommended that the LTV for phosphorus be low enough to restrict algal growth (i.e. 0.05 mg P/L), assuming all other conditions for algal growth are adequate. It is recommended that the STV be low enough to prevent phosphorus in irrigation water overloading soil with phosphorus and allowing environmentally significant concentrations of phosphorus to move from soils into water bodies. In some cases, for the

benefit of the environment, this may mean yield reductions due to insufficient phosphorus supplies to meet plant demands. This is an area requiring discussion in the future.

Below is an interim model for calculating site-specific STVs for phosphorus. This model attempts to balance phosphorus inputs and output as a means of restricting excesses entering water bodies (Daniel et al. 1998). However, good irrigation management should also be adopted to restrict water movement and soil erosion (Daniel et al. 1998).

$$STV_P = P_{es} + P_{sorb} + P_{removed} \tag{9.35}$$

where:

STV_P = phosphorus in irrigation water (mg/L)

P_{es} = environmentally significant phosphorus concentration, i.e. algal blooms occur >0.05 mg P/L

P_{sorb} = phosphorus in irrigation water sorbed by soil (mg/L)

$P_{removed}$ = phosphorus removed from irrigation water in harvestable portion of the plant (mg/L)

For calculation of P_{sorb}:

$$P_{sorb} = \frac{\left[\frac{\left(\frac{Depth \times BD \times P_{ssc}}{100}\right) - P_{fert}}{I_w \times 10}\right]}{Years} \tag{9.36}$$

where:

P_{sorb} = total P sorbed from water by soil (mg/L)

Note: Phosphorus sorption capacity of soils could change with time through the slow irreversible absorption of phosphorus (Barrow 1974). A continual (annual) assessment of soil P sorption capacity is recommended.

Depth = soil depth (m)

BD = soil bulk density (kg/m^3)

P_{ssc} = phosphorus soil sorption capacity (mg/kg) with 50 µg P/L in solution at equilibrium. Sorption capacity should be representative of the soil depth used above.

I_w = irrigation water height (m)

P_{fert} = phosphorus input from fertiliser (kg/ha)

Years = years water will be applied (i.e. 20 years assumed for STV)

Note: P_{ssc} should be calculated from a P sorption curve measured as described by Rayment and Higginson (1992, Method 9J1). An example is given below (fig 9.2.6). The P_{ssc} should be taken when the extractant P concentration is 50 µg P/L (i.e. the P_{es} value). Ideally this value should be included within the points determined by the buffer curve. From figure 9.2.6 if x = 50, y (mg P sorbed by soil/kg soil) = 57. The concentrations of 50 µg P/L in the extractant solution will generally be overprotective as this is assumed to be an estimate of the

phosphorus concentration in soil solution. On leaching or surface flow this concentration may be diluted through rainfall, or diluted by entering a receiving water body with lower phosphorus concentrations. Site-specific soil sorption tests are required as soil sorption of P is dependent on soil type and can differ in orders of magnitude between soil types (Singh & Gilkes 1991, Sen Tran et al. 1988).

For example, if the soil depth was 0.15 m, the soil bulk density 1300 kg/m^3, P_{ssc} calculated to be 57 mg P/kg soil, 15 kg P/ha was applied as fertiliser, this type of cropping was expected to last 20 years, and the annual irrigation water applied was 1.00 m:

$$P_{sorb} = \frac{\left[\frac{\left(\frac{0.15\times1300\times57}{100}\right)-15}{1\times10}\right]}{20}$$

$$= \quad 0.48\text{mg/L}$$

For calculation of $P_{removed}$:

$$P_{removed} = \frac{P_{harv}}{I_w \times 10} \tag{9.37}$$

where:

$P_{removed}$ = phosphorus removed from irrigation water through harvestable portion of the plant (mg/L)

P_{harv} = phosphorus removed in harvestable portion of crop (kg/ha). Calculated by multiplying the mean P concentration in the particular crop to be grown (kg/Mg; table 9.2.21) by the expected yield (Mg/ha; site-specific data)

I_w = irrigation water height (m)

For example, if cabbage were grown with 1.00 m of irrigation water:

$$P_{removed} = \frac{50\times0.6}{1\times10}$$

$$= \quad 3.0 \text{ mg/L}$$

Using the above assumptions, the STV from an environmental perspective would be as follows:

$$STV_P = 0.05 + 0.48 + 3.0 = 3.5 \text{ mg/L}$$

The model assumes that the non-harvested portion of crops will be returned to the soil and phosphorus in this portion contributes to the following crop's phosphorus demands.

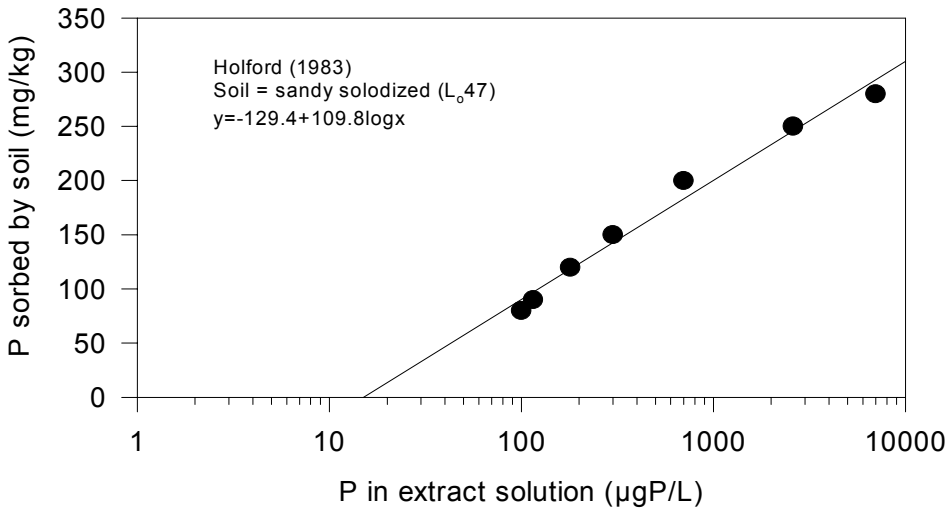

Figure 9.2.6 Soil phosphorus sorption curve (data modified from Holford 1983)

The STV range for P was calculated with the following assumptions:

Bulk density	=	1300 kg/m^3 soil, top soil depth = 0.15 m, irrigation height = 1.00 m
P_{ssc}	=	57 mg P/kg (typical more of a sandy soil and this value may be overprotective for soils with higher clay content and insufficient in other cases)
P_{fert}	=	0 kg/ha
Years of irrigation	=	20
P_{harv}	=	minimum (2 kg/ha) and maximum (116 kg/ha) P removal by crops listed in table 9.2.20 (excludes stubble crops)

Note:

1. In view of the range of values obtained for phosphorus sorption capacity and plant removal of phosphorus, it is recommended that site-specific data be assessed when assessing the STV for phosphorus.

2. Current research suggests that, in some soils, phosphorus can move overland or through some soils (preferential flow). This phosphorus will not be exposed to the soil matrix where sorption occurs. In this case, a large portion of the phosphorus sorbed (P_{sorb}) and phosphorus taken up by plants (P_{harv}) would not apply, and a more environmentally protective STV derived from the above equation would be 0.05 mg P/L. This would not be practical considering the nutritional requirements of plants for phosphorus, and current phosphorus fertilisation practices.

   It may also be appropriate to include a factor of soil texture or/and soil erodibility in to the model for determining the STV. However, there are currently insufficient data available to quantify such a factor.

9.2.7 Pesticides

Trigger values for pesticides in irrigation waters are listed in table 9.2.22. They consider likely adverse effects of herbicides on crop growth but do not consider potential impacts on aquatic ecosystems. They are based on relatively limited information and include only a sub-set of herbicides (and no other pesticides) that might be found in irrigation waters.

Table 9.2.22 Interim trigger value concentrations for a range of herbicides registered in Australia for use in or near waters[a]

Herbicide	Residue limits in irrigation water (mg/L)[b]	Hazard to crops from residue in water[c]	Crop injury threshold in irrigation water (mg/L)
Acrolein	0.1	+	Flood or furrow: beans 60, corn 60, cotton 80, soybeans 20, sugar-beets 60. Sprinkler: corn 60, soybeans 15, sugar-beets 15
AF 100	*	+	Beets (rutabaga) 3.5, corn 3.5
Amitrol	0.002	++	Lucerne 1600, beans 1200, carrots 1600, corn 3000, cotton 1600, grains sorghum 800
Aromatic solvents (Xylene)	*	+	Oats 2400, potatoes 1300, wheat 1200
Asulam	*	++	
Atrazine	*	++	
Bromazil	*	+++	
Chlorthiamid	*	++	
Copper sulfate	*	+	Apparently above concentrations used for weed control
2,4-D	*	++	Field beans 3.5–10, grapes 0.7–1.5, sugar-beets 1.0–10
Dicamba	*	++	Cotton 0.18
Dichlobenil	*	++	Lucerne 10, corn 10, soybeans 1.0, sugar-beets 1.0–10, corn 125, beans 5
Diquat	*	+	
Diuron	0.002	+++	
2,2-DPA (Dalapon)	0.004	++	Beets 7.0, corn 0.35
Fosamine	*	+++	
Fluometuron	*	++	Sugar-beets, alfalfa, tomatoes, squash 2.2
Glyphosate	*	+	
Hexazinone	*	+++	
Karbutilate	*	+++	
Molinate	*	++	
Paraquat	*	+	Corn 10, field beans 0.1, sugar-beets 1.0
Picloram	*	+++	
Propanil	*	++	Alfalfa 0.15, brome grass (eradicated) 0.15
Simazine	*	++	

Table 9.2.22 continued

Herbicide	Residue limits in irrigation water (mg/L)[b]	Hazard to crops from residue in water[c]	Crop injury threshold in irrigation water (mg/L)
2,4,5-T	*	++	Potatoes, alfalfa, garden peas, corn, sugar-beets, wheat, peaches, grapes, apples, tomatoes
TCA (Trichloroacetic Acid)	*	+++	0.5
Terbutryne	*	++	
Triclopyr	*	++	

a From ANZECC (1992). These should be regarded as interim trigger values only.

b Trigger values not set except as a general limit (0.1 mg/L) for specific herbicides in Tasmania and all herbicides in NSW.

c Hazard from residue at the expected maximum concentration: + = low, ++ = moderate, +++ = high.

9.2.7.1 Description

The presence of pesticide residues in waters has become an issue of public concern in recent years. In waters used for irrigation, issues concerning potentially harmful impacts include not only those to crops and pastures under irrigation, but also to the health of human consumers and to aquatic ecosystems receiving drainage waters. There is currently very limited scientific information on pesticide levels in irrigation waters and their likely impacts.

Pesticides are mainly organic compounds, or in some cases organo-metallic compounds, and are categorised according to their intended use; as insecticides (controlling insect pests), herbicides (controlling weeds), fungicides (control of fungal pests) and veterinary medicines (for animal health). Each category of pesticide is often grouped into classes of chemically similar compounds; for example, the organochlorine and organophosphate insecticides, and the phenoxy herbicides (Schofield & Simpson 1996).

Pesticides encompass a broad range of natural and synthetic compounds of widely differing chemical composition. All are carefully screened for health and environmental effects prior to registration for use. The use of pesticides for crop protection varies depending on the nature of the cropping or pasture system, crop value, pest pressure, environmental conditions and industry culture (Schofield & Simpson 1996).

Pesticide residues can sometimes be found in surface waters, as a result of: direct application (e.g. for weed control); careless use or disposal of pesticides and their containers; aerial drift and wind erosion; and transport in runoff waters (Hunter 1992, CCREM 1987, Schofield & Simpson 1996). Movement of pesticide residues which bind strongly to soil particles and are relatively insoluble in water occurs mainly through soil erosion processes. Runoff waters may also contain other pesticide residues in dissolved form. Leaching of some pesticide residues to groundwaters can occur, with the extent of leaching dependent on the chemical and physical properties of both the pesticide compound and the soil. Residues of several pesticides, notably the herbicide atrazine, have been found in surveys of some Australian groundwaters, but generally at very low concentrations (Keating et al. 1996, Schofield & Simpson 1996, HM Hunter, unpublished).

Many factors influence the persistence of pesticide residues in aquatic environments, including processes such as decomposition by sunlight, chemical transformation and microbial decomposition. Residues of some persistent organochlorines, such as DDT and dieldrin, can still be found in the environment although they were withdrawn from use, or have had restricted use in Australian agriculture for decades (Schofield & Simpson 1996).

9.2.7.2 Derivation of guidelines

While there is a comprehensive list of guideline values for pesticide residues in drinking water in Australia (NHMRC & ARMCANZ 1996), few guidelines exist for residues in waters used for irrigation purposes. In light of the limited information available, the ANZECC (1992) guideline values have been included here for use as interim guidelines. However, the guidelines consider only likely adverse effects of herbicides on crop growth and do not account for potential impacts on aquatic ecosystems. Moreover, the guidelines are based on relatively limited information and include only a sub-set of all herbicides (and no other pesticides) that potentially could be found in irrigation waters. The topic is further discussed in Section 9.2.10 regarding further research and information needs.

9.2.8 Radiological quality

Trigger values for the radiological quality of irrigation waters are given in table 9.2.23. The same trigger values also apply for livestock drinking water use.

Table 9.2.23 Trigger values for radiological contaminants in irrigation water[a]

Radionuclide	Trigger value
Radium 226	5 Bq/L
Radium 228	2 Bq/L
Uranium 238	0.2 Bq/L
Gross alpha	0.5 Bq/L
Gross beta (excluding K-40)	0.5 Bq/L

a These trigger values also apply for livestock drinking water.

9.2.8.1 Description

As groundwater is the major source of water for agriculture in Australia, the most significant radiological contaminants are those arising from naturally occurring radioactive species, particularly from natural uranium and thorium series. Radium-226, radium-228 and uranium-238 are the natural radionuclides which are often detectable in groundwater supplies. Surface water generally contains considerably lower concentrations of these radionuclides. Other long-lived natural radionuclides, for example thorium isotopes and lead-210, are normally not found in surface waters or groundwaters in significant quantities (UN 1993). The possibility of enhanced levels of natural radionuclides arising from activities such as processing of minerals containing uranium and thorium also needs to be considered in assessing the radiological quality of stock or irrigation waters.

Potassium-40 is a common radioactive constituent of groundwater. However, this radionuclide occurs in a fixed ratio to stable potassium and is not considered a health risk because a constant level is maintained in the human body (UN 1993).

Levels of radionuclides from nuclear fallout (e.g. strontium-90 and caesium-137) have decreased substantially in the Australian environment and no longer are significant. They have still been detected in some Australian soils, however their concentrations are well below the levels of natural radionuclides (M Cooper, unpublished). It is also unlikely that radioactivity from medical or industrial use of isotopes will be potential contaminants in stock water but may be important in irrigation (due to the increased risk of contamination of surface water supplies).

9.2.8.2 Effect on human and animal health

The main risks to health due to radioactivity in water will arise from the transfer of radionuclides from irrigation or stock water to crop or animal products (such as grains, meat and milk) and their subsequent consumption. Cancer is the potential health concern for humans associated with exposure to natural radionuclides.

An important consideration is that the naturally occurring radionuclides representing the most significant radiological health risk, radium isotopes and uranium-238, are not taken up readily into animal tissues or organs. Moreover, these radionuclides do not concentrate in meat tissue or milk (International Atomic Energy Agency 1994, Brown & Simmonds 1995).

Radiologically significant natural radionuclides do not concentrate in plants and crops (with rare exceptions) and transfer factors in the human food chain are usually well below unity. It is not considered feasible that levels of radioactivity in stock drinking water or irrigation waters used on pastures would be a direct threat to the health of the animals (UN 1993, International Atomic Energy Agency 1994).

Internal radiation exposure is measured in terms of 'committed effective dose' which is the dose received over a lifetime following the intake of a radionuclide. The unit of dose is the sievert (Sv). The average annual radiation dose from natural sources in Australia is estimated to be about 2 mSv (Webb et al. 1999). National guidelines for drinking water quality in Australia were based upon an annual committed effective dose of 0.1 mSv. For an individual, this represents an annual additional risk of developing cancer of about 5×10^{-6}.

In applying these guidelines it should be noted that the gross alpha and beta recommendations are given to simplify screening measurements and monitoring procedures. Specific radionuclide analysis would only be appropriate if these values are exceeded.

A water supply should not be considered to be unsafe for irrigation or stock water if specific radionuclide levels are exceeded. In such cases, further assessment of the supply should be conducted, including possible alternatives. If all or most other water quality parameters are acceptable, it may be possible to accept higher radionuclide concentrations without jeopardising health risks.

9.2.8.3 Derivation of guideline values

Minimising human exposure to radiation where possible should be a major consideration in establishing guidelines for radiological water quality. An ideal approach may be to maintain the same set of radiological guidelines for stock water as apply for drinking water quality in Australia and New Zealand. However, in most cases this would be impractical. Given that the main source of potential contamination will be naturally occurring radioactivity, it would be sensible to derive guideline values based upon the same dose limit (0.1 mSv) as applies to drinking water but to take into account the low transfer factors for such radionuclides into the human food chain via the animal pathway.

This review follows the methodology outlined in the *Australian Drinking Water Quality Guidelines* (NHMRC & ARMCANZ 1996), but using an annual committed effective dose of 1 mSv instead of 0.1 mSv to calculate trigger values for specific radionuclides in irrigation and stock waters. Note that it is proposed that 1 mSv will also be used in the forthcoming revision of the Australian Drinking Water Quality Guidelines based on recent information from the International Commission on Radiological Protection (Malcolm Cooper, pers. comm.). Only key natural radionuclides have been considered. It should be noted that the trigger value for uranium-238 is based on chemical toxicity considerations rather than on radiological grounds.

No trigger values are presented for other natural nuclides, such as thorium isotopes, lead-210 or polonium-210, because they are rarely found in surface or groundwater in significant quantities.

In order to have a practical monitoring program, it would be appropriate to use gross radioactivity as a screening technique with a level established above which specific radionuclide analysis should be carried out. Gross alpha radioactivity will indicate the presence of radium-226 and uranium isotopes. Potassium-40 will be the most likely contributor to gross beta radioactivity, along with radium-228. The contribution of potassium-40 to the gross beta activity should be determined prior to further assessment being carried out.

Taking into account the recommended trigger value concentrations for specific radionuclides, it is recommended that screening values should be established with a gross alpha level of 0.5 becquerel per litre (Bq/L) and a gross beta concentration of 0.5 Bq/L, after discounting the contribution due to potassium-40.

9.2.9 General water uses

9.2.9.1 pH

To limit corrosion and fouling of pumping, irrigation and stock watering systems, pH should be maintained between 6 and 8.5 for groundwater systems and between 6 and 9 for surface water systems.

Description

Measurement of pH is made to assess the acidity or alkalinity of a particular water in terms of its hydrogen ion (H^+) activity where:

$$pH = - \log [H^+] \quad (9.38)$$

A unit change in pH corresponds to a logarithmic (10x) change in H^+ activity . The pH scale ranges between 0 and 14, with 7 considered neutral, values <7 acidic, and values >7 alkaline.

In itself, pH does not actually represent a water quality issue, but rather it can give an indication of the presence of a number of water quality related problems. The greatest hazard encountered with low or high levels of pH is the potential for deterioration as a result of corrosion or fouling. Elevated levels of pH (>8.3) can indicate the presence of bicarbonate, carbonate and sodium; these issues are addressed separately (see Sections 9.2.4.1 and 9.2.4.3).

Effect on agriculture

Besides corrosion and fouling of water infrastructure, high or low pH can give an indication of potentially adverse conditions which may affect soil and crop health. In the case of irrigation water, slight deviations from guideline values will not greatly affect soil which is generally well buffered and can withstand change. However with significant variations, soil may be affected resulting in an overall modification of soil pH.

Acidic irrigation water can result in the mobilisation of various ions in the upper soil profile, for example, metals such as aluminium and manganese, in concentrations large enough to be toxic to plant growth (Gill 1986). Alkaline irrigation water can affect plant growth when applied to soil by reducing the availability of trace elements and potentially causing nutrient imbalance (Slattery et al. 1999).

Derivation of trigger values

Guidelines for pH levels to minimise corrosion and fouling are provided for agricultural waters by Gill (1986). These indicate that a pH <5 could potentially be corrosive. Values between 5 and 6 should be regarded with caution and an overall pH of >6 should be maintained to limit the level of corrosion in a system. Because of the increased potential in groundwaters for encrustation and fouling (McLaughlan 1996), their recommended upper limit (pH <8.5) is slightly lower than for surface waters (pH <9).

9.2.9.2 Corrosion

Trigger values for assessing the corrosiveness of water are given in table 9.2.24.

Table 9.2.24 Corrosion potential of waters on metal surfaces as indicated by pH, hardness, Langelier index, Ryznar index and the log of chloride to carbonate ratio

Parameter	Value	Comments
pH	<5	High corrosion potential
	5 to 6	Likelihood of corrosion
	>6	Limited corrosion potential
Hardness	<60 mg/L CaCO3	Increased corrosion potential
Langelier Index	<-0.5	Increased corrosion potential
	-0.5 to 0.5	Limited corrosion potential
Ryznar Index	<6	Limited corrosion potential
	>7	Increased corrosion potential
Log of chloride to carbonate ratio	>2	Increased corrosion potential

Corrosion of pumping, irrigation and stock watering equipment is a common problem in many agricultural areas of Australia, particularly where groundwater sources are used. It often results in the deterioration of well and pumping equipment, pipelines, channels, sprinkler devices and storage tanks, leading to decreased or uneven water distribution. Corrosion can be based on chemical, physical or microbiological processes acting on metal surfaces in contact with water. Plastics and concrete may also deteriorate through processes similar to corrosion, if elevated levels of certain constituents are present.

Description

Most corrosion problems in relation to agriculture are generally associated with the use of groundwater rather than surface waters, due to differences in their chemical composition. Corrosive failure of pipes and groundwater wells may also occur from contact with certain soil types.

The extent and likelihood of corrosion depends on a number of parameters including water quality, flow rate, temperature, pressure and the types of materials which are in direct contact with water. These factors form the basis of a complex set of interactions which may lead to the corrosion of surfaces and fittings.

The economic cost of maintaining and replacing corroded equipment is often a significant component of overall farm expenditure, and as a result should be taken into account when considering agricultural water quality. The following Section outlines the main types of corrosion which can affect water pumping and distribution equipment through a number of different mechanisms (for further information see review of McLaughlan 1996).

Metal corrosion

Chemical processes

Metal corrosion is most commonly the result of electrochemical reactions based on the transfer of electrons through oxidation-reduction reactions. Electrons are generated at the anode (the site where oxidation and corrosion occurs) and are transferred through the metal to the cathode (the site where reduction occurs). An electric current is generated between the two points and transferred via dissolved ions present in the water to form a closed circuit. For the corrosion process to occur, there must be the formation of ions and release of electrons simultaneously and at an equivalent rate to the acceptance of electrons at the cathode (McLaughlan 1996).

The extent of metal corrosion can be influenced by other parameters including polarisation and external electrical currents. Polarisation is the retardation of electrochemical reactions due to the formation of a protective film (or scale) over the metal surface. This scale may be formed from corrosion products, or ions in solution which may precipitate out. External currents (e.g. currents produced from the grounding of electrical equipment such as pumps) can increase corrosion rates at the anode where it enters groundwater or adjacent soil.

Biological processes

Microorganisms can also increase the rate of corrosion through the formation of biofilms on the metal surface. Biocorrosion may then occur through electrochemical reactions within this micro-environment. These reactions generally do not occur in the water away from these surface sites.

Physical processes

Erosion of protective layers on the metal surface can lead to corrosion. Artificial coatings (e.g. precoated metal) or natural coatings (e.g. build-up of iron oxides and carbonates) can be removed as a result of particles in suspension impacting on a surface in combination with elevated flow rate. The critical level of particles above which corrosion will establish varies, depending on the individual flow situation.

Erosion can also occur through the formation and subsequent collapse of gas bubbles during groundwater pumping. Water entering the pump at low pressure vaporises, forming pockets which implode when subjected to high pressure on flowing through the pump. When this occurs against a solid surface, the localised pressure change can damage the metal surface or remove protective surface films, leaving a roughened surface which can then provide sites for further bubble formation (McLaughlan 1996).

Degradation of synthetic materials

With increasing use of synthetic material in smaller-scale agricultural systems, the structural degradation of materials such as plastics and PVC has become a significant issue. The mechanisms involved differ from the standard corrosion processes, with organic contaminants carried in groundwater being primarily responsible for degradation of these materials.

Penetration of synthetic materials by chemical compounds may alter their structural properties through swelling and softening, leading to potential failure. This has normally been associated with waters containing relatively high contaminant loads, which should not be present in a raw water source, but it may potentially be an important issue in regard to on-site reuse of wastewaters.

Concrete corrosion

Corrosion of concrete irrigation channels and pipelines occurs through the three mechanisms of leaching, ion exchange and expansion (Ayers & Westcot 1985), which may interact or act independently. Moreover, bacteria can cause biocorrosion in concrete through the breakdown of exposed surfaces in contact with water. This is often associated with the conversion of hydrogen sulfide gas to sulfuric acid by certain species of bacteria e.g. *Thiobacillus* sp (Tiller 1982).

Leaching occurs when lime in concrete is dissolved by water containing free carbon dioxide in the form of carbonic acid or by low salinity soft water (low carbonate hardness) (Ayers & Westcot 1985). Although this form of corrosion does not cause major damage to expansive areas of concrete, it can significantly affect jointing fixtures which may lead to structural weakness.

Alkaline cations (e.g. calcium, magnesium, potassium and ammonium) in irrigation water react with soluble compounds in cement through base-exchange reactions to produce exchange products. These may then be leached or remain in situ as non-binding components, reducing concrete strength.

Concrete compounds chemically react with components in groundwater (e.g. sulfate) and are replaced by new compounds which occupy a larger volume. This leads to swelling and internal stress, resulting in the potential breakdown of concrete structure.

Water quality parameters which influence corrosion

pH

Acidity is one of the important factors which influences the extent of corrosion in an irrigation system. Guidelines provided for agricultural waters by Gill (1986) indicate that a pH <5 could potentially be corrosive. Values between 5 and 6 should be regarded with caution, and a pH of >6 should be maintained to limit the level of corrosion in a system. NHRMC and ARMCANZ (1996) give a slightly more conservative pH limit of >6.5, based on studies of reticulation systems for potable water supply. Along with several other parameters, pH is used in the calculation of the Langelier Index, which provides an indication of the corrosion or scaling potential of a water.

One of the constraints in using pH as a corrosion indicator in groundwater is that it may be difficult to get an accurate measurement. There is often an increase in pH once water has come in contact with the atmosphere, which means that water bought to the surface or measured sometime later in a laboratory may not accurately reflect in situ levels. Using equipment designed to overcome these limitations, a study by the Australian Geological Survey Organisation of several bores in the Great Artesian Basin identified a strong inverse relationship between borewater pH and initial rates of corrosion (Larsen et al. 1996).

Hardness

The hardness or softness of water is based on the level of dissolved calcium and/or magnesium salts. This is normally expressed as a calcium carbonate equivalent (mg/L $CaCO_3$). Other cations (e.g. barium, iron, manganese and strontium) can also influence the level of hardness.

Two types of hardness have been identified, carbonate (temporary) and non-carbonate (permanent). This classification can be used to determine the potential for corrosion or fouling of pumping, irrigation and distribution equipment in waters. NHMRC and ARMCANZ (1996) define carbonate hardness as the total alkalinity expressed as calcium carbonate (where alkalinity is the sum of the carbonate, bicarbonate and hydroxide content),

and non-carbonate hardness as the difference between the total and carbonate hardness. Soft water has a tendency to be more corrosive than hard water (Awad 1989). It is recommended that waters be maintained at a hardness level of >60 mg/L ($CaCO_3$) to minimise corrosion (NHMRC & ARMCANZ 1996, EEC 1997).

Dissolved oxygen

Dissolved oxygen (DO) is the main oxidising agent which causes corrosion, with the tendency to corrode increasing with increasing DO concentration. The rate of corrosion in iron and steel increases with increased DO concentrations to a maximum and then decreases. A number of reasons for the decrease have been put forward, including the passivity behaviour of iron at high oxygen concentrations (Frese 1938, Streichner 1949).

The rate of oxygen transfer to the cathode is basically a function of temperature, time, flow rate and the presence of a scale (McLaughlan & Knight 1989). The complex interaction of these factors makes it difficult to determine a threshold value for corrosion based on DO concentration. It is recognised, however, that although elevated levels of DO can cause corrosion, low DO levels can also create environments where biocorrosion may occur.

Dissolved oxygen is not commonly used as a corrosion indicator in agricultural water due to the problems encountered in accurate sampling and analysis. Although special precautions can be taken to 'fix' the DO in the sample at the time of sampling, this is a costly option which, furthermore, does not give conclusive evidence of corrosion.

Carbon dioxide

Free or 'aggressive' carbon dioxide (CO_2) is defined as the amount of dissolved CO_2 in excess of that required to stabilise the bicarbonate ion present in water (Denaro 1991). This excess CO_2 combines with water to form carbonic acid, which can further dissociate to form hydrogen ions, according to the following reaction:

$$CO_2 + H_2O \leftrightarrow H_2CO_3 \leftrightarrow H^+ + HCO_3^- \qquad (9.39)$$

This gives an acidic solution which provides a suitable environment for metal corrosion.

Trigger values for CO_2 in water are hard to define in a general sense due to the complexity of interactive factors involved and the difficulty of accurate analysis. Crolet (1983) noted that to predict CO_2 corrosion, temperature, partial pressure of CO_2, fluid velocity and chemistry of the water must be considered. 'Aggressive' carbon dioxide can only be measured accurately at the water source, as levels are likely to decline due to degassing of the water sample during collection and transportation.

Corrosion of steel by CO_2 is often very localised in the form of pits, gutters or attached areas with abrupt changes from corroded to non-corroded areas (Crolet 1983, Denaro 1991).

Hydrogen sulfide

Hydrogen sulfide (H_2S) is a common constituent of many groundwaters, and forms as a result of the breakdown of organic matter or mineral release. Corrosion from hydrogen sulfide occurs in two different forms, sulfide stress cracking and hydrogen stress corrosion cracking (McLaughlan & Knight 1989).

Sulfide stress cracking is the result of brittle failure caused by tensile stress and corrosion by water and hydrogen sulfide. Hydrogen stress corrosion cracking occurs through cracking caused by a combination of tensile stress and a specific corrosive medium. Further details of the processes involved are provided by McLaughlan and Knight (1989), Treseder (1981) and Fontana (1986).

In certain environments, hydrogen sulphide can be converted to sulfuric acid, leading to the potential acidification of waters and possible corrosion of exposed surfaces and fittings in distribution systems.

Electrical conductivity

Electrical conductivity (EC) is a measure of the ability of water to conduct an electric current, which is dependent on the concentration of dissolved ions. In general, agricultural waters with high EC values are more corrosive than those with low EC values. This, however, is dependent on the types of ions present, as some are more corrosive than others (e.g. elevated levels of sodium in waters are more likely to cause corrosion than calcium). Electrical conductivity is not generally used as a specific indicator of corrosion because it represents the total ion content rather than types of ions present in water.

Corrosion indices

Langelier Saturation Index (LI)

Langelier (1946) developed a saturation index (LI) based on the tendency of a water to deposit or dissolve calcium carbonate. The index gives an estimate of its corrosion potential by indicating whether a protective film (or scale) may be formed on a metal surface, based on the reaction:

$$CaCO_3 + H^+ \leftrightarrow Ca^{2+} + HCO_3- \quad (9.40)$$

The saturation index is calculated as:

$$LI = pH - pH_s \quad (9.41)$$

where:

pH = measured pH of water;

pH_s= pH of the water if saturated with $CaCO_3$ at the measured calcium and alkalinity value. Values for pH_s can be found in texts detailing corrosion mechanisms (e.g. Kelly & Kemp 1975).

Thus, waters that have a negative LI are undersaturated, a value of 0 is saturated and a positive LI is supersaturated with respect to $CaCO_3$. For agricultural waters, Awad (1989) suggested that values ranging between -0.5 and 0.5 should not lead to corrosion or carbonate encrustation problems. Values below -0.5 indicated potential corrosivity, while values above 0.5 indicated the likelihood of excess encrustation.

Although the LI does not consider other parameters such as flow rate, organic content and the influence of other chemical compounds in water (e.g. phosphates and silicates) and is therefore limited in its accuracy, it is one of the few methods available.

An adaptation by Snoeyink and Jenkins (1980) measures pH_s in terms of calcium and bicarbonate values (replacing alkalinity). Rossum and Merrill (1983) found that this adaptation provided a more accurate approximation when compared with theoretical precipitation potential.

Ryznar Index

Ryznar (1944) developed a stability index (RI) which is used quite often as a corrosion predictor. It is based on the following empirical equation. He found that with RI values <6, a film of $CaCO_3$ was deposited, however, for values >7 there may not be a film.

$$RI = 2pH_s - pH \quad (9.42)$$

where:

pH_s= pH of the system if saturated with $CaCO_3$ at the measured calcium and alkalinity value;

pH = measured pH of water.

Ratio of chloride to carbonate

High chloride content in water has often been associated with corrosion, however this is not an accurate assumption. Although chloride is an active agent in the corrosion process, it is dependent on other water quality parameters such as pH, temperature and the presence of other dissolved ions.

Kelly and Kemp (1975) noted that a relationship existed between chloride content and the passivating ions bicarbonate and carbonate. A useful approximation of potential corrosiveness is based on the log of the ratio of chloride to carbonate concentrations, which assesses the level of corrosive agent (chloride) to the potential of scale formation (carbonate).

To calculate the ratio of chloride to carbonate, the pH, temperature, chloride content, alkalinity and conductivity of the irrigation water need to be known from laboratory analysis. The procedure for calculation is explained simply in Kelly and Kemp (1975).

When the ratio of log ($^{[Cl^-]}/_{[CO^{3-}]}$) is small, corrosion potential is considered to be low. However, as the chloride concentration increases relative to carbonate, and the log of the ratio ($^{[Cl^-]}/_{[CO^{3-}]}$) exceeds about 2, pump metals are likely to corrode (Kelly & Kemp 1975).

Control measures

The parameters described previously can only give a rough estimate of whether corrosion is likely to occur. Even if water quality meets the guideline values, corrosion still may occur. For further information on methods of analysis for corrosion the reader is referred to APHA, AWWA & WEF (1998). A brief description is provided below of the three main approaches to minimising the effect of corrosion in agricultural water systems.

Monitoring

Part of the maintenance of an agricultural water distribution system should be the keeping of adequate records covering issues like relevant water quality parameters, power usage, construction, maintenance and hydraulic details. The collection frequency of water quality data for an agricultural well need not be as stringent as for domestic water supply. Visual assessment of water quality for increased turbidity and suspended solids can be done at the time of pump start, as this may indicate an increase in corrosion. An analysis of water quality, taking into consideration the parameters and appropriate guidelines outlined above, should be conducted before construction of the distribution system. Monitoring should be undertaken on a regular basis.

Materials

When establishing a distribution system, the material chosen for construction is important, as this will influence the extent of corrosion in the system. Materials found to be the most resistant to corrosive waters include austenitic stainless steel (e.g. 316 stainless steel, 904 L stainless steel), zinc-free bronze for pumping equipment (with the option of coating with epoxy if needed) and synthetic materials such as plastics (for piping and fittings). Corrosion-resistant pumps and fittings are also available. Although corrosion-resistant materials are generally more expensive during initial establishment, long-term savings can occur as a result of decreased maintenance and replacement costs.

If possible, McLaughlan (1996) recommends the use of inert or corrosion-resistant thermoplastic, fibreglass or PVC wells with plastic or stainless steel screens. It is also recommended that sulfate-resistant cement is used in the upper casing areas where saline water may be intersected. Oxygen should also be limited within the system, as it is one of the primary influences on corrosion. Lining an already existing system with inert material is also an option. This method is particularly suited to joints and fittings, which are often the most vulnerable parts of water infrastructure. There are a number of different options available including abrading the surface, cementing and coating with coal tar products.

Chemical treatment

In some cases of corrosion, chemical treatment of water can be used. This will not overcome problems in flow rate or the distribution system itself, however it can be implemented in some instances in the short term. Awad (1989) recommends the use of lime or soda ash, which can make water less corrosive by increasing pH. The amount needed is based on the hardness or softness of water. It is recommended that alkalinity levels of 50–100 mg/L and calcium levels of 30–50 mg/L be maintained at normal temperatures to minimise corrosion.

Derivation of trigger values

A literature review was undertaken on corrosion in relation to water quality and agricultural issues. Limited research was found directly relating to agriculture, however information was available on groundwater extraction and corrosion, directly applicable to agricultural systems using groundwater resources.

The National Centre for Groundwater Management (University of Technology, Sydney) has recently released a number of publications based on extensive research, bringing together current issues on corrosion and groundwater wells. The research involved a critical literature review on corrosion mechanisms and data, and incorporated relevant information from the petroleum and water supply industries. These provide a valuable guide to corrosion-related issues and solutions in Australia, and have been used as a basis for several of the trigger values given in this review.

9.2.9.3 Fouling

Trigger values for assessing the fouling potential of waters are given in table 9.2.25.

Table 9.2.25 Fouling potential of waters as indicated by pH, hardness, Langelier index, Ryznar index and the log of chloride to carbonate ratio

Parameter	Value	Comments
pH	<7	Limited fouling potential
	7 to 8.5	Moderate fouling potential (groundwater)[a]
	>8.5	Increased fouling potential (groundwater)[b]
Hardness	>350 mg/L $CaCO_3$	Increased fouling potential
Langelier Index	>0.5	Increased fouling potential
	-0.5 to 0.5	Limited fouling potential
Ryznar Index	<6	Increased fouling potential
	>7	Limited fouling potential
Log of chloride to carbonate ratio	<2	Increased fouling potential

a For surface waters, pH range 7 to 9

b For surface waters, pH >9

Fouling of agricultural water systems can lead to decreased water quality and yield as a result of clogging, encrustation and scaling. All parts of the system can be affected including wells,

pumping equipment, pipes and sprinklers. The main causes of fouling in agricultural water distribution systems can be attributed to physical, chemical and biological properties of the water.

Types of fouling

The main causes of fouling in agricultural water distribution systems are associated with water quality. Physical, chemical and biological parameters can affect the type of fouling (table 9.2.26), which is also influenced to some extent by the materials used in construction. Of particular concern is the effect of fouling in localised (drip) irrigation systems, which deliver water to the crop at a low flow rate and have a tendency to become easily clogged.

Table 9.2.26 Principal causes of fouling in agricultural water distribution systems

Parameter	Description
Physical	Accumulation of sand, silt, clay and organic matter causing clogging
Chemical	Precipitation of chemical compounds (e.g. calcium carbonate, iron compounds) causing encrustation and scaling
Biological	Encrustation or scaling formed as a result of build-up of microbial populations or precipitation of chemical compounds forming a biofilm

Physical

Accumulation of particles within the distribution system can occur as a result of construction, poor design, weathering of the surrounding geological strata or transportation in the water source. Sand, silt, clay and organic matter are the most common particulates and these are generally carried in water in the form of suspended material at elevated flow rates. This is particularly the case in surface waters, as groundwater tends to contain limited organic matter and solids in suspension.

In groundwater wells, particulate matter tends to enter the system during construction or from natural weathering processes. Accumulation of this material can lead to a decrease in aquifer permeability and clogging of screens, resulting in a potential yield reduction.

Filtration is the most effective method of removal of particulate material from the water source. Water is normally passed through graded sand which removes organic matter, sand, silt and clay. Screening may also be used and is adequate for the removal of larger particles, however fine material may still pass through. Ayers and Wescot (1985) give recommended guidelines for concentrations of suspended solids to avoid clogging in localised (drip) irrigation systems. These are given below, based on the degree of restriction of use:

<50 mg/L	no restriction
50–100 mg/L	slight to moderate restriction
>100 mg/L	severe restriction

Chemical

Chemical precipitation of compounds can result from an excess of calcium or magnesium carbonates and sulfates, or from iron in the soluble ferrous state which is oxidised to the insoluble ferric form on exposure to oxygen (Ayers & Westcot 1985). It may also occur as a result of changes in temperature and pressure, or mixing of different quality waters.

Fouling caused by a build-up of precipitates is referred to as scaling, and is more commonly associated with groundwaters. Changes in water temperature and pressure can cause scaling when groundwater is pumped to the surface, through the degassing of carbon dioxide (CO_2).

This process alters the concentration of CO_2 in water, and may subsequently trigger the formation of precipitate scale. A detailed description of the chemical processes involved is provided in McLaughlan and Knight (1989).

Wells can often intersect several different groundwater chemistries and combining of these waters through parting or corrosion of the well, or screening of the well through different layers, can promote the accumulation of scale (McLaughlan 1996). The main chemical precipitates which can lead to fouling of agricultural and groundwater pumping systems are iron, manganese and carbonates. These will be briefly outlined in the following text.

Iron and manganese precipitation

Iron and manganese are common constituents in groundwater and may be present in surface waters depending on the surrounding catchment geology. They can occur in the divalent form as dissolved ions (Fe^{2+}, Mn^{2+}), as solids (FeS_2, $FeCO_3$), or in more oxidised forms (Fe^{3+}, Mn^{3+}, Mn^{4+}) which can form precipitates ($Fe(OH)_3$, MnO_2) (McLaughlan & Knight 1989).

The precipitation of iron and manganese is influenced by various water quality parameters including pH, redox potential (Eh), concentrations of dissolved CO_2, sulfur, organic matter and the presence of microorganisms. Generally, if pH is maintained in the range of 5–9 at a low Eh (0.1–0.2v), then iron will remain in solution (Hem 1970). Manganese, which is generally more stable, will also remain in solution under these conditions.

Elevated levels of dissolved CO_2, sulfur and organo-metal complexes can also lead to the formation of precipitates, however, trigger values are hard to define due to the complex interactions involved. A comprehensive description is provided in Hem (1970).

Microorganisms often play a substantial role in the formation of scale from iron and manganese precipitates and this issue is discussed in the Section on biofouling.

Carbonate compounds

Precipitation of carbonate compounds occurs in conjunction with the release of CO_2, as a mechanism to buffer water against significant changes in pH. It involves the following series of reactions (McLaughlan & Knight 1989), which are dependent on a number of factors including the temperature and ionic strength of water.

$$CO_2\,(g) \leftrightarrow H_2CO_3 \tag{9.43}$$

$$H_2CO_3 \leftrightarrow H^+ + HCO_3^- \tag{9.44}$$

$$HCO_3^- \leftrightarrow H^+ + CO_3^{2-} \tag{9.45}$$

$$CO_3^{2-} + Ca^{2+} \leftrightarrow CaCO^3 \tag{9.46}$$

Calcium carbonate is one of the most common carbonate compounds to cause fouling in agricultural water distribution systems. The potential for fouling of agricultural waters can be predicted using Langelier's Index. Positive values of the index indicate a tendency for precipitation of $CaCO_3$, while negative values indicate the potential for $CaCO_3$ to dissolve and for corrosion to occur.

Biological (biofouling)

Fouling as a result of microorganisms such as bacteria, algae, slimes and fungi can occur in groundwater wells, storage tanks, irrigation and pumping equipment. Microorganisms can accelerate the rate of the chemical reactions described in the previous Section or can cause clogging due to excessive growth.

Biofouling deposit formation involves the production of extracellular polymeric substances and the subsequent accumulation of inorganic elements and colloids (McLaughlan et al. 1993). A number of different mechanisms can result in different forms of biofouling including iron, sulfur, aluminium and organic based deposits. The most common type of deposit however, is usually a combination of physical, chemical and biological processes.

Important factors influencing the rate of biofouling are listed in table 9.2.27.

Table 9.2.27 Factors influencing the rate of biofouling[a]

Water quality	Bacterial activity	Particle availability	Biofilm shear forces
Dissolved ions	Nutrient availability	Level of particles in suspension	Flow rate
Precipitation mechanisms (e.g. CO_2 degassing, temperature, pH changes, oxidation)	Production of extracellular polymers	Volume of flow	Turbulence
		Aquifer composition	

a Adapted from McLaughlan 1996

Iron biofouling

Iron biofouling is one of the most common causes of deterioration of groundwater wells and pumping equipment. A number of species of iron bacteria are known to cause this form of biofouling including *Gallionella*, *Pseudomonads* and *Siderocapsaceae* (Cullimore 1992). A biofilm is produced consisting of bacteria, iron hydroxides and other inorganic precipitates trapped in an extracellular polymeric matrix (McLaughlan 1996) which, under the right conditions can form a thick encrustation leading to flow restriction and eventual blockage. Iron bacteria can be aerobic or anaerobic depending on the species, and generally function by oxidising iron for a number of different metabolic purposes (McLaughlan & Knight 1989).

Manganese is commonly associated with iron in groundwaters and can contribute to biofouling through similar depositional mechanisms. Precipitation of iron sulfides by sulfate bacteria can also contribute to biofouling. This is believed to be primarily undertaken by sulfate reducing bacteria which require anaerobic conditions and an organic substrate to produce a biofilm.

Aluminium biofouling

Aluminium biofouling is associated with the use of acidic waters and can cause deterioration of pumping and distribution systems. It usually occurs in the presence of bacteria and sulfate (e.g. in acid sulfate soil environments) and is sometimes related to iron biofouling. This form of microbial encrustation is not as common as iron biofouling (McLaughlan et al. 1993).

Organic biofouling

Organic biofouling requires high levels of available nutrients, organic matter and suitable environmental conditions for the growth and reproduction of microorganisms (in particular bacteria), and is becoming a significant problem with the increasing reuse of effluents in agriculture. It is associated in most cases with other forms of fouling including precipitation of chemical compounds and attachment of particulate matter (McLaughlan et al. 1993).

Water quality parameters that influence fouling

A number of water quality parameters can be used to indicate the potential for carbonate fouling by agricultural waters. In some cases, these are closely related to corrosion indicators and are based on the same reaction processes. Indices for biofouling have not yet been established.

Hardness

The level of hardness of a water can give an indication of the potential for fouling through the precipitation of calcium or magnesium carbonates. It is normally expressed as a calcium carbonate equivalent (mg/L $CaCO_3$). Soft water can be associated with corrosion while hard water can lead to encrustation and scaling of distribution systems (e.g. drip irrigation lines) or other equipment.

For general agricultural water uses a trigger value of 350 mg/L $CaCO_3$ is recommended to limit excess encrustation. This value takes into consideration the influence of hardness on fouling rates.

pH

pH, which is a measure of the acidity or alkalinity of a water based on its hydrogen ion [H^+] activity, can influence the rate of fouling in distribution systems (see Section 9.2.9.1).

Fouling indices

Langelier's index

Langelier's Index (see Section 9.2.9.2 on corrosion) uses pH values to estimate the potential for calcium carbonate precipitation. A positive value indicates that precipitation is likely to occur while a negative value indicates the potential for corrosion.

Ryznar's index

Ryznar's Index (see Section 9.2.9.2 on corrosion) is also used in estimating the potential for fouling or corrosion based on pH of water. All values in this index are positive, with values under 6 indicating a tendency for fouling and values over 7 indicating a tendency for corrosivity.

Ratio of chloride to carbonate

When the log of ($[Cl^-]/[CO_3^{2-}]$) is small (see Section 9.2.9.2 on corrosion), the potential for precipitation of calcium carbonate is high. However, as the Cl^- concentration increases relative to CO_3^{2-}, and the log of the ratio exceeds about 2, corrosion is more likely to occur (Kelly & Kemp 1975).

Control measures

A number of methods are available to limit fouling of agricultural water distribution systems. These tend to be based on treating the symptoms rather than the cause, as it is difficult to prevent most fouling processes unless water quality is altered.

Monitoring and maintenance of water quality used for agriculture is an important part of an overall farm management strategy to ensure long-term sustainability. Fouling of pumping and distribution systems may be detected through changes in some water quality parameters including increase in sulfides, bacterial count, and sporadic increases in turbidity and iron levels. Monitoring of indicative water quality parameters should be conducted on a regular basis to ensure early detection of fouling problems.

Equipment maintenance

Equipment fouling can be minimised through a number of different methods. It is recommended that where possible, flow rate and temperature are kept fairly constant to minimise the likelihood of precipitation. The use of joints and other fittings which may also

alter flow rate and flow diameter should also be kept to a minimum, as these often represent an ideal environment for accumulation of precipitates and biofilm.

Where a pipeline or well continues to be blocked through fouling, poor design combined with unsuitable water quality could be the problem. Replacement or closure may prove a more viable option than continued treatment, as the cost of ongoing maintenance can often be quite substantial.

Changes in water quality

Reduction of pH or hardness may reduce the likelihood of fouling in some systems depending on the mechanism involved. Lowering pH by the addition of hydrochloric or sulphuric acids to water can prevent fouling in the distribution system. In most cases, water with a pH <6 will ensure that iron, calcium and magnesium ions, the principal cations involved in fouling, remain in solution.

Hardness can be controlled by treating the water source through a number of different methods, including, ion exchange, lime softening, and reverse osmosis. Although these methods are effective in reducing hardness in small-scale situations (e.g. domestic consumption), the economic viability in treating large volumes for irrigation or stock use must be considered.

The ion exchange process softens water by passing it through an exchange resin where the calcium and magnesium are replaced with sodium. The resin requires regeneration periodically, which can be done by flushing with a solution of sodium chloride (common salt). Optimum operating conditions for this method are a pH range between 7 and 8, and temperature <32°C (Awad 1989).

Lime softening is usually used in situations where softening of water is needed on a continual basis. The process consists of a number of steps and requires the establishment of pumps, filters and settling tanks for water treatment. Hydrated lime is added to water to precipitate out calcium carbonate, which is separated and removed through filtration and settling.

Reverse osmosis is used primarily to desalinate water by reducing ion concentrations in solution. Water travels through a semi-porous membrane under pressure. This results in a weaker concentration of ions (approximately 90% less than the original solution) and reduced hardness, due to the removal of salts and other components (including calcium and magnesium). Reverse osmosis is expensive for general agricultural practice but may have applications in the amenity horticulture industry.

Sequestering agents and acids

Sequestering agents are sometimes used in the prevention of iron, manganese and calcium carbonate deposits in water distribution systems. They are usually based on phosphate compounds (e.g. sodium hexametaphosphate) and are added to water to act as a form of water softener.

In some cases, acids can be flushed through pipelines to dissolve any deposits forming within the distribution system. Hydrochloric acid is most widely used, however special attention must be paid to the corrosive potential of this acid on metal and concrete surfaces. The most effective treatment method involves recirculating the acid to ensure removal of precipitates, however, this may not be possible in all situations.

Derivation of guideline values

Due to the interrelationship between corrosion and fouling, a combined literature review was conducted on the effect of these processes in relation to agricultural water quality. As discussed in the corrosion Section, the National Centre for Groundwater Management (University of Technology, Sydney) has undertaken a literature review and extensive research on these issues in relation to groundwater, which can be applicable to many agricultural situations. Publications from these studies provided the basis of many issues discussed in this review, in conjunction with other relevant information.

9.2.9.4 Agricultural chemical preparation

Salinity

Elevated salinity levels in agricultural waters can result in the formation of precipitates after mixing with particular chemical compounds. This can adversely affect chemical performance, particularly when the active ingredient is removed from solution. Brackish water commonly contributes to this situation and is generally considered unsuitable.

However, it has been noted that the use of sea water does not influence the efficacy of most herbicides (Anderson & van Haaren 1989, Bovey 1985). This is most likely due to its chemical composition, which is highly buffered and generally composed of only a small percentage of calcium and magnesium salts.

Surface waters and groundwaters (which are generally unbuffered), usually have a lower total salt content that may comprise up to 99% calcium and magnesium salts. When combined in solution with other chemical compounds, these have a greater tendency to precipitate out through supersaturation and alteration of equilibrium conditions.

pH

Extremes in pH causing elevated acidity or alkalinity in waters, can result in the hydrolysis of pesticides and other agricultural chemicals; e.g. carbamate and organophosphorus insecticides will hydrolyse rapidly in alkaline waters with pH levels >7 (Banks et al. 1989, Lantzke 1999).

To minimise the likelihood of hydrolysis occurring, it is recommended that waters with pH around 7 be used. If this option is not feasible, it is recommended that the solution be used immediately after mixing or that pH be altered through the addition of chemicals such as monoammonium phosphate or sulfate, which can be added at a rate of 0.5–1.0 g/L to decrease alkalinity.

Hardness

Hardness is generally defined by the presence of calcium and magnesium salts in water. High levels of these ions can result in the formation of unwanted precipitates.

Specific ions

The presence of certain ions in agricultural waters can lead to mixing problems through the occurrence of unwanted chemical reactions and reduced or altered product performance. This is of particular concern in the case of fertiliser composition, which may alter dramatically with the presence of particular ions (e.g. iron). Phytotoxicity, degradation of soil structure and other adverse impacts can occur if ionic species present in agricultural waters are not considered.

Suspended or dissolved solids

Chemical components of pesticides, fertilisers and other products can often bind with particulate material present in water, resulting in blockage of spray equipment or reduced product performance.

Banks et al. (1989) noted that the presence of suspended clay minerals, which can be a common problem in spray waters, greatly reduced the efficacy of some pesticides (e.g. paraquat). This may occur through adsorption of certain active chemical constituents to the clay particles which are subsequently removed from solution as sediment. The effect of elevated levels of solids can be minimised by checking water visually before use and ensuring it appears clear.

Determination of water quality suitability

To check whether a particular water is suitable for use with an agricultural chemical, it is best to make up and test a trial solution first. Specific details on water quality requirements should be noted from the product label or by contacting the manufacturer.

9.2.10 Future information needs for irrigation and general water use

9.2.10.1 Biological parameters

The issues of both animal and plant pathogens in irrigation waters are becoming of increasing importance, as greater emphasis is placed on the re-use of wastewaters from sewage and intensive animal and plant production industries.

Detection of pathogens in irrigation water is time consuming and expensive. Currently, it is common practice to monitor and control microbiological water quality on the basis of concentration of indicator organisms. This method may not be suitable for irrigation water quality. Further research is needed to determine survival rates of pathogens after irrigation, on vegetative surfaces and in soil before realistic trigger values can be set.

Present information on plant pathogens is limited. The nursery industry has conducted preliminary research, but further work must be undertaken before trigger values can be set for individual species of pathogen.

There is presently little information available concerning guidelines for cyanobacteria in irrigation water. ARMCANZ and the NHMRC have established a working group as part of the National Algal Management Strategy to examine the issue of guidelines for cyanobacteria and their toxins in surface waters (including drinking, recreation and irrigation waters). It is likely that considerably more research will be needed before guidelines can be developed for irrigation water.

9.2.10.2 Salinity and sodicity

New guidelines developed in this document for salinity and sodicity of irrigation waters incorporate a considerable body of recent research information. A key priority now is to make the information available in a variety of forms to suit the needs of different user groups. Further development of simple-to-use decision support tools, including those that utilise computer software packages, will greatly enhance the adoption of the salinity and sodicity trigger values and facilitate more sustainable management of irrigated land in Australia.

Guidelines within this document have focused on steady state predictions in a summer rainfall environment across a wide range of soil types. However, an understanding of the dynamics and transient changes of soil salinity and sodicity will be required to implement management options at the farm level for marginal quality waters over a wider climatic range.

Current and ongoing research into salinity processes operating at the catchment scale will require an assessment of the localised impact of salinity and sodicity as a key component of any management options. It will be important for sustainable irrigation management to fully integrate all aspects of salinity (at both the local and regional scale) in any assessment of irrigation water quality. Integrated catchment modelling is a relatively new field of research and it would be expected that any guidelines relating to salinity would be reviewed as new information and understanding is developed.

9.2.10.3 Heavy metals and metalloids in irrigation water

While the potential toxicity of metals and metalloids to the soil biota (micro- and macro-flora and fauna) is an issue receiving international attention, and ecotoxicity is generally observed at lower soil concentrations than phytotoxicity, research in this area is in its infancy. Although the guidelines have considered these aspects of the potential environmental impacts of inorganic contaminants in irrigation water on soil biota, insufficient information is available at present to be able to set water quality guideline values based on ecotoxicity to soil biota. Future revisions of the guidelines should consider ecotoxicological impacts of contaminants when suitable background information becomes available.

The guidelines have taken the step of assessing chromium on the basis of the chromium (VI) ion as there is little evidence that the chromium (III) ion is a significant environmental risk. However, almost no data are available regarding chromium (VI) levels in irrigation waters, and Australian soils or on toxicity thresholds in soils. It is strongly recommended that these data be obtained and a soil loading limit (CCL) for chromium (VI) be determined as a matter of priority.

A CCL has not been determined for fluoride, as there are insufficient Australian soils data to determine background concentrations and soil concentrations which may be phytotoxic. A similar situation exists for boron. It is recommended that an attempt is made to obtain sufficient data to allow a soil loading limit (CCL) to be determined for these two elements. However, it should be noted that a CCL for boron should be based on extractable concentrations in soils.

Future guidelines should consider the bioavailable fraction of the contaminant in irrigation waters and soil rather than the total concentration as in the current guidelines. There are many factors that can modify the bioavailability and toxicity of contaminants, such as soil pH, texture, irrigation water salinity, organic matter content of soils, and the chemical form of the contaminant in irrigation waters. Total concentrations can therefore be poor indicators of potential negative impacts.

9.2.10.4 Phosphorus

Prior to these guidelines, no guideline value for phosphorus (P) concentration in irrigation water had been set (ANZECC 1992, DWAF 1996a). The interim model described in Section 9.2.6.3 has been developed to restrict environmentally significant concentrations of P (i.e. concentrations with the potential to cause algal blooms) moving into water bodies.

In developing the model, the major sinks of P in the soil environment, and the variable nature of P reactions in soils, were considered. Many of these reactions are complex and site-specific, but in order to aid functionality, the model was kept as simple as possible. To minimise off-site impacts, the model therefore considered P removal from irrigated soils through the harvestable portion of crops, soil P sorption/retention capacities of soils and P fertiliser input to the soil.

In its present form the model does not consider soil colloidal P, preferential macropore flow or surface fluxes of P, and there are presently limited data available to quantify these fluxes (Ritchie & Weaver 1993, Sharpley 1993, Stevens et al. 1999, Kirkby et al. 1997, Nash & Murdoch 1997). The model also assumes that soil solution P concentrations are related to P movement through and over soils into water bodies. However, data relating soil solution, or soil extractant, P concentrations to surface and subsurface P pathways are currently limited (Daniel et al. 1998, Dils et al. 1999, Edwards & Withers 1998, Ulen 1998).

Many of the limitations above are areas that require further research, focusing on achieving a balance between plant availability of phosphorus in soil and restriction of phosphorus leaching/movement into waterways. As more is learned regarding the movement of P in catchments, the interim model calculating site-specific STVs for P presented in these guidelines should be progressively refined.

9.2.10.5 Pesticides

Few guidelines exist for acceptable concentrations of pesticide residues in waters used for irrigation purposes. Those that do exist are made up of a small subset of herbicides that potentially could be found in irrigation waters and they consider only likely adverse effects on crop growth. They do not address the issue of potential impacts on downstream aquatic ecosystems, although this is arguably an issue of greater relevance to on-site management and disposal of irrigation waters. Moreover, the guidelines are based on relatively limited information.

All available information for deriving irrigation water quality guidelines needs to be collated, and priority given to studies that will extend the database to enable the range of pesticides covered by guidelines to be expanded.

9.2.10.6 Other irrigation water quality issues

Many of the irrigation water quality guidelines provided in this document require consideration of soil properties in assessing the suitability of waters for irrigation in specific situations. The guidelines have been derived on the basis of irrigating ‘natural’ soils and in some instances they may not be appropriate for use where artificial media are being irrigated.

The use of soil-less media is growing rapidly (e.g. in the nursery and landscaping industries), with a diversity of products used in media formulations. The applicability of the present guidelines for use with these media needs to be assessed including e.g. salinity and toxicity issues.

Another topic not addressed in the present guidelines concerns water quality issues for use in hydroponics. As well as information on plant pathogens (where water is recycled), other research/information needs include salinity issues and major ion concentrations compatible with mixing nutrient formulations.

9.2.10.7 Corrosion and fouling issues

There have been many attempts to relate corrosion and fouling to water quality in both surface waters and groundwaters, but no indicators have been found to be universally applicable. Current evaluation criteria take into account only the inorganic precipitation of compounds and do not include microbial factors, the interaction with other compounds in solution or the rate at which the reaction will occur (McLaughlan 1996).

Further research is continuing through the University of Technology, Sydney (National Centre for Groundwater Management) and through borehole corrosion studies conducted by the Australian Geological Survey (AGSO). Priorities identified by the AGSO for further research on corrosion processes in the Great Artesian Basin include the role/s of anaerobic bacteria in corrosion processes and the role of shear stress and protective film formation (Larsen et al. 1996).

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9.3 Livestock drinking water guidelines

9.3.1 Introduction

Livestock production in Australia and New Zealand relies on both surface water and groundwater supplies. Water quality in streams and dams (surface waters) is influenced by catchment geology, topography, soil type and climate. Groundwater, which is used as a source of drinking water for livestock over a large area of Australia (and in parts of New Zealand), 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. The quality of both groundwaters and surface waters may be affected by catchment land use practices, including agriculture, mining and other industries, with the potential for increased concentrations of salt, nutrients and other contaminants, such as pesticide residues and heavy metals.

Daily water intake varies widely among different forms of livestock and is also influenced by factors such as climate and the type of feed being consumed. Average and peak daily water requirements for a range of livestock are given in table 9.3.1.

Table 9.3.1 Stock water requirements[a]

Type of livestock		Average daily consumption	Peak daily consumption
Sheep		(litres/head)	(litres/head)
	Nursing ewes on dry feed	9	11.5
	Mature sheep on dry pastures	7	8.5
	Mature sheep on green pastures	3.5	4.5
	Fattening lambs on dry pasture	2.2	3
	Fattening lambs on green pasture	1.1	
Cattle			
	Dairy cows in milk	70	85
	Dairy cows, dry	45	60
	Beef cattle	45	60
	Calves	22	30
Horses			
	Working	55	70
	Grazing	35	45
Pigs			
	Brood sows	22	30
	Mature pigs	11	15
Poultry		(litres/100 birds)	(litres/100 birds)
	Laying hens	32	40
	Non-laying hens	18	23
	Turkeys	55	70

a From Burton (1965)

9.3.2 Derivation and use of guidelines

Information used to determine the trigger values was sourced from the current literature and evaluated for relevance, with preference given to data from Australia and New Zealand. Details of the databases searched are provided in Section 9.2. Material provided by the public was also considered. Much of the information found in the literature was based on field observations rather than rigorous experimentation. In several cases it was possible to calculate trigger values using data on chronic and toxic effect levels on animals, taking into consideration animal weights, percentage intake from water, and safety factors for data not

specific to the species. Derivation of most trigger values for livestock drinking water requires further validation and they should be considered interim at this stage. The particular methodologies used to develop specific trigger values are discussed further in relevant Sections.

Consistent with guidelines derived for other environmental values, these guidelines are trigger values. Below the trigger value there should be little risk of adverse effects on animal health. Above the trigger value, investigations are recommended (e.g. of other factors such as age, condition, other dietary sources) to further evaluate the situation.

9.3.3 Biological parameters

9.3.3.1 Cyanobacteria (blue-green algae)

Algal blooms should be treated as possibly toxic and the water source should be withdrawn from stock until the algae are identified and the level of toxin determined.

An increasing risk to livestock health is likely when cell counts of* Microcystis *exceed 11 500 cells/mL and/or concentrations of microcystins exceed 2.3 µg/L expressed as microcystin-LR toxicity equivalents. There are insufficient data available to derive trigger values for other species of cyanobacteria.

Source

Cyanobacteria (often called blue-green algae because they are similar to algae in habitat, morphology and photosynthetic activity) are a component of the natural plankton population in healthy and balanced surface water supplies. They are found as single cells or in clumped or filamentous colonies. Cyanobacteria can move vertically through water by adjusting their buoyancy (Ressom et al. 1994).

In Australia the most common genera of toxic cyanobacteria associated with known animal poisoning incidents are *Microcystis* (colonial); and *Anabaena* and *Nodularia* (filamentous) (Steffensen et al. 1998). The genus *Cylindrospermopsis* has been identified in surface waters, mainly in tropical and subtropical areas (Queensland Water Quality Task Force 1992, Jones et al. 1993, Jones 1994). Cyanobacteria only become a potential hazard when they are present in large numbers (blooms). Blooms typically occur on warm days with light to calm winds (summer to autumn) in waters of neutral to alkaline pH containing elevated levels of inorganic phosphorus and nitrogen, although blooms at other times are possible (Carmichael 1994). There may be often more than one species of cyanobacteria associated with a bloom (Ressom et al. 1994).

Animal health

The toxins associated with cyanobacteria are mostly intracellular in healthy blooms and only affect stock following direct ingestion of cells (either in the water or as dried mats left on the shore), or from drinking water where the death of cells has caused a considerable release of toxins into the water supply. In the latter situation it may take weeks for toxins to be degraded by naturally occurring bacteria (Carmichael 1994, Jones 1994).

Not all blooms of cyanobacteria appear to be hazardous to animals for the following reasons (Carmichael & Falconer 1993):

- only low concentrations of toxins may be associated with the bloom;

- stock are not equally susceptible to algal intoxication — species, age and sex affect susceptibility;
- the amount of toxin consumed may be small and/or countered by the amount of other food in the animal's gut.

Worldwide, the most common cyanobacterial toxin is microcystin, a hepatotoxin which is produced predominantly by the genus *Microcystis,* and occasionally by species of *Anabaena*, although this appears to be rare in Australia. There may be some differences between animal species in the symptoms of this type of poisoning, but typically they include a display of weakness, lethargy, anorexia, paleness, sometimes mental derangement, and often accompanied by diarrhoea. In serious cases animals suffer general distress, muscle tremors and coma which is followed by death within a few hours to a few days. Animals, particularly cattle, which survive hepatotoxicosis may suffer from photosensitisation resulting in cows refusing to suckle their young (Carmichael & Falconer 1993). *Nodularia spumigena,* which produces another hepatotoxin, nodularin, was the first well-documented case in the world of a cyanobacterial outbreak, at Lake Alexandrina, South Australia in 1878 (Francis 1878). Domestic animals in Australia have been affected by exposure to nodularin (Steffensen et al. 1998).

The neurotoxins produced by *Anabaena circinalis* are a group of closely related alkaloids known as saxitoxins. When ingested by animals, these toxins restrict message transmission between neurones which affects muscle tissues, including those required for breathing. Death is almost always due to respiratory failure (Negri et al. 1995, Steffensen et al. 1998). Water containing *A. circinalis* at 50 000 cells/mL caused the death of sheep in Central New South Wales (Negri et al. 1995). Since the neurotoxins act more rapidly, their effects will be more obvious than the effects of hepatotoxins, in cases where both are present (Carmichael & Falconer 1993).

Cylindrospermopsin is a cytotoxic alkaloid associated with the nitrogen fixing *Cylindrospermopsis raciborskii*. This toxin affects the liver, kidney, small intestine and lungs of animals which can result in death (Hawkins et al. 1996).

There have been few toxicological trials carried out to determine safe levels of intake of cyanobacterial cells or toxins for domestic animals. Falconer et al. (1994) in experiments with bloom material of *Microcystis aeruginosa* showed there was no adverse effect on the livers of pigs supplied with 280 μg toxins/kg/day via drinking water over a period of 44 days. Long-term effects of ingestion of lower levels of toxins are not well understood.

While the risk of possible accumulation of toxins in animal products for human consumption is not fully known, a study of dairy cattle ingesting up to 15 mg of Microcystin-LR over a period of three weeks showed no transmission of toxin into the milk (G Jones, pers comm).

Derivation of trigger value

Establishing trigger values based on health considerations of animals is difficult for the following reasons:

- not all blooms appear to be toxic, and toxic and non-toxic blooms of the same species have been found;
- the toxicity per cell can vary over time (weeks to months), making it difficult to relate cell numbers to toxicity (toxin levels); and
- insufficient toxicological data are available for all toxins.

To derive reliable trigger values, accurate and accessible methods for determination of toxins in water need to be further developed, and data provided on the acute and chronic effects of these toxins on domestic animals.

Microcystin

The following calculations and assumptions were used to derive a trigger value for microcystin-LR toxicity equivalents. They are based on the principles adopted by the United States Environmental Protection Agency (Belluck & Anderson 1988, cited by Hamilton & Haydon 1996) and the World Health Organisation (Falconer et al. 1999). The example given is for pigs; data for other livestock are provided in table 9.3.2.

For pigs:

$$\text{trigger value} = \frac{\text{LOAEL} \times \text{animal weight}}{\text{max daily water intake} \times \text{safety factor}} = \frac{100\,\mu\text{g/kg/day x 110kg}}{15\text{L/day} \times 45} = 16.3\,\mu\text{g/L} \qquad (9.47)$$

where:

- 100 µg microcystin-LR toxicity equivalents/kg bw/day is the Lowest Observed Adverse Effect Level (LOAEL) for pigs fed over 44 days (Falconer et al. 1994, Kuiper-Goodman et al. 1999);
- 110 kg is the upper weight of pigs going to market;
- 15 L/day is the peak consumption of water for pigs at this stage of development;
- 45 is the safety factor to allow for the less than lifetime study, varying susceptibilities of animals and deriving a NOEL (No Observed Effect Level) from the LOEAL of the pig study.

Table 9.3.2 Summary of calculations for microcystin-LR equivalent levels and cell numbers of *Microcystis aeruginosa* used to develop a guideline for a range of livestock

Animal	Body weight	Peak water intake	Safety factor					Toxin level calc.	Equivalent cell number[a]
	(kg)	(L/day)	Less than lifetime	Inter-species variation	Intra-species variation	LOAEL to NOEL	Total	(µg/L)	(cells/mL)
Cattle	800	85	3	5	3	5	225	4.2	21000
Sheep	100	11.5	3	5	3	5	225	3.9	19500
Pigs	110	15	3	1	3	5	45	16.3	81500
Chickens[b]	2.8	0.4	3	5	3	5	225	3.1	15500
Horses	600	70	5[c]	5	3	5	375	2.3	11500

a Assuming 0.2 pg total microcystins/cell (Falconer et al. 1994)

b These values can be taken to represent all poultry, since all poultry have a very similar body weight/water intake ratio.

c Horses generally live longer than other livestock

Using the above approach, estimated trigger values for microcystin-LR toxicity equivalents for various types of livestock range from 2.3 to 16.3 µg/L, equivalent to 11 500 to 81 500 cells/mL of *Microcystis aeruginosa* (table 9.3.2). Taking the most sensitive animals (horses), the value of 11 500 cells/mL can be used as a trigger value, below which little or no risk to stock should occur.

Other cyanotoxins

There are presently insufficient animal toxicity data available to derive trigger values for cyanotoxins other than microcystins in livestock drinking water.

Diagnostic procedure

The presence of an algal bloom does not necessarily mean that animals will be poisoned, so the following steps should be taken to assess the risk from such a bloom (after Carmichael & Falconer 1993).

1 Establish that animals are drinking the water or eating algal mats from the area where there is a substantial bloom.

2 Indentify the algae associated with the bloom to determine whether cyanobacteria are present in numbers large enough to constitute a risk.

3 If necessary, chemically analyse a sample of the bloom to identify and quantify toxins present.

Since all blooms of cyanobacteria have the potential to be toxic and all livestock are susceptible, it is prudent to consider all scums toxic until proven safe, as described above. In the interim, stock should be withdrawn from the water supply and an alternative source used. Where an alternative source is not available and the bloom is localised, it may be possible to allow stock to drink from an area on the upwind side of the bloom. In the long term, prevention of blooms is by far the best strategy and water supplies should be managed so that nutrient inputs are minimal.

9.3.3.2 Pathogens and parasites

Drinking water for livestock should contain less than 100 thermotolerant coliforms/100 mL (median value).

Source

A large variety of microbial pathogens can be transmitted to stock from drinking water supplies contaminated by animals and their faeces. The risk of contamination is greatest in surface waters (dams, watercourses, etc) which are directly accessible by stock or which receive runoff or drainage from intensive livestock operations or human wastes. The incidence of groundwater contamination by pathogens is generally low, particularly for deep bores and wells. Some shallow groundwater supplies have the potential to be contaminated, particularly in sandy soils.

Management of water supplies to minimise contamination is the best strategy for protecting livestock from water-borne microbial pathogens. Effective measures include preventing direct access by stock to watercourses and minimising drainage of waters containing animal wastes to streams and groundwaters.

Animal health

Infections in livestock often result in reduced growth and morbidity and possibly mortality (Smith et al. 1974).

The bacteria of most concern in water supplies with unacceptably high bacterial counts are the enteric bacteria, *Escherichia coli* and *Salmonella* and to a lesser extent *Campylobacter jejuni* and *C coli, Yersinia enterocolitica* and *Y pseudotuberculosis*. Other bacteria known to affect stock and which may be transmitted through water supplies include *Leptospira*

(leptospirosis), *Burkholderia (Pseudomonas) pseudomallei* (melioidosis), *Clostridium botulinum* (botulism), Mycobacteria (pulmonary disease), *Pseudomonas* (mastitis) and Cyanobacteria (blue-green algal toxicosis, see Section 9.3.3.1).

A number of serious pathogenic conditions in livestock can be caused by viruses. Water supplies have been implicated in transmitting Newcastle disease and infectious bursitis in poultry (CCREM 1987).

Well-managed livestock usually have a relatively low incidence of parasitic infections. Most infections do not cause mortality directly, but reductions in growth rates and vitality occur and susceptibility to fatal infectious disease organisms increases (CCREM 1987). A number of stock parasites spend part of their life-cycles in water, and faecal contamination of water is the usual means of introduction. One parasitic disease of concern in Australia is cysticercosis in cattle (beef measles) caused by the tapeworm *Taenia saginata* (Arundel 1972).

Experiments with lambs have shown that the minimum infectious dose of the protozoan *Cryptosporidium parvum* may be as little as one oocyst and that the infection may be water-borne (Blewett et al. 1993). *Giardia* is another protozoan which can be transmitted in water. Weight loss in stock has been reported from infection with *Giardia* (Olson et al. 1995).

Water-borne pathogens not only affect stock health, but may also impact on human health. It is reasonable to assume that a contaminated water supply introducing high numbers of organisms into a group of animals may create a 'multiplier' effect through the food chain. High numbers of pathogens (e.g. the enterohaemmorhagic *E. coli*) in the herd could then lead to high numbers of organisms on meat, with increased risk of infections in human consumers.

Derivation of trigger value

Expanding interest worldwide in the use of reclaimed wastewaters for agricultural purposes has generated much of the recent activity in developing guidelines for their safe use for this and other purposes. Although the present guidelines concern natural waters rather than reclaimed waters, the underlying issues regarding risks to human and animal health are the same.

In Australia and New Zealand, the management and use of reclaimed water from sewerage systems forms an important component of the National Water Quality Management Strategy (NWQMS). Guidelines for pathogen levels in stock drinking water have been proposed in the NWQMS document, *Guidelines for sewerage systems — use of reclaimed water* (ARMCANZ, ANZECC & NHMRC 2000). These guidelines have been adopted for use in the present water quality guidelines for primary industries.

It is generally not feasible nor warranted to test livestock drinking water for the presence of the wide range of water-borne microbial pathogens that may affect stock health. In practice, water supplies are more commonly tested for the presence of thermotolerant coliforms (also known as faecal coliforms), to give an indication of faecal contamination and thus the possible presence of microbial pathogens. However, note that in tropical and sub-tropical areas thermotolerant coliforms may on some occasions include microorganisms of environmental rather than faecal origins (NHMRC & ARMCANZ 1996). Moreover, the test does not specifically indicate whether pathogenic organisms are present or not. Testing for specific organisms may be necessary in these situations if animal health is affected.

The NWQMS guidelines for pathogens in stock drinking water (ARMCANZ, ANZECC & NHMRC 2000) were proposed after consideration of the methodologies and information used in developing guidelines proposed by the World Health Organization (WHO 1989) and the United States Environmental Protection Agency (USEPA 1992), together with local

considerations. This is consistent with WHO recommendations that the WHO (1989) guidelines be adapted according to local conditions and socio-economic factors (Hespanhol & Prost 1994). The ARMCANZ, ANZECC & NHMRC (2000) guidelines are based on:

- the best available scientific evidence;
- worldwide practice in reclaimed water use;
- a consensus of local practice demonstrated to be safe.

It is recommended that a median value of thermotolerant coliforms is used, based on a number of readings generated over time from a regular monitoring program. Investigations of likely causes are warranted when 20% of results exceed four times the median guideline level (ARMCANZ, ANZECC & NHMRC 2000).

9.3.4 Major ions of concern for livestock drinking water quality

9.3.4.1 Calcium

Stock should tolerate concentrations of calcium in water up to 1000 mg/L, if calcium is the dominant cation and dietary phosphorus levels are adequate. In the presence of high concentrations of magnesium and sodium, or if calcium is added to feed as a dietary supplement, the level of calcium tolerable in drinking water may be less.

Source

Calcium is found in natural waters over a wide range of concentrations. The level of calcium in water is related closely to the geology of the source areas, the calcium being derived by weathering processes from minerals such as gypsum, limestone and dolomite. Calcium contributes to the hardness of the water, which may cause scaling problems in pipes, troughs and fittings (see Section 9.2.9.3).

Animal health

Calcium is an essential element in the animal diet. However, high calcium concentrations may cause phosphorus deficiency by interfering with phosphorus absorption in the gastrointestinal tract and calcious formation in the body (Mulhearn 1964). Long-term intake by sheep of water containing around 1100 mg/L calcium was found to have no adverse effect on health and wool production, although the calcium concentration of plasma increased, while the sodium concentration decreased (Peirce 1960).

Derivation of trigger value

The ANZECC (1992) guideline for calcium has been retained in the absence of any new contradictory information. The trigger value of 1000 mg/L is consistent with guidelines developed in both Canada (CCREM 1987) and South Africa (DWAF 1996b).

9.3.4.2 Magnesium

Insufficient information is available to set trigger values for magnesium in livestock drinking water.

Source

The concentration of magnesium in natural waters varies considerably, with concentrations in natural freshwaters ranging from <1 mg/L to >1000 mg/L, depending on catchment geology (Meybeck 1979, Galvin 1996, APHA, AWWA & WEF 1998). Magnesium contributes to the hardness of water and may cause scaling problems in troughs and fittings (see Section 9.2.9.3).

Animal health

Recent work by CSIRO in Queensland suggests that Brahman steers can tolerate magnesium concentrations in drinking water up to 2000 mg/L with no adverse effects (GS Harper, pers comm). Several earlier studies have reported possible adverse effects on livestock from drinking water containing magnesium at concentrations of 250 mg/L and higher (Peirce 1960, Saul & Flinn 1978, 1985, VIRASC 1980). However, it is not clear whether the reported effects were due to magnesium *per se* or whether they were confounded by other issues such as the overall salinity of the water or the presence of other specific ions (e.g. sulfate) known to have adverse effects.

High magnesium concentrations in water are generally associated with high concentrations of total dissolved salts (TDS), hence many problems attributed to magnesium may well be due to the high TDS levels. Flinn (1980) showed that concentrations of 400–600 mg/L magnesium were typically found in water containing 8000–12 000 mg/L TDS which is at the upper limit of tolerance by stock. The findings of Saul and Flinn (1985) would also seem to support this position.

Derivation of trigger value

Present information is inconclusive regarding the effects of magnesium levels in drinking water on animal health. No trigger value is recommended until further information from animal feeding trials becomes available.

The ANZECC (1992) guidelines (based on Flinn 1984) gave an upper limit for magnesium for all forms of livestock of 600 mg/L but this is not now supported, for the reasons given above. Present Canadian Water Quality Guidelines (CCREM 1987) do not include a guideline for magnesium in livestock drinking water; while in South Africa, an upper limit of 1000 mg/L magnesium is proposed, with some adverse effects considered likely to occur at magnesium concentrations between 500 and 1000 mg/L (DWAF 1996b).

9.3.4.3 Nitrate and nitrite

Nitrate concentrations less than 400 mg/L in livestock drinking water should not be harmful to animal health. Stock may tolerate higher nitrate concentrations in drinking water provided nitrate concentrations in feed are not high. Water containing more than 1500 mg/L nitrate is likely to be toxic to animals and should be avoided.

Concentrations of nitrite exceeding 30 mg/L may be hazardous to animal health.

Source

Nitrate and nitrite are oxidised forms of nitrogen, both of which can occur naturally in waters, although nitrate generally predominates. Nitrate is usually present in unpolluted streams at concentrations below 1 mg/L (Meybeck 1982). Higher concentrations are often associated with over-use of nitrogen fertilisers and manures; intensive livestock operations; and/or leakage from septic systems and municipal wastes. Elevated nitrite concentrations typically are found only under anoxic conditions, for example where waters are polluted by organic wastes.

Groundwaters may contain elevated nitrate concentrations due to natural processes (Lawrence 1983) but more typically, high nitrate concentrations in groundwaters are associated with contamination. Nitrate concentrations >20 mg/L have been reported in many Australian groundwaters, with a small proportion showing concentrations >100 mg/L nitrate (Lawrence 1983, Keating et al. 1996).

Overfertilisation of plants with nitrogen fertilisers, poultry litter or animal manures can lead to excessive nitrate accumulation in plants. Plants under stress (e.g. from drought, or a lack of adequate nutrition or sunlight) may also accumulate nitrate. Animals are likely to be at higher risk of nitrate/nitrite poisoning through consumption of pastures, forages and feeds containing high levels of nitrate than from their water supplies.

Confusion can arise concerning guideline values for nitrate and nitrite, because concentrations are sometimes reported on the basis of their respective nitrogen (N) contents, that is, as nitrate–N (NO_3–N) and nitrite–N (NO_2–N). The conversions are as follows:

$$1\text{mg/L } NO_3\text{–N} = 4.43 \text{ mg/L } NO_3 \quad (9.48)$$

$$1\text{mg/L } NO_2\text{–N} = 3.29 \text{ mg/L } NO_2 \quad (9.49)$$

Note that guideline values presented here are for nitrate and nitrite.

Animal health

Both nitrate and nitrite can cause toxicity, with nitrite being 10–15 times more toxic than nitrate (Case 1963). To cause toxicity, nitrate must first be reduced to nitrite, which is an intermediate product of the reduction of nitrate to ammonia by bacteria in the rumens of sheep and cattle and to some degree in the cecum of horses. Non-ruminants (pigs and chickens) are less susceptible as they rapidly eliminate nitrate in the urine.

Nitrite is absorbed into the bloodstream, where it converts haemoglobin to methaemoglobin, thus reducing the oxygen-carrying capacity of the blood and causing eventual suffocation due to a lack of oxygen in body tissues. Symptoms of acute poisoning include increased urination, restlessness and cyanosis, leading to vomiting, convulsions and death.

Rumens of animals previously fed high nitrate diets show an increased rate of nitrate/nitrite reduction. Nitrate toxicity is also dependent on the rate of consumption, with slow intake and a balanced ration reducing toxicity (Crowley 1985).

Winks (1963) reported death of calves and cattle in Queensland from drinking water containing 2200 mg/L nitrate. He suggested a toxic nitrate concentration for cattle as somewhere between 300 mg/L and 2200 mg/L. Seerly et al. (1965) concluded that drinking water containing approximately 300 mg/L nitrate–N had no effect on the health of pigs or sheep and that levels of nitrite–N less than 100 mg/L over 105 days did not adversely affect pig health. Anderson and Stothers (1978) similarly reported no ill effects in weanling pigs after 6 weeks of drinking water containing around 1300 mg/L nitrate. Sorensen et al. (1994) found no effect on early weaned piglets and growing pigs from water containing up to 2000 mg/L nitrate or up to 17 mg/L nitrite. In experiments carried out in Queensland, pigs raised from 20 to 80 kg showed no decrease in performance and no adverse effects on health, when given water containing up to 500 mg/L nitrate or up to 50 mg/L nitrite (McIntosh 1981). A national survey of pig farms in the US showed no association between animal health or performance and drinking water containing up to 460 mg/L nitrate (Bruning-Fann et al. 1996). In dairy cows, nitrate concentrations up to 180 mg/L in drinking water did not increase the concentration of nitrate in milk (Kammerer et al. 1992).

Derivation of trigger values

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. The total dietary intake of nitrate by livestock needs to be considered when interpreting the trigger values. High nitrate concentrations in the water supply may indicate that nitrate levels in locally grown feed may also be elevated.

Trigger values of 400 mg/L nitrate and 30 mg/L nitrite are recommended for livestock drinking water. Depending on the nitrate content of feed, the type of livestock and other factors such as animal age and condition, concentrations up to 1500 mg/L nitrate may tolerated, at least for short-term exposure.

The recommended trigger values are consistent with present Canadian guidelines for livestock drinking water (100 mg/L nitrate–N; 10 mg/L nitrite–N) (CCREM 1987). In South Africa, trigger values range from 100 to 400 mg/L nitrate, depending on the type of livestock, animal condition and period of exposure (DWAF 1996b).

9.3.4.4 Sulfate

No adverse effects to stock are expected if the concentration of sulfate in drinking water does not exceed 1000 mg/L. Adverse effects may occur at sulfate concentrations between 1000 and 2000 mg/L, especially in young or lactating animals or in dry, hot weather when water intake is high. These effects may be temporary and may cease once stock become accustomed to the water. Levels of sulfate greater than 2000 mg/L may cause chronic or acute health problems in stock.

Source

Sulfate is found in most natural waters as a result of the dissolution of sulfate-bearing minerals in soils and rocks. Sulfate can occur naturally at concentrations up to thousands of milligrams per litre, particularly in groundwaters. Mine waste waters, tannery wastes and other industrial discharges often contain high concentrations of sulfate, while the use of alum as a flocculant may increase the levels of sulfate in stock drinking water.

Under anoxic conditions bacteria in water can reduce sulfate to sulfide, which results in the release of hydrogen sulfide, causing an unpleasant taste and odour and increasing the potential for corrosion of pipes and fittings.

Animal health

Sulfate is an essential element for animal nutrition. Excessive concentrations of sulfate in water typically cause diarrhoea in stock. Animals generally avoid water containing high sulfate concentrations in favour of water containing lower concentrations, where available (Weeth & Capps 1972).

Sulfate can cause diarrhoea in young animals at concentrations of 1000 mg/L (Church 1979). Higher concentrations of sulfate may be tolerated, depending on the species of livestock, age, and the principal cations associated with the sulfate ion, but loss of production may be expected (CCREM 1987). Weanling pigs showed no significant effect on performance from drinking water containing up to 2400 mg/L sulfate for 20 days (although scouring was reported), but performance was reduced at 4880 mg/L sulfate (McLeese et al. 1992). An improvement was reported in productivity and health of dairy cattle when their source of drinking water was changed from deep-well water containing 1500–2500 mg/L sulfate to surface water containing less than 1000 mg/L sulfate (CCREM 1987). Hereford cattle

showed decreased water and food consumption, weight loss and diuresis when consuming water containing 3380 mg/L sulfate (Weeth & Hunter 1971).

Brahman steers fed diluted coal mine pit water containing approximately 2000 mg/L sulfate showed no reduction in performance over 46 days when progressively adapted to the high sulfate concentrations under controlled experimental conditions (Robertson et al. 1996). Similarly, beef steers showed no ill effects when introduced gradually to water containing 2000 mg/L sulfate, but water and dry matter intakes were reduced when animals were exposed to drinking water containing 4000 mg/L (Harper et al. 1997). However, liveweight gains for lactating cows and their calves were found to be significantly reduced by drinking water containing ≥1300 mg/L sulfate, but not at 630 mg/L (Harper et al. 2000).

Very high concentrations of sulfate in drinking water (7200 mg/L) have been associated with an outbreak of polioencephalomalacia in cattle, with symptoms including depression, ataxia, cortical blindness, dysphagia and death (Hamlen et al. 1993).

Derivation of trigger value

The trigger value for the concentration of sulfate in the drinking water of livestock has been adopted after consideration of reported experimental findings from trials feeding water to animals. The guideline is consistent with values recommended for sulfate is livestock drinking water in Canada (CCREM 1987) and South Africa (DWAF 1996b).

Interactions such as those with dietary copper and molybdenum (see Section 9.3.5.14) should be taken into account when deciding the suitability for stock of water containing high sulfate concentrations.

9.3.4.5 Total dissolved solids (salinity)

Recommended concentrations of total dissolved solids in drinking water for livestock are given in table 9.3.3.

Table 9.3.3 Tolerances of livestock to total dissolved solids (salinity) in drinking water[a]

Livestock	Total Dissolved Solids (mg/L)		
	No adverse effects on animals expected	Animals may have initial reluctance to drink or there may be some scouring, but stock should adapt without loss of production	Loss of production and a decline in animal condition and health would be expected. Stock may tolerate these levels for short periods if introduced gradually
Beef cattle	0–4000	4000–5000	5000–10000
Dairy cattle	0–2400	2400–4000	4000–7000
Sheep	0–4000	4000–10000	10000–13000[b]
Horses	0–4000	4000–6000	6000–7000
Pigs	0–4000	4000–6000	6000–8000
Poultry	0–2000	2000–3000	3000–4000

a Adapted from ANZECC (1992); b Sheep on lush green feed may tolerate up to 13000 mg/L TDS without loss of condition or production.

Source

Total dissolved solids (TDS) is a measure of all inorganic salts dissolved in water and is a guide to water quality. The measurement also includes other dissolved substances such as organic compounds, when present. The concentration of TDS in natural waters ranges

widely, from <1 mg/L in rainwater to about 35 000 mg/L in seawater and higher in brines and some natural waters. The TDS of natural waters reflects the geology of source areas; the major contributing ions are typically the cations calcium, magnesium, sodium and potassium, and the anions bicarbonate, chloride, sulphate and in some cases, nitrate.

Surface waters generally have lower TDS concentrations than groundwaters. In streams, TDS can increase through the continual addition of salts by both natural weathering processes and human activities, such as discharges of domestic and industrial effluents and runoff from urban and rural areas. Water supplies in dams, lakes and water troughs can increase in TDS concentrations due to evaporation, particularly if they are not flushed out regularly.

Animal health

Highly mineralised waters can cause physiological upset and sometimes death in terrestrial animals, including humans. Animals under physiological stress, for example due to pregnancy, lactation or rapid growth, are particularly susceptible to mineral imbalances. Livestock generally find water of high salinity unpalatable. Water of marginal quality can cause gastrointestinal symptoms and a reduction in weight gain and milk or egg production. However, livestock can acclimatise physiologically to some extent to water of higher salinity when the level is adjusted over several weeks.

In dairy cattle, a reduction in milk production in cows and decreased liveweight gain have been reported at TDS levels of 4360 mg/L (Challis et al. 1987); 3574 mg/L (Solomon et al. 1995) and 2696 mg/L (Jaster et al. 1978). Saul and Flinn (1985) reported losses in animal production when Hereford heifers were introduced to water containing TDS levels of 5000–11 000 mg/L.

The tolerance of sheep to saline drinking water may depend on the type of forage consumed. Sheep raised in pens were shown to tolerate up to 13 000 mg/L TDS (Peirce 1966, 1968a). However, with sheep raised on pasture, lambs showed increased diarrhoea, heavier mortality and decreased body weight gains at 13 000 mg/L TDS; and reduced body weight gains and wool production at 10 000 mg/L TDS (Peirce 1968b).

The incidence of egg shell defects (thin and cracked shells) in chickens was shown to be significantly increased by an increased intake of mineral salts (Balnave & Scott 1986). Municipal water supplemented with 250 mg/L sodium chloride (NaCl) increased shell defects two fold, while 2000 mg NaCl/L added to drinking water produced up to 50% of all eggs with defects (Balnave & Yoselwitz 1987, Brackpool et al. 1996). The adverse effect of drinking the saline water even for short periods of time during early lay was not overcome when the water supply was replaced with lower salinity water (Balnave & Zhang 1998). Equivalent levels of sodium chloride in feed did not adversely affect egg shell quality (Yoselwitz & Balnave 1989).

While increased water consumption and some initial diarrhoea are common observations when pigs are introduced to water containing >4000 mg/L TDS, concentrations as high as 6000 mg/L TDS are unlikely to adversely effect pigs that have become accustomed to the water (Robards & Radcliffe 1987, Williams 1990). In experiments carried out in Queensland, pigs raised from 20 to 80 kg showed no decrease in performance and no adverse effects on health, when given water containing up to 8000 mg/L TDS, although water consumption did increase with increasing salinity, particularly in summer months (McIntosh 1982).

Derivation of trigger values

Salinity (TDS) is used throughout Australia as a convenient guide to the suitability of water for livestock watering. However, if a water has purgative or toxic effects, especially if the

TDS is above 2400 mg/L, the water should be analysed to determine the concentrations of specific ions.

Table 9.3.3 summarises the salinity tolerances of livestock (from ANZECC 1992), taking into consideration the information supplied above. The guidelines are broadly consistent with those recommended in Canada (CCREM 1987) and South Africa (DWAF 1996b), although there are some differences in TDS concentration ranges proposed for different types of livestock. In Canada, the maximum TDS level that is recommended as safe for livestock consumption is 10 000 mg/L (CCREM 1987).

In natural waters, the electrical conductivity (EC, in dS/m) is directly proportional to TDS (mg/L) by a factor ranging from 550 to 900, depending on the types of dissolved salts present in the water. Typical conversion factors used in Australia include 640 (Gill 1986) and 670 (Rayment & Higginson 1992). For convenience, TDS is often estimated from EC. The following are some useful conversions:

$$1 \text{ dS/m} = 1000 \text{ µS/cm} \quad (9.50)$$

$$\text{EC (dS/m)} \times 670 = \text{TDS (mg/L)} \quad (9.51)$$

$$\text{EC (µS/cm)} \times 0.67 = \text{TDS (mg/L)} \quad (9.52)$$

TDS is sometimes expressed as total dissolved ions (TDI), which is a summation of the concentrations of inorganic ions present in water, but does not include any other substances (e.g. organic compounds) that may also be dissolved in the water.

9.3.5 Heavy metals and metalloids

9.3.5.1 Aluminium

Where aluminium concentrations in water exceed 5 mg/L, stock intake of phosphorus in the diet should be investigated. Animals, particularly ruminants, may tolerate much higher levels of aluminium as long as there is sufficient phosphorus in the diet to compensate for the effects of aluminium.

Source

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 as high as 10 mg/L, due to the increased solubility of soil aluminium oxides and clay minerals (Galvin 1996). The use of alum and other aluminium based flocculants may also be responsible for increased concentrations of aluminium in water supplies.

Animal health

High levels of aluminium react with phosphorus in the intestine of animals to form a non-absorbable complex, thus affecting phosphorus absorption and metabolism and resulting in symptoms of phosphorus deficiency (NRC 1980). Symptoms include reduced growth and disturbances in carbohydrate metabolism. Ruminants may be less susceptible than monogastrics, since organic anions in the rumen may complex the aluminium and prevent it precipitating with phosphate (Thompson et al. 1959, cited by NRC 1980).

No adverse effects were observed when aluminium sulfate was fed to sheep and cows at concentrations of 1215 mg Al/kg (Bailey 1977), or when aluminium chloride was added to feed for steers at concentrations of 1200 mg Al/kg (Valdivia et al. 1978). Based on these results the

NRC (1980) set a maximum tolerable level of aluminium in the diet of cattle and sheep of 1000 mg/kg. Chicks and turkeys showed no effects when fed 486 mg Al/kg, but there is no information on the tolerance of pigs to aluminium (Cakir et al. 1978, cited by NRC 1980).

Derivation of trigger value

The ANZECC (1992) trigger value of 5 mg/L has been retained and is supported by calculation of a theoretical trigger value based on a toxicological approach using data from the literature and assumptions as detailed below.

For cattle:

$$\text{trigger value} = \frac{\text{NOEL x daily feed intake x proportion from water}}{\text{max daily water intake x safety factor}} = \frac{1200\text{mg/kg/day x 20kg x 0.2}}{\text{15 L/day x 10}} = 5.6\ \text{mg/L} \qquad (9.53)$$

where:

1200 mg/kg is the level in the diet for cattle fed over 84 days used as the no observed effect level (NOEL) (Valdivia et al. 1978);

20 kg/day is an estimate of the average food consumption of cattle at this weight assuming they consume about 2.5% their bodyweight in feed;

0.2 is the proportion of aluminium attributed to the intake of water;

85 L/day is the peak consumption of water for cattle;

10 is the safety factor for possible long-term effects and tissue accumulation.

Based on the above approach, estimated trigger values for various types of livestock range from 3.6 to 5.6 mg Al/L (table 9.3.4), consistent with a trigger value of 5 mg/L for all livestock. The guideline is also consistent with present Canadian (CCREM 1987) and South African (DWAF 1996b) guidelines for aluminium in livestock drinking water of 5 mg/L, with both the Canadian and South African guidelines indicating that much higher levels of aluminium may be tolerated in many instances.

Table 9.3.4 Summary of calculations used to develop a trigger value for aluminium in drinking water for a range of livestock

Animal	Quantity of element[a] (mg/kg)	Daily feed intake (kg/day)	Peak water intake (L/day)	Safety factor[b]	Calculated value (mg/L)
Cattle	1200	20	85	10	5.6
Sheep	1215	2.4	11.5	10	5.1
Chickens[c]	486	0.15	0.4	10	3.6

a From summary of toxic responses of animals to levels of aluminium given in feed in NRC (1980).
b Safety factor for possible long-term effects and tissue accumulation.
c All poultry have a very similar body weight/water intake ratio, hence these values can be taken to represent all poultry.

9.3.5.2 Arsenic

A concentration of total arsenic in drinking water for livestock exceeding 0.5 mg/L may be hazardous to stock health. If arsenic is not provided as a food additive and natural levels of arsenic in the diet are low, a level of 5 mg/L in drinking water may be tolerated.

Source

Arsenic occurs naturally in surface waters at low concentrations, generally <0.01 mg/L. Higher concentrations are found in some groundwaters and as a result of mining or industrial activities (Fergusson 1990, Galvin 1996).

Arsenic is used in a number of industrial processes. It is no longer used as an insecticide in sheep dips but organic forms of arsenic are included in certain herbicide formulations (Hamilton & Haydon 1996). Organic arsenic compounds are sometimes used as feed additives to enhance growth in pigs and poultry (Gough et al. 1979).

Animal health

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). NRC (1980) suggested a maximum tolerable dietary level for livestock of 50 mg/kg in feed for inorganic forms and 100 mg/kg for organic forms of arsenic.

Acute effects such as diarrhoea, loss of coordination and anaemia are symptoms of arsenic intoxication. Non-ruminants (pigs and poultry) are more susceptible than ruminants and horses. Although the level of arsenic in animal tissue increases proportionally with the amount ingested, it does not accumulate in tissue and is efficiently excreted (NRC 1980).

Derivation of trigger value

The ANZECC (1992) guideline of 0.5 mg As/L has been retained in the absence of any new contradictory information and is consistent with the present Canadian guideline for arsenic in livestock drinking water (CCREM 1987). Recent South African guidelines suggest that arsenic concentrations less than 1.0 mg/L are unlikely to cause adverse effects on animal health, but long-term exposure to concentrations >1.5 mg As/L may be harmful to sensitive species such as pigs and poultry (DWAF 1996b).

9.3.5.3 Beryllium

There are insufficient data to set trigger values for animal consumption of beryllium in livestock drinking water.

Source

Beryllium may be present in water supplies through the weathering of rocks containing feldspars or it may be deposited from the atmosphere, predominantly as a result of burning fossil fuels. The concentration of beryllium in freshwaters is usually <1 μg/L (Galvin 1996).

Animal health

Beryllium is generally poorly absorbed from the gastrointestinal tract, and toxicity due to ingestion is low (WHO 1984). 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 & Mitchener 1975 a,b). In another study, rats fed with beryllium in the diet at levels of 5 mg/kg, 50 mg/kg and 500 mg/kg of feed, showed no evidence of carcinogenic response related to beryllium (WHO 1984).

In a review of the limited amount of toxicity data available for animals, IPCS (1990) indicated that ingestion of beryllium in the water supply for long periods of time caused no ill effects.

Derivation of trigger value

The data presently available are insufficient and inconclusive. Derivation of a trigger value should be deferred until more data become available.

9.3.5.4 Boron

If the concentration of boron in water exceeds 5 mg/L, the total boron content of the livestock diet should be investigated. Higher concentrations in water may be tolerated for short periods of time.

Source

Boron concentrations in unpolluted waters are generally <0.1 mg/L (Galvin 1996). Boron concentrations in groundwater may be higher, although are normally <4 mg/L (Hart 1974). Pesticides and fertilisers containing boron are a potential source of contamination of farm water supplies.

Animal health

Boron dissolved in water or contained in food is rapidly absorbed from the gastrointestinal tract in animals and excreted via the urine.

Green and Weeth (1977) reported that boron concentrations of 150 mg/L in drinking water for cattle resulted in reduced hay consumption and a loss of weight. The tolerance concentration of boron was estimated to be between 40 mg/L and 150 mg/L. NRC (1980) suggested a maximum tolerable level of 150 mg B/kg (as borax) in the diet of cattle, and presumed that this value should be reasonable for other species of livestock.

Derivation of trigger value

The following calculations and assumptions, based on the principles adopted by the World Health Organization (Albanus et al. 1989, cited by Hamilton & Haydon 1996), were used to derive a guideline value. Based on this approach, guideline values for various types of livestock range from 5.8 to 11.3 mg B/L (table 9.3.5).

Table 9.3.5 Summary of calculations used to develop a guideline for boron in livestock drinking water

Animal	Body weight (kg)	Peak water intake (L/day)	Peak food intake (kg/day)	Calculated value (mg/L)
Cattle	150	85	20	7
Pigs	110	15	2.9	5.8
Sheep	100	11.5	2.4	6.2
Chickens[a]	2.8	0.4	0.15	11.3
Horses	600	70	20	8.6

a All poultry have a very similar body weight/water intake ratio; hence these values can be taken to represent all poultry

For cattle:

$$\text{trigger value} = \frac{\text{MTDL x daily feed intake x proportion from water}}{\text{max daily water intake}} = \frac{150\text{mg/kg/day x 20kg x 0.2}}{85\text{L/day}} = 7\ \text{mg/L} \qquad (9.54)$$

where:

MTDL is the suggested maximum total dietary level of 150 mg/kg B in the animal diet (NRC 1980);

20 kg/day is an estimate of the average food consumption of cattle at this weight assuming they consume about 2.5% their bodyweight in feed;

0.2 is the proportion of boron attributed to the intake of water;

85 L/day is the peak consumption rate of water by cattle.

Note that a safety factor for possible long-term effects was not included in the calculations because it is considered that there is little likelihood of there being long-term effects due to boron ingestion (NRC 1980).

A value of 5 mg/L has been proposed for livestock use in both Canada (CCREM 1987) and South Africa (DWAF 1996b) and although somewhat contrary to evidence in Green and Weeth (1977), the values calculated here tend to support this value. It is likely, however, that stock would tolerate much higher levels if the feed concentration of boron was low or for short periods of time (NRC 1980).

9.3.5.5 Cadmium

A concentration of total cadmium greater than 0.01 mg/L in drinking water for livestock may be hazardous to animal health.

Source

Cadmium concentrations in surface waters are usually extremely low (<0.001 mg/L). In unpolluted streams the cadmium occurs predominantly in association with suspended particulate matter, rather than in the dissolved state. Concentrations of cadmium in groundwaters may be slightly higher in some areas (Fergusson 1990). The solubility of cadmium in water increases with decreasing pH. Industrial waste waters, metallurgical industries and fertilisers which contain cadmium as an impurity can be sources of cadmium released into the environment. Corrosion of galvanised tanks and pipes and solders can contaminate water supplies with cadmium.

Animal health

Usually only a small amount of the total cadmium intake by livestock comes from drinking water, with most coming from food. Nevertheless, cadmium concentrations in drinking water for livestock should be restricted because of its toxic and possibly teratogenic, mutagenic and carcinogenic effects (CCREM 1987, CCME 1996).

Miller (1971) reported that only a small part of the ingested cadmium in ruminants was absorbed, with most absorbed cadmium going to the kidney and liver. Taking into consideration the accumulation in liver and kidney and long-term exposure, NRC (1980) suggested a concentration of 0.5 mg/kg as the maximum tolerable dietary intake.

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.

Derivation of trigger value

The ANZECC (1992) guideline for cadmium (based on Hart 1982) has been retained until more information becomes available from animal feeding trials. The guideline value of

0.01 mg/L is consistent with guidelines developed for cadmium in South Africa (DWAF 1996b); a value of 0.08 mg/L has been proposed in Canada (CCME 1996).

9.3.5.6 Chromium

Levels of total chromium exceeding 1 mg/L in the drinking water of livestock may be hazardous to animal health.

Source

Chromium occurs in the environment in two forms; as trivalent chromium, chromium (III), and hexavalent chromium, chromium (VI). Total chromium concentrations in natural unpolluted waters are generally very low (<0.025 mg/L, Galvin 1996). Chromium may enter water supplies through the waste discharge of a range of industrial processes in which it is used.

Animal health

Trivalent chromium is an essential element in the diet of mammals, being required for carbohydrate and lipid metabolism. Salts of chromium (III) are poorly absorbed by the gastrointestinal tract, whereas the absorption rate of chromium (VI) is much higher. Chromium (VI) is much more toxic to animals than chromium (III) (WHO 1984, NRC 1980, CCREM 1987).

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 tissue accumulation of chromium, 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.

Derivation of trigger value

The ANZECC (1992) guideline for chromium has been retained until more information becomes available from animal feeding trials. The trigger value of 1 mg/L is consistent with guidelines developed for chromium in Canada (CCREM 1987); while in South Africa a guideline value of 1 mg/L chromium (VI) has been proposed (DWAF 1996b).

9.3.5.7 Cobalt

Levels of total cobalt in drinking water for livestock exceeding 1 mg/L may be hazardous to animal health, particularly if cobalt supplements are being used.

Source

Cobalt normally occurs in natural waters at levels well below 0.01 mg/L and in most cases below 0.001 mg/L, but may be higher in some wastewaters (Galvin 1996, APHA, AWWA & WEF 1998).

Animal health

Cobalt is an essential element in the diet of animals, and is important in several enzyme systems, particularly as a component of vitamin B12. Generally cobalt has a low toxicity to animals and in ruminants, cobalt deficiency, in practice, is more likely to occur (NRC 1980).

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 (1987), drinking water for calves would have to contain at least 10 mg/L cobalt

before the symptoms observed by Underwood would be evident. Pigs, cattle and poultry may tolerate cobalt at concentrations of 10 mg/kg in their diet, which is about 100 times normal requirements (NRC 1980).

Derivation of trigger value

The ANZECC (1992) guideline for cobalt has been retained until more information becomes available from animal feeding trials. The guideline value of 1 mg/L is consistent with guidelines developed for cobalt in Canada (CCREM 1987) and South Africa (DWAF 1996b).

9.3.5.8 Copper

Concentrations of total copper in drinking water for livestock exceeding 0.5 mg/L may be hazardous to the health of sheep. Adverse effects may be experienced in cattle at concentrations above 1 mg/L copper, and in pigs and poultry concentrations exceeding 5 mg/L. If animal diets are high in copper, the levels in drinking water should be revised downwards. Animal intake of sulfur and molybdenum should also be considered in conjunction with copper.

Source

Copper is generally found in natural waters at concentrations much less than 1 mg/L, often in association with organic compounds (Galvin 1996). However, concentrations in groundwater as high as 12 mg/L have been reported (Hart 1982). Copper concentrations in water supplies can be elevated as a result of copper-based algicide treatment or corrosion of copper and brass fittings in waters of low pH.

Animal health

Copper is an essential element in the animal diet. Copper nutrition in animals is influenced by the dietary intake of molybdenum, iron and sulfur (see Section 9.3.5.14 for molybdenum). Copper deficiency can result in morbidity and, in some cases, death (NAS 1977b). Cattle given water with 2.5–5 mg Cu/L added were prevented from developing seasonal decline in plasma copper levels and showed no ill effects (Humphries et al. 1983). Copper nutrition in animals is influenced by the dietary intake of molybdenum, iron and sulfur (see Section 9.3.5.14 on molybdenum).

Excessive intake of copper can lead to copper toxicosis in livestock, which generally would be expected to relate to a high intake from feed rather than from water. Initially, copper accumulates in the liver of animals and may cause some reduction in growth. Chronic and acute effects such as liver damage and haemolytic jaundice can occur with extended exposure to high levels of copper. The tendency of copper to accumulate in the liver has potential implications for the health of consumers.

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 1987). Sheep are particularly sensitive to copper. Demayo and Taylor (1981), who reviewed maximum levels of dietary copper intake by livestock, suggested that, to avoid toxicosis, the maximum 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.

Derivation of trigger value

The ANZECC (1992) guideline for sheep has been retained at 0.5 mg Cu/L, which is consistent with present guidelines proposed for use in Canada (CCREM 1987) and South Africa (DWAF 1996b). Trigger values for pigs and poultry of 5 mg Cu/L, and for cattle 1 mg Cu/L, are consistent with current Canadian (CCREM 1987) and South African (DWAF 1996b) guidelines and take into account the relatively greater susceptibility of cattle to copper toxicity. In all cases the trigger values should be revised downwards if the total intake of copper by stock is high.

Further information is needed from animal feeding trials before more definitive guidelines for copper in livestock drinking water can be set.

9.3.5.9 Fluoride

Fluoride concentrations greater than 2 mg/L in drinking water for livestock may be hazardous to animal health. If livestock feed contains fluoride, the trigger value should be reduced to 1.0 mg/L.

Source

Unpolluted surface waters generally contain low concentrations of fluoride but concentrations in groundwater may be higher in some areas. For example, groundwater at Carnarvon, Western Australia, contains fluoride at concentrations up to 5 mg/L (Hart 1974). Groundwater fluoride concentrations >2 mg/L have been reported at several locations in Queensland, mainly in the Great Artesian Basin, with a few cases showing concentrations >10 mg/L fluoride (Gill 1986).

Animal health

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).

The diet may be another source of excessive ingestion of fluoride if the vegetation is contaminated by aerial deposition in industrial areas (NAS 1971), but no toxic effects were reported 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) showed that levels of fluoride >5 mg/L in drinking water adversely affected breeding efficiency in cattle. Moreover, Hibbs and Thilsted (1983) reported erosion of teeth at concentrations of 3.3 mg/L. Experiments with laying hens showed a significant reduction in egg production for hens receiving 6 and 20 mg/L sodium fluoride (2.7 and 9 mg/L fluoride) in their drinking water but that successful production could continue with concentrations up to 14 mg/L sodium fluoride (6.3 mg/L fluoride) (Coetzee et al. 1997).

The risk of fluorosis in either sheep or cattle may be avoided if sufficient water of low fluoride concentration (e.g. surface water) is available and paddocks arranged so that young stock have access only to fluoride-free water for the first three years of life. Where only limited quantities of low-fluoride water are available, the damage from fluorosis will be minimal if young stock are exposed to fluoride-enriched water for no more than three months at a time and then kept for at least three months on low-fluoride water. Control measures are less important in good seasons when stock receive the bulk of their fluid requirements from pasture.

The fluoride concentration in water is rapidly increased by evaporation. This is particularly evident in flowing bores where the water is reticulated through shallow bore drains. As a temporary measure while paddocks are being arranged so that young stock may be kept on low-fluoride water, it is important that the young stock should be watered as near to the bore head as possible.

Derivation of trigger value

The ANZECC (1992) guideline for fluoride has been retained in the absence of any new contradictory information. The trigger value of 2 mg/L is consistent with guidelines developed for fluoride in Canada (CCREM 1987) and South Africa, although the South African guidelines suggest that adverse effects are unlikely to occur in ruminants at concentrations less than 4 mg F/L (DWAF 1996b).

9.3.5.10 Iron

No guideline has been established for iron in drinking water for livestock as it poses a very low health risk to animals.

Source

Iron occurs naturally in water through dissolution of iron-bearing rock and minerals. It is present in waters as soluble Fe^{2+} ions or in the much less soluble Fe^{3+} form. In aerated surface waters iron concentration is usually <1 mg/L. Groundwaters rich in dissolved carbon dioxide and poorly oxygenated have been reported to have a total iron content of up to 100 mg/L (Galvin 1996, NHMRC & ARMCANZ 1996).

Animal health

Iron is essential to animal life and has a low toxicity, being harmful to livestock only if ingested in large amounts. Coup and Campbell (1964) reported slight scouring and blackening of the faeces after administering a daily dose of 30 g iron as ferric hydroxide. At a dosage of 60 g/day, scouring and blackening were pronounced and associated with a decline in bodyweight, reduced milk and fat yield and a general worsening in the condition of the coat. No adverse effects were reported from a dosage of 15 g iron/day.

Iron-contaminated water does not contain enough iron to cause the abovementioned problems, but toxic effects have been reported when cows were grazed on pastures heavily irrigated with groundwater containing 17 mg Fe/L (Hart 1974).

Derivation of trigger value

No trigger value for iron is recommended since water sources generally do not usually contain enough iron to cause health problems in livestock. There is no guideline recommended for iron in livestock drinking water in Canada (CCREM 1987). A guideline value of 10 mg/L has been tentatively proposed in South Africa, although it was noted that adverse effects of excessive iron intake have not yet been well documented in that country and concentrations up to 50 mg Fe/L may be tolerated in many situations (DWAF 1996b).

9.3.5.11 Lead

Concentrations of total lead in drinking water for livestock exceeding 0.1 mg/L may be hazardous to animal health.

Source

Dissolved lead concentrations in unpolluted freshwaters are generally <0.01 mg/L (Fergusson 1990, Galvin 1996), and over 90% of lead transported by unpolluted streams is associated with suspended particulate matter (Salomons & Förstner 1984).

Animal health

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). Chronic effects such as anorexia and respiratory distress are associated with low level poisoning. Severe poisoning causes acute effects such as frothing at the mouth, uncoordination and convulsions (DWAF 1996b).

Hammond and Aronson (1964) suggested that daily ingestion of 6–7 mg Pb/kg bodyweight is the minimum dose that causes poisoning to cattle. Calves were killed by accidental exposure to an estimated dose of 5–8 mg Pb/kg/d for 30 days (Osweiler & Ruhr 1978). Sheep deaths were reported following dietary exposure to 5.7 mg Pb/kg bodyweight/day (James et al. 1966). Horses have been reported to be both more sensitive to lead poisoning than cattle and sheep (CCREM 1987) and less sensitive (DWAF 1996b). In one case, chronic poisoning occurred after horses received 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 reported following low-level intake of lead (Hemphill et al. 1971).

A maximum tolerable dietary level of lead for all animals of 30 mg/kg was suggested by NRC (1980) in a summary of available toxicological data. At high dosage rates lead can accumulate in soft tissues of animals to a degree which might exceed acceptable levels for human consumption if livestock are raised in areas contaminated with Pb (NRC 1980).

Derivation of trigger value

The ANZECC (1992) guideline for lead has been retained in the absence of any new contradictory information. The trigger value of 0.1 mg/L is consistent with guidelines developed for lead in Canada (CCREM 1987) and South Africa, although the latter guidelines suggest that for pigs, no adverse effects are likely to occur at concentrations up to 0.5 mg Pb/L (DWAF 1996b).

9.3.5.12 Manganese

No guideline has been established for manganese in drinking water for livestock.

Source

Manganese occurs in water in several ionic states; Mn^{2+}, Mn^{4+} and Mn^{7+}, of which the divalent compounds are soluble. Unpolluted surface waters usually have low concentrations of manganese (0.001–0.6 mg/L), as contact with air rapidly oxidises the divalent compounds resulting in the precipitation of the insoluble Mn^{4+} compounds. Similarly to iron, manganese can be found in dissolved and colloidal forms, as well as complexed with organic matter.

Higher concentrations of manganese may be found under anoxic conditions (which may occur in groundwater or the lower strata of deep dams and lakes) particularly if the pH of the water is low (Galvin 1996, NHMRC & ARMCANZ 1996).

Animal health

Manganese is an essential element for animal nutrition, but only about 3% of ingested manganese is absorbed. Manganese has low toxicity unless ingested in large amounts (NRC 1980).

Derivation of trigger value

No trigger value for manganese is proposed as there is little information to indicate that manganese concentrations high enough to cause any adverse health effects are likely to be found in waters used for livestock drinking purposes. This is consistent with present Canadian guidelines (CCREM 1987). Recent South African guidelines (DWAF 1996b) recommend an upper limit of 10 mg Mn/L in livestock drinking water, and suggest the possibility of adverse chronic effects such as weight loss and anaemia at higher concentrations.

9.3.5.13 Mercury

Levels of total mercury exceeding 0.002 mg/L in drinking water for livestock may accumulate in edible animal tissue to a level which may pose a human health risk.

Source

The concentration of mercury found in unpolluted streams and groundwaters is generally well below 0.001 mg/L (Fergusson 1990, Galvin 1996). Contamination through industrial emissions and spills can elevate mercury levels. Mercury is also used in certain pesticide formulations.

Organic compounds of mercury, particularly methylmercury, are more bioavailable and more toxic than the inorganic salts, many of which are insoluble. However, inorganic salts of mercury in sediments can enter the food chain through biological conversion to organic forms (Hart 1982).

Animal health

The toxicity of mercury depends on its chemical form, with alkylmercury compounds, particularly methylmercury, being the most toxic due to its greater absorption rate and increased retention in the body of animals. Ingestion of feed is the predominant path of animal exposure to mercury. Symptoms of mercury poisoning in animals vary with the chemical form of mercury, amount ingested and route of intake (Hart 1982).

Signs of mercury 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 27 days; sheep accumulated 120–210 mg/kg under the same conditions (Palmer et al. 1973).

Chronic mercury poisoning in animals results in loss of appetite, with consequent weight loss leading to possible hair loss, anal lesions and paralysis. Severe poisoning results in nervous system disorders (such as lack of coordination, tetanic spasms, convulsions) and is usually fatal.

Ingestion of inorganic mercury by animals results in the accumulation of mercury primarily in the kidney and liver, whereas methylmercury is more evenly distributed through all tissues (NRCC 1979).

Derivation of trigger value

In establishing guidelines for mercury in drinking water for livestock, consideration must be given to both the toxic effects of mercury on animals and its possible accumulation in animal

tissues used for human consumption. Reeder et al. (1979) suggested that drinking water 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 0.003 mg/L, assuming a maximum concentration of 0.2 mg/kg in edible animal tissue. Hart (1982) suggested a value of 0.002 mg/L as more appropriate under Australian conditions.

The ANZECC (1992) guideline for mercury of 0.002 mg/L has been retained in the absence of any new contradictory information. The guideline value developed for mercury in Canada is 0.003 mg/L (CCREM 1987) and in South Africa, 0.001 mg/L (DWAF 1996b).

9.3.5.14 Molybdenum

Concentrations of molybdenum in livestock drinking water greater than 0.15 mg/L may cause health problems to stock, depending on total dietary intakes of molybdenum, copper, iron and sulfur. At molybdenum concentrations greater than 0.15 mg/L, the animal diet should be investigated to ensure that copper levels are sufficient to account for the total dietary intake of molybdenum.

Source

Molybdenum is usually found at concentrations of 0.05 mg/L or less in natural waters (Galvin 1996). Higher concentrations are generally associated with human activities such as mining, industry fallout and chemical fertilisation. The predominant ion is molybdate which is more soluble at higher pH (Cotton & Wilkinson 1972).

Health effects on stock are more likely to occur through the ingestion of forages which can accumulate and hence concentrate molybdenum, than through the intake of water. The level of molybdenum in plants reflects the level in the soils in which they are grown. High concentrations of molybdenum in plants may occur where soils are enriched with molybdenum (e.g. from fertilisers) but can occur naturally, particularly when soils are of neutral to high pH, are very moist and have a high organic content, such as peats and mucks (NRC 1980, 1988, 1996, Jones et al. 1994). Pastures containing high molybdenum levels have been found on calcareous soils in southern Australia (McFarlane et al. 1990).

Animal health

Molybdenum is an essential element in animal nutrition. It is associated with various enzyme systems and seems to be of most importance during early foetal development. There is little information on molybdenum requirements of domestic animals but levels in the diet of <0.02 mg/kg for chicks and around 0.01 mg/kg for sheep have been suggested by Mills and Davis (1987) (cited by Jones et al. 1994).

Ruminants are most susceptible to elevated levels of molybdenum with cattle more sensitive than sheep (NRC 1980, Jones et al. 1994). Molybdenosis ('teart' disease or 'peat scours' in New Zealand) in cattle is characterised by severe scouring and loss of condition, and secondary copper deficiency. Inorganic molybdenum combines with sulfide in the rumen to form thiomolybdates, which bind copper and interfere with its absorption. This increases the animal's requirement for copper and raises its tolerance level to copper. The condition can be treated by adding sufficient copper to the diet. Low dietary copper levels will result in a lesser amount of molybdenum being toxic (NRC 1980, 1988, 1996, Jones et al. 1994).

Other effects of excessive molybdenum intake in ruminants other than those attributed to copper deficiency have been suggested, such as infertility, increased age at puberty, testicular damage and disorders of phosphorus metabolism that produce skeletal abnormalities and cause lameness. Concentrations as low as 5 mg Mo/kg feed have been reported to cause infertility effects such as increased age at puberty and reduced conception rate (Phillipo et al. 1987, cited by Jones et al. 1994 and NRC 1996).

Levels of 5–6 mg Mo/kg in the diets of cattle have resulted in copper deficiency, depending on the level of copper in the diet and the period of exposure (NRC 1980, 1996). The National Research Council (1980) has estimated a maximum tolerable level of 10 mg/kg in the diet of cattle and sheep for short-term intake. In a survey of copper deficiencies in herds in South Australia, McFarlane et al. (1990) observed that the risk of copper deficiency is associated with moderate concentrations of molybdenum, sulfur and iron in pasture, rather than low copper levels; and that copper from these pastures would rarely meet the requirements of cattle when there are levels of molybdenum >2 mg/kg.

In non-ruminant species the Mo-Cu antagonism only occurs with lower gut sulfide generation associated with high sulfur intake (as inorganic sulfur or in high protein feed). Molybdenum seems to be rapidly absorbed and excreted by pigs which makes them extremely tolerant of high levels of intake. Pigs fed diets containing up to 1000 mg Mo/kg for three months have shown no ill effects. Poultry appear to be more sensitive to molybdenum and levels in the diet of 200 mg/kg have resulted in reduced growth (NRC 1980, Mills & Davis 1987, cited by Jones et al. 1994).

The type of diet may also influence animal tolerance of molybdenum. In dry forages molybdenum may not be as available as it is in green feed, possibly due to the availability of soluble sulfur containing proteins. Ratios of copper:molybdenum in animal feeds of 2:1 and 4:1 have been reported to prevent copper deficiency (NRC 1988, 1996).

Derivation of trigger value

The following calculations and assumptions, based on the principles adopted by the World Health Organization (Albanus et al. 1989, cited by Hamilton & Haydon 1996) were used to derive a trigger value. Based on this approach, a trigger value of 0.15 mg/L was derived for molybdenum in drinking water for both cattle and sheep (table 9.3.6).

Table 9.3.6 Summary of calculations used to develop a trigger value for molybdenum in livestock drinking water

Animal	Quantity of element (mg/kg)	Daily feed intake (kg/day)	Peak water intake (L/day)	Safety factor[a]	Calculated value (mg/L)
Cattle	10	20	85	3	0.15
Sheep	10	2.4	11.5	3	0.15

a For possible long-term effects

For cattle:

$$\text{trigger value} = \frac{\text{MTDL x daily feed intake x proportion from water}}{\text{max daily water intake x safety factor}} = \frac{10\,\text{mg/kg/day x } 20\,\text{kg } 0.2}{85\,\text{L/day x } 3} = 0.15\,\text{mg/L} \qquad (9.55)$$

where:

MTDL is the suggested short-term maximum total dietary level of molybdenum in feed of 10 mg/kg (NRC 1980);

20 kg/day is an estimate of the average food consumption by cattle at this weight assuming consumption of about 2.5% of bodyweight in feed;

0.2 is the proportion of molybdenum attributed to water intake;

85 L/day is the peak rate of water consumption by cattle; and

3 is the safety factor for possible long-term effects.

As cattle and sheep (ruminants) appear to be most sensitive to molybdenum this value can be used as a guide for other livestock. However, the levels of copper, iron and sulfur in the diet and the type of pasture may greatly influence animal tolerance of molybdenum. Animals may tolerate concentrations of molybdenum in water considerably higher than the guideline value provided dietary levels of copper are adequate to compensate for the high level of Mo.

The guideline recommended in South Africa for molybdenum in livestock drinking water is 0.01 mg/L, with concentrations <0.02 mg/L considered likely to be tolerated provided copper and sulfur intakes are adequate (DWAF 1996b). Canadian guidelines recommend an upper limit of 0.5 mg Mo/L in livestock drinking water (CCREM 1987).

9.3.5.15 Nickel

Concentrations of total nickel in livestock drinking water greater than 1 mg/L may have adverse effects on animal health.

Source

The concentration of nickel in natural waters is usually below 0.01 mg/L unless contaminated by industrial waste, fallout from burning fossil fuels or the corrosion of nickel-plated plumbing fittings (NHMRC & ARMCANZ 1996, Galvin 1996).

Animal health

Nickel is an essential element in animal nutrition and is considered to have low toxicity (NRCC 1981). 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 of nickel deficiency on reproduction in pigs have been reported (Nielsen & Ollerich 1974, Anke et al. 1974).

Growth reduction in calves was induced by adding nickel salts to the diet at concentrations of 250 mg Ni/kg (O'Dell et al. 1970). A concentration of 5 mg Ni/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 Ni/L in the drinking water of rats through three generations resulted in increased peri-natal mortality and an increased number of runts (Schroeder & Mitchener 1971).

Derivation of trigger value

The ANZECC (1992) guideline for nickel has been retained until more information becomes available. The trigger value of 1 mg/L is consistent with guidelines developed for nickel in Canada (CCREM 1987) and South Africa (DWAF 1996b).

9.3.5.16 Selenium

Concentrations of total selenium in drinking water for livestock exceeding 0.02 mg/L may be hazardous to stock health.

Source

Selenium occurs in the environment in association with metal sulfides and is derived from igneous rocks (Ehrlich 1990). In surface waters selenium is generally present at concentrations below 0.01 mg/L, although groundwaters have been reported to contain up to 1 mg Se/L, usually in association with areas of volcanic activity (Galvin 1996). Selenium can be released into the environment through the burning of coal and as a discharge from the processing of sulfide ores (NHMRC & ARMCANZ 1996).

Animal health

Selenium is an essential element for animal nutrition. Diets containing less than 0.02–0.04 mg Se/kg can result in deficiency symptoms in cattle, sheep, pigs and poultry (Oldfield et al. 1974, Underwood 1977).

At elevated concentrations selenium is toxic to animals. 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 S selenium-rich soil (Johnson 1976). The chronic symptoms of selenium poisoning (Alkali Disease) include loss of hair, lameness and a decrease in food intake, which may result in death by starvation. The symptoms of acute selenium poisoning include stumbling, difficulty breathing, diarrhoea and bloat, with death resulting from respiratory failure (NRC 1980).

In lactating animals, an additional problem is the transmission of selenium to the milk, forming selenomethionine proteins. Milk from cows in areas where selenium poisoning occurred was reported to have contained 0.3–1.2 mg Se/L; normal concentrations range from 0.003–0.007 mg/L (Underwood 1971).

Derivation of trigger value

In the absence of any new contradictory information the existing guideline (ANZECC 1992) has been is retained. Recent guidelines developed in Canada (CCREM 1987) and South Africa (DWAF 1996b) recommend an upper limit of 0.05 mg/L.

9.3.5.17 Uranium

Concentrations of uranium less than 0.2 mg/L in livestock drinking water are unlikely to be harmful to animal health.

Source

Uranium may be found in natural waters, particularly groundwaters and may be the result of natural processes or may arise from mineral processing.

Animal health

According to Garner (1963), the minimum concentration of uranium found to cause poisoning was 50 mg/d for sheep and 400 mg/d for cattle. Phosphorus supplements fed to dairy cattle may contribute 16 mg/d uranium, depending on the source of phosphorus (Reid et al. 1977).

Derivation of trigger value

CCREM (1987) developed a guideline value of 0.2 mg U/L in livestock drinking water by the inclusion of a safety factor, estimation of allowable intake of urganium through water and the volume of water animals drink based on the above level for cattle. A concentration of 0.2 mg U/L in stock drinking water is recommended as an interim trigger value until further

information from animal feeding trials becomes available. For information on radiological quality concerning uranium (and other radionuclides) see Section 9.2.8.

9.3.5.18 Vanadium

Insufficient information is available to set a trigger value for vanadium in livestock drinking water.

Source

Vanadium salts are soluble in water and do not normally adsorb onto clay particles. Vanadium compounds are used as catalysts in many industrial processes. The concentration of vanadium in natural waters is usually less than 0.001 mg/L (DWAF 1996b).

Animal health

Some experiences with rats and chicks suggest that vanadium is required for lipid, tooth and bone metabolism (Hopkins & Mohr 1971). Concentrations of 2 mg V/L (as NH_4VO_3) in drinking water improved the development of growing chicks. According to Van Zinderen Bakker and Javorski (1980), reduced growth rate resulted when chickens and rats were given diets containing 13 mg V/kg and 25 mg V/kg respectively.

Derivation of trigger value

Present information is inconclusive regarding the effects of vanadium levels in drinking water on animal health. No guideline is recommended until further information from animal feeding trials becomes available.

The ANZECC (1992) guidelines gave an upper limit for vanadium for all forms of livestock of 0.1 mg/L but this seems contradictory to some of the evidence given above. Present Canadian Water Quality Guidelines (CCREM 1987) give the same guideline value for vanadium in livestock drinking water; while in South Africa, an upper limit of 1 mg V/L is proposed, with some adverse effects considered likely to occur at higher concentrations (DWAF 1996b).

9.3.5.19 Zinc

Total zinc concentrations in livestock drinking water less than 20 mg/L are unlikely to pose a threat to animal health.

Source

Concentrations of zinc rarely exceed 0.01 mg/L in natural waters (Galvin 1996). Higher concentrations in waters can be associated with pollution from industrial wastes (Hart 1982) or corrosion of zinc coated plumbing or galvanised iron water tanks, particularly at low pH (NHMRC & ARMCANZ 1996).

Animal health

Zinc is an essential element in the animal diet 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). Requirements for zinc range from 50 mg/kg to 100 mg/kg in the 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, 1000 mg/kg for chicks, pigs and mature cattle, and 2000 mg/kg for turkeys. NRC (1980)

proposed maximum tolerable levels of zinc of 500 mg/kg for cattle, 300 mg/kg for sheep and 1000 mg/kg for pigs and poultry.

Derivation of trigger value

The ANZECC (1992) guideline for zinc (based on Hart 1982) has been retained until more information becomes available from animal feeding trials. The trigger value of 20 mg/L is consistent with guidelines developed for zinc in South Africa (DWAF 1996b); a value of 50 mg/L has been proposed in Canada (CCREM 1987).

9.3.6 Pesticides

In the absence of adequate information derived specifically for livestock under Australian and New Zealand conditions, it is recommended that the guidelines set for raw water for drinking water supply be adopted.

Source

The use of pesticides to control insects, pathogens and weeds is an integral part of the economic production of many agricultural commodities. Pesticides are also widely used for weed control along roads, waterways, etc and are sometimes applied in urban areas to control insects such as mosquitoes.

Pesticides are mainly organic compounds, or in some cases organo-metallic compounds, and are categorised according to their intended use: as insecticides (controlling insect pests), herbicides (controlling weeds), fungicides (control of fungal pests) and veterinary medicines (for animal health). Each category of pesticide is often grouped into classes of chemically similar compounds; for example, the organochlorine and organophosphate insecticides, and the phenoxy herbicides (Schofield & Simpson 1996). Pesticides encompass a broad range of natural and synthetic compounds of widely differing chemical composition. All are carefully screened for health and environmental effects prior to registration for use.

Pesticide residues can sometimes be found in surface waters, as a result of: direct application (e.g. for weed control); careless use or disposal of pesticides and their containers; aerial drift and wind erosion; and transport in runoff waters (Hunter 1992, CCREM 1987, Schofield & Simpson 1996). Movement of pesticide residues which bind strongly to soil particles and are relatively insoluble in water occurs mainly through soil erosion processes. Runoff waters may also contain other residues in dissolved form. Leaching of pesticide residues to groundwaters can occur and is dependent on the chemical and physical properties of both the pesticide compound and the soil. Residues of several pesticides, notably the herbicide atrazine, have been found in surveys of some Australian groundwaters, but generally at very low concentrations (Keating et al. 1996, Schofield & Simpson 1996).

Many factors influence the persistence of pesticide residues in aquatic environments, including processes such as decomposition by sunlight, chemical transformation and microbial decomposition. Residues of some persistent organochlorines, such as DDT and dieldrin, can still be found in the environment although they were withdrawn from use or have had restricted use in Australian agriculture for decades (Schofield & Simpson 1996).

Animal health

The organophosphate and carbamate pesticides are relatively toxic to livestock causing symptoms such as diarrhoea, salivation, excessive urination and respiratory and muscle

malfunction. These pesticides break down quite rapidly in the aquatic environment through microbial action (DWAF 1996b).

Most commonly used herbicides are considered not to be highly toxic to mammals (CCREM 1987). Of primary concern is that some pesticides or their metabolites may accumulate in animal tissues or products meant for human consumption at levels which may affect the saleability of these products (DWAF 1996b).

Derivation of guidelines

Information is not yet available on guidelines for pesticide residues in drinking water derived specifically for livestock under Australian and New Zealand conditions. Adoption of the Australian Drinking Water Guidelines (NHMRC & ARMCANZ 1996) should provide a margin of safety for livestock and prevent accumulation of unacceptable pesticide residues in animal products. Additional information can be obtained from guideline values for certain pesticides developed in Canada (CCREM 1987), mainly using data obtained from animal toxicological studies (summarised in table 9.3.7).

Table 9.3.7 Canadian water quality guidelines for pesticides in livestock drinking water[a]

Pesticide	Guideline value µg/L
Insecticides	
Aldicarb	11[b]
Carbofuran	45
Dimethoate	3[b]
Herbicides	
Bromoxynil	11[b]
Cyanazine	10[b]
Dicamba	122
Diclofop-methyl	9
Dinoseb	150
Glyphosate	280
Simazine	10[b]
Tebuthiuron	130[b]
Triallate	230
Trifluralin	45[b]
Fungicides	
Chlorothalonil	170[b]

a From CCREM (1987)

b Toxicological data available only sufficient to produce an interim guideline value

9.3.7 Radiological quality

Please refer to Section 9.2.8. The same trigger values and discussion apply to radiological quality for both irrigation and livestock drinking water uses.

9.3.8 Future information needs for livestock drinking water

In this review we have updated the information that was previously used in determining the guidelines for livestock drinking water (ANZECC 1992). With several notable exceptions, few examples of new studies were found, with most information coming from the 1960s and 1970s.

Two differing approaches have been used in developing guidelines in other countries. A toxicological approach as proposed by the Canadian Council of Ministries of the Environment (CCME 1993) is based on the following principles:

- the method of developing guideline values is transparent and consistent;
- selection criteria and appraisal protocols ensure only valid sound scientific data are used;
- data can be obtained through feeding trials with animals.

Some disadvantages of this approach include:

- the need to make many assumptions on, for example, the value of a 'safety factor' for inter- and intra-species differences, long-term effects, and the contribution of water consumption to total intake of a chemical;
- no account is taken for the risk of animals consuming the contaminants;
- differing climatic conditions, feed types, animal ages and condition are not usually addressed;
- interactions with other elements in the metabolism of animals are not considered;
- users of the guidelines have to interpret the suitability of the water in specific cases.

An alternative is a more 'holistic' approach, as taken by the Department of Water Affairs and Forestry (DWAF 1996b) in developing the South African guidelines. This approach includes the use of in situ observations and studies to identify the level of a constituent at which no adverse effect would be expected, taking into consideration the major synergistic and antagonistic factors affecting the onset of adverse effects. Guidelines are given in the context of a risk-based approach, with an indication given of contaminant levels that might be tolerated for short periods of exposure, or following adaptation to the water source. Where possible, differences among animal species and stages of life are considered.

9.3.8.1 Biological parameters

Detection of pathogens in water supplies is time consuming and expensive. Currently, it is common practice to monitor and control microbiological water quality on the basis of concentration of indicator organisms. The presence of indicator organisms does not always mean that pathogens are present and conversely a lack of these indicators does not mean the water is free of other pathogens. A single bacterial indicator may not be suited to all situations and a combination of organisms may be required to assess the levels of viruses and parasites. The lack of data available on pathogens in livestock water supplies, while making the development of accurate guideline values difficult, may in fact reflect the extent of the problem.

A study being set up in New Zealand by the Ministry for the Environment, Ministry of Agriculture and Ministry of Health has proposed a procedure for developing a risk model from information gained from a pathogen characterisation study at different sites around New Zealand and then refined with data from epidemiological studies on animal health. Although the use of indicator organisms may still be necessary, the risk model may allow for a 'best'

choice of an indicator for different situations and sites. The outcome from this study will act as a model for developing guidelines for freshwater usage.

Work is currently being undertaken into developing guidelines for cyanobacteria and cyanobacterial toxins. A working group set up by ARMCANZ and the NHMRC as part of the National Algal Management Strategy is examining the issues for developing guidelines for, for example, drinking water, recreation, livestock and irrigation use.

9.3.8.2 Pesticides

Emerging issues for agriculture concerning pesticide residues in irrigation waters and drinking water for livestock are not adequately covered in the present guidelines. Accumulation of pesticide residues at detectable levels in plant and animal tissues has implications for animal and human health, as well as potentially serious consequences for Australian and New Zealand agriculture. There are implications for our domestic and export meat and grain trades, in particular. The present guidelines have not been scientifically derived from first principles. Livestock guidelines are based on those for human drinking water, which may well be inappropriate and/or unnecessarily restrictive of farmers' options in providing water for their stock.

Development of guidelines for livestock should be based on estimates of permissible intakes for each pesticide, which can be derived from animal metabolism and animal feeding studies. Each pesticide residue would need to be considered individually, since pesticides cover a very diverse group of compounds with widely differing properties. Each trigger value will need to be derived from an evaluation that takes into account the numerous factors affecting the nature of the residue and likely levels in animal tissues. A comprehensive search of the literature for basic data for deriving each guideline level would be required. Issues include pesticide chemistry, environmental fate of pesticides, daily water intake by animals, likely additional intake of pesticides in food, animal liveweight gains, pesticide metabolism and accumulation in animal tissues. Priority should be given to developing guidelines for residues of those pesticides commonly employed in Australian and New Zealand agriculture that are likely to be found in surface waters and groundwaters used for stock watering.

A risk-based approach is recommended, with the following principles applied in the development of guidelines:

- guidelines must be based on scientific data and information;
- derivation of the guidelines should be fully documented and transparent, with all sources, deductions, extrapolations and conclusions fully explained; and
- studies producing the primary data must be subject to critical review (validity of methods, conclusions, etc).

9.4 Aquaculture and human consumers of aquatic foods

9.4.1 Introduction

This environmental value includes aquaculture as well as human consumers of aquatic foods. The Chapter marks the first occasion in which joint guidelines have been provided for the protection of aquaculture in Australia and New Zealand. These guidelines, which are mostly based on value judgements for acceptable risks, are for influent water quality only. Effluent water quality is not considered in these guidelines as it is dealt with through State and Federal Government legislation and regulations in Australia and through the Resource Management Act and Industry Agreed Implementation Standards in New Zealand.

It is generally agreed that good quality water is the most important input for aquaculture and thus a key element in the success of all phases of culture operations, including hatchery, nursery, growout and holding or transport of live product to market. Poor water quality can adversely effect the development and growth of cultured aquatic organisms and even result in death. As noted by Zweig et al. (1999), it may also degrade the quality of the product by tainting the flavour or by causing accumulation of high enough concentrations of toxic substances to endanger human health.

Some of the guidelines presented here should be used with some caution as they are not based on a critical assessment of a wide data set. Rather they are based on the personal experience of a number of industry specialists (noted as 'pers comm' in the tables; the sources are listed in Appendix 9.1) or are taken from recommendations of 'safe' levels in technical and scientific literature (a discussion of the confidence levels is provided in Section 9.4.1.5).

The Chapter focuses mostly on cultured species of finfish, molluscs and crustaceans, although as detailed in Section 9.4.1.1, a wide range of other aquatic species are cultured including plants, reptiles and invertebrates. The report is in two main parts, the first deals with the growth and survival of culture species, the second deals with residues and contaminants in products for human consumption.

Water quality guidelines are provided in Section 9.4.2 for optimising growth and survival of aquaculture species. These are divided into:

- physico-chemical stressors
- inorganic toxicants
- organic toxicants
- pathogens and biological contaminants

Section 9.4.3 discusses the issues of, and provides guidelines for, the safety for human consumers of aquatic foods. It must be noted that these aquatic foods for human consumption can be sourced through aquaculture as well as recreational (including indigenous fishing) and commercial fisheries. The main difference is that aquaculture products are usually harvested from a partly controlled or carefully selected environments, whereas recreational and commercial fisheries are based upon wild populations of fish, crustacean and mollusc species, which are supported by natural habitats and food webs. Thus to protect wild stocks

of aquatic organisms, it is recommended that the water quality guidelines for the protection and maintenance of aquatic ecosystems (Chapter 3 of Volume 1, and Volume 2) be applied.

As discussed in Section 2.1.3 (Volume 1), the different environmental values are interdependent and the uses within each can have impacts on others. For example, agricultural runoff can often contain contaminants which adversely affect downstream aquaculture or fisheries. Conversely, aquacultural activities can affect environmental values downstream.

A check of the recommendations provided for ecosystem protection often will see lower guidelines than those provided for protection of aquaculture species (Section 9.4.2). The main reason is that the aquaculture species are held in a specific water environment for shorter periods of time, usually less than 12 months, than those wild species which can spend all of their life in the one water body. In fact, the various life cycle stages of aquaculture species may be held in totally separate culture environments (e.g. where the hatchery, nursery and growout facilities are in different locations and changes of water are undertaken regularly). As indicated above, control or selection of the environment is undertaken to reduce risks to the health and survival of the culture species. Furthermore, the cultured organisms are often fed artificial (formulated) or selected diets, reducing the potential for exposure from contaminants in the natural environment. However, these formulated diets could include contaminated ingredients, and so the sources of all constituents of the feeds need to be identified and checked to prevent possible adverse effects of the culture species.

A range of chemicals and therapeutants are used in aquaculture operations for the control of a variety of pathological conditions in the culture organisms. Many therapeutants are administered on veterinary advice, and provided they are used under the appropriate instructions, should not cause problems. Therefore, they are not included in this report, except for brief notes in Section 9.4.3.1. In Australian and New Zealand aquaculture the level of use of the chemicals is much lower than that found in other primary industries, and certainly much lower than the levels of use in overseas aquaculture operations. However, their use can create potential problems; for example, formaldehyde (formalin) is commonly used by prawn farmers to control algal blooms and reduce gill fouling in concentrations that could be toxic to prawns and humans (Burford, pers comm). Some of these potential contaminants are not included in this report, however, a process of registration is being undertaken by the National Registration Authority (NRA). Readers are advised to consult the NRA web site (www.dpie.gov.au/NRA/index.html).

9.4.1.1 Aquaculture in Australia and New Zealand

Aquaculture involves the production of food (plant and animal food) for human consumption, fry for recreational fishing and natural fisheries, ornamental fish and plants for the aquarium trade, raw materials for energy and biochemicals (algal extracts and pigments), and a number of items for the fashion industry (shell buttons, pearls and fish and crocodile skins).

With wild fisheries approaching maximum sustainable levels and many already being over exploited, aquaculture is increasingly important worldwide as a source of aquatic food and other products.

For the financial year 1997/98, almost 30 700 tonnes of product and around 9.3 million juveniles (mostly finfish fry and ornamental fish), were produced at an estimated farm gate value in excess of $517.4 million (O'Sullivan & Roberts 1999). This represents approximately 25% of total aquatic food production in Australia.

During 1997/98 over 60 species were cultured on a commercial scale, with several other species undergoing pilot or experimental production. The main commercial species included salmonids (5 species), southern bluefin tuna, barramundi, native freshwater fish (at least 10 species), introduced freshwater finfish (2 species), marine fish (at least 4 species), aquarium fish (many species), eels (2 species), freshwater crayfish (3 species), Penaeid prawns (2 species), brine shrimp, mud crabs, freshwater shrimp, freshwater prawns, edible oysters (at least 5 species), pearl oysters (at least 3 species), blue mussels, freshwater mussels, scallops, clams (2 species), abalone (2 main species), trochus (1 species), microalgae (1 species), crocodiles (2 species) and polychaete worms (2 species).

In order of value, the most important sectors were pearl oysters ($229.4 million), southern bluefin tuna ($87.2 million), salmonids ($82.7 million), edible oysters ($47.9 million) and prawns ($35.4 million). Together these sectors contribute over 90% of total value of production.

Other valuable species included barramundi ($7.0 million), freshwater crayfish ($4.9 million), mussels ($4.1 million), native freshwater fish ($3.9 million), microalgae ($3.0 million), crocodiles ($3.0 million), aquarium fish ($2.8 million), eels ($2.3 million), scallops ($1.2 million), abalone ($1.1 million) brine shrimp ($0.9 million), and aquatic worms ($0.3 million). Other species beginning to move from research to pilot-scale production include marine fish, crabs, freshwater shrimp, and freshwater mussels.

Since 1988/89, there has been almost a 160% increase in the tonnes produced and a 280% increase in the value of this production. It is likely that a moderate rate of increase (10%+) will continue for another few years providing access to sites and venture capital is not limited.

In New Zealand, the main culture species are green shell mussels, Pacific salmon and Pacific oysters. According to data provided by the New Zealand Fishing Industry in 1998 (Maddock pers comm. 1999), annual production of these species was, respectively, approximately 33 203 tonnes (worth NZ$118.2 million), 3841.7 tonnes (NZ$32.1 million) and 1 0342.5 tonnes (NZ$11.5 million). This represents an increase of 6 300 tonnes and a value of NZ$37 million over the previous year. The annual increase in value for the current year is estimated to be approximately 30%. Aquaculture now contributes to over 13% of all New Zealand aquatic food exports (little production is consumed domestically).

A range of other species are being cultured in New Zealand including rock lobsters, scallops, seaweeds, sponges, freshwater shrimp, flatfish and paua (abalone).

9.4.1.2 Relationship between water quality, aquaculture production and human food safety

Aquatic organisms are in such intimate association with their water environment that their performance is strongly influenced by water quality parameters. Schreck and Li (1991) noted that any environmental factor that has a level of toxicity can cause a stress response and reduce the capacity of the cultured organism to grow, resist disease or reproduce.

Aquaculture is often heralded as the ‘farming of the seas’, however, there are several important differences between terrestrial and aquatic farming. Most importantly, aquatic animals maintain a high rate of respiration since less oxygen is present in a given volume of water than in an equal volume of air. This high respiration rate, coupled with a large presence of dissolved substances in water, provides the basis for a greater potential for aquatic organisms to be exposed to toxic substances (Brune & Tomasso 1991).

There is a strong relationship between water quality and product performance. To produce finfish, crustaceans, molluscs and other aquatic animals and plants successfully and efficiently, maintaining the water quality to suit the environmental requirements of the particular culture species is of paramount importance. Appropriate culture conditions, which include optimal water quality, mean:

- good growth, reproduction and survival;
- higher production and market value, reduced costs;
- improved profits.

A number of different production systems are utilised in the aquaculture industry. However, all product containment methods can be placed in any one of three groups based on how the water is sourced:

- in or on the water source (e.g. cages, long lines, racks, bottom culture);
- water is extracted from the source and, via a flow through system, is returned to the water source at a point other than the supply point (e.g. ponds, raceways, tanks);
- water is extracted from the source and then recirculated with treatment so that the water quality is optimised (e.g. tanks, ponds).

Water quality in aquaculture encompasses all the physical, chemical and biological parameters that affect aquaculture production. Appropriate site selection is a key factor in managing many physical and some chemical factors.

Apart from correct site selection, farm management procedures are aimed at improving the biological and in some instances the physical and chemical conditions of the aquaculture water (e.g. through aeration in ponds or tanks).

Aquaculturists have a strong commercial interest in maintaining as close to optimal water quality conditions as possible. However, the aquaculturist also must ensure that a specific water source has a suitable quantity of water for the production of a particular species. *Both water quality and quantity are of utmost importance to aquaculture*.

With respect to ensuring the safety of human consumers of aquatic foods, even if the culture species (or wild fishery stock) was able to grow and thrive in a given water source, low levels of pollutants or biological organisms can cause the products to be contaminated or have off-flavour.

As described by Zweig et al. (1999), the process by which pollutants concentrate in aquatic foods is called bioaccumulation. Entry of pollutants into an aquatic organism can be through the gills, the gut, or by direct exposure to the skin. Many pollutants, especially those which are fat soluble, collect in the tissues of aquatic organisms. This process results in higher concentrations of pollutants in body tissues of aquatic organisms than in the surrounding water. This can produce a potential health risk in human consumers of these organisms.

In Section 4.4.5, the relationship between contaminant concentration in source water and/or tissues of food species with the protection of human consumers is discussed. A model is described which shows the relationship of contaminants in culture feeds and/or source waters with human food residues. The model provides a means of predicting contaminant concentrations in the final aquaculture product given the concentrations in culture feeds and source waters.

Another problem is off-flavour or tainting which occurs when certain pollutants, such as petroleum hydrocarbons or metals, accumulate in aquatic organisms to a level at which the flavour is affected, making the product undesirable for human consumption (Zweig et al. 1999). This is discussed in Section 9.4.3.3.

9.4.1.3 Philosophy behind setting the water quality guidelines

The objective was to develop a set of water quality guidelines that would:

- promote the quality of water necessary for use by the aquaculture industry;
- protect human consumers of harvested aquatic food species.

1. Aquaculture guidelines

No comprehensive compilation of water quality guidelines for the protection of aquaculture species has been available in Australia. Most aquaculturists have relied on documents outlining general practices for specific species, often depending on their own experiences and use of qualitative information. According to Busby (pers comm), the situation is different in New Zealand where IAIS 005.1 (Industry Agreed Implementation Standards — which is law pursuant to the *Meat Act 1981*) has clear requirements on the mandatory water quality requirements for aquaculture. This standard has been successfully used in court hearings regarding abuse of water quality.

The water quality guidelines provided here will be of great benefit to the aquaculture industry in Australia and New Zealand. They have been developed to assist water resource managers to maintain an appropriate level of water quality where aquaculture activities exist, or may exist in the future. Farmers will now have a scientifically determined set of water quality targets which are designed to protect the quality of their culture waters. As well, the guidelines provide a quick reference guide for industry and researchers to ensure the quality of the source water.

They are not intended specifically to regulate activities of the aquaculture industry, although the aquaculturist must be concerned with the potential for downstream impacts on ambient water quality where effluent discharge occurs. The guidelines also should assist in providing a baseline for negotiations between farmers, governments and other relevant groups, and to protect waters used for aquaculture. The guidelines also should assist proponents of new aquaculture ventures to select areas with adequate water quality.

The water quality guidelines will provide the basis for aquaculture management decisions, such as:

- environmental planning and management
- environmental assessment and monitoring requirements
- appropriate environmental zoning and legislation
- appropriate species and suitable site selection
- site capacity
- farm design criteria
- stocking densities
- feeding activity
- production schedules.

It also is recognised that farms can impact on each other. For example, effluent from one farm may become the influent water for another. Thus, the numbers and sizes of farms which can be built in an area may need to be limited. Farms can also impact on other users of the waters, and operators need to comply with a number of government regulations as to the quality of their effluent water.

2. Human food safety guidelines

Standards for chemical contaminants in food for the protection of human consumers of aquatic foods have been set by the Australia New Zealand Food Authority (ANZFA) and are statutory.

ANZFA develops and administers uniform standards for contamination in foods under a treaty between Australia and New Zealand. These standards identify a limit to contamination in food (including aquatic foods) above which is considered injurious to human health, and are measured as concentrations in the flesh of organisms (mg/kg). The standards are listed in the *Food Standards Code* (ANZFA 1996) which is regularly updated for the protection of public health and safety. Unlike the water quality guidelines, food standards are enforceable through legislation.

The relationship between contaminant concentration in water and consequent concentration in the flesh of aquatic organisms is not well known and it has not been possible to provide water quality guidelines that will guarantee that the Australian and New Zealand food standards will be achieved. To provide some guidance to the users of this document the food standards for a number of contaminants are repeated in this Chapter, however, the reader is referred to the *Food Standards Code* which is the authoritative document on this issue. These standards are continually under review and can be examined on the appropriate web sites — www.anzfa.gov.au (Australia) or www.anzfa.govt.nz (New Zealand).

The guidelines provided here will assist in expanding demand for aquatic foods. For example, the reduction or minimisation of exposure to chemical residues, toxins and off-flavour compounds, will improve overall product quality. It is possible that clean waters will be used as a marketing tool, enhancing the ‘green’ image of aquaculture products and the sensory (taste) perceptions which can lead to premium market prices.

9.4.1.4 Approach to deriving water quality guidelines

Based on the approach undertaken in South Africa (DWAF 1996), a list of water quality indicators and contaminants was distributed to industry and researchers to determine the relative level of importance for aquaculture as well as for human food safety. Responses were provided with respect to the likelihood of exposure of adult stock under normal growing conditions in Australia and New Zealand, (i.e. under usual water quality conditions without exposure to major pollution). A wide variety of physico-chemical conditions as well as exposure to inorganic and organic toxicants and pathogenic organisms were considered important.

The guidelines are provided under four main categories:

- physico-chemical stressors
- inorganic toxicants (heavy metals and others)
- organic toxicants (pesticides, detergents, petrochemicals, etc.)
- pathogens and biological contaminants.

1. Scope of the study

The guidelines:

- Are concerned with the quality of water required to carry out aquacultural activities. The quality of effluent discharges are dealt with under a number of State and Federal Government regulations.
- Consider those contaminants, chemicals, elements, microorganisms, toxins, etc, likely to present a problem for aquaculture.
- Are concerned with protecting the health of culture species during the growing period (pre-harvest), but not during those processes (e.g. slaughter, processing, transport, marketing) considered to be post-harvest.
- Consider the effects on adult forms of cultured species, although it is recognised that hatcheries and nurseries also utilise large quantities of water (larval and juvenile stages of the life cycle usually have lower tolerance levels than the adult stages of the life cycle).[1]
- Consider the protection of human consumers of harvested aquatic food species from the toxic effects of contaminants and from tainted flesh. This applies to aquaculture enterprises as well as recreational and commercial harvesting of aquatic food species from natural waters.

2. Methodology

Species groups

As there are more than 100 species currently cultured in Australia and New Zealand, a comprehensive literature review of the information available for all these species was not considered appropriate here. It was also recognised that the paucity of information with regard to Australian and New Zealand culture species would make compilation of data to set the guidelines difficult. Instead, all finfish, molluscan and crustacean species were divided into eight indicative groups so that efforts could be concentrated on reviewing the data for one or two common species.

Representative species for each group were chosen based on the level of production (i.e. commercial or experimental) and availability of scientific data.

The groups, representative species, their occurrence and their status are summarised in table 9.4.1 (equivalent to table 4.4.1 in Chapter 4, Volume 1). The classification suggested by Lawson (1995) is used to determine the salinity requirements of the species groups, e.g. saltwater or marine species are those which prefer salinities between 33 and 37 g/L (ppt), for estuarine or brackish water species it is 3 to 35 ppt, while freshwater species prefer below 3 ppt.

As indicated in table 9.4.1, a range of aquatic plants, reptiles and invertebrates which are cultured are not included in the list of representative species. At present the production of these species which were left out contributed less than 1.5% of the total value of aquaculture production in Australia in 1997/98 (O'Sullivan & Roberts 1999). Whilst no figures are available for recreational and indigenous fishery, their contribution is thought to be close to zero. Likewise with the commercial fisheries, an examination of the Australian Fisheries Statistics for 1998 (ABARE 1999) shows no specific production data for these species. In New Zealand it is

[1] Given that larval and juvenile stages are invariably the most sensitive to water quality, additional research is required to redress this deficiency. Until this is done, operators of hatcheries and nurseries should use special water treatment to ensure that the water to which the larvae/fry are exposed is the best possible quality.

presumed that the situation would be much the same. Thus the authors feel justified in confining the data collation to the species groups provided in table 9.4.1.

Table 9.4.1 Representative species, occurrence and culture status

Species group	Representative species[a]	Occurrence	Aquaculture Status [b]
freshwater fish	rainbow trout	Australia/New Zealand	commercial/none
	silver perch	Australia	commercial
marine fish	snapper	Australia/New Zealand	commercial/commercial
	flounder/whiting	Australia/Australia	experimental/experimental
brackish water or euryhaline fish	barramundi	Australia	commercial
	black bream	Australia	experimental
freshwater crustaceans	marron	Australia	commercial
	yabbies	Australia	commercial
	red claw	Australia	commercial
	freshwater shrimp	Australia/New Zealand	experimental/commercial
marine crustaceans	black tiger prawns	Australia	commercial
	kuruma prawns	Australia	commercial
edible bivalves	Sydney rock	Australia	commercial
	Pacific oysters	Australia/New Zealand	commercial/commercial
	blue mussels	Australia/New Zealand	commercial/none
	green shell mussels	New Zealand	commercial
pearl oysters	golden lip	Australia	commercial
gastropod/molluscs	abalone/paua	Australia/New Zealand	commercial/commercial
	trochus	Australia	experimental

a The groups of aquaculture species not included in this list are: seaweeds and aquatic plants; crocodiles; a range of live feed and microalgal species; sea cucumbers (beche-de-mer), sponges and other invertebrates.

b Commercial = products offered for sale; Experimental = production but no sales; None = species occurs but there is no culture undertaken.

Data sources

The information used to derive these water quality guidelines was collated during a comprehensive review of the appropriate levels in relevant literature, databases, documents and the internet, including:

- previous Australian and New Zealand Environment Conservation Council (ANZECC) guidelines;
- Australian and New Zealand National Food Authority and state health departments standards and guidelines;
- Australian/New Zealand and international shellfish sanitation programs' requirements;
- Australian Quarantine Inspection Service requirements for seafood export;
- World Bank, South African (freshwater only), European, Canadian and USA aquaculture and general water quality guidelines;
- aquaculture textbooks and reviews;
- extensive industry and expert review of criteria and guidelines including mail surveys, guideline reviews and telephone discussions;
- database searches, specifically CD ROM: ASFA 1978–1987; ASFA 1988–6/96; *Life Sciences* 1982–1985, 1986–1989, 1990–1992, 1993–1995, 1/96–6/96; *Current Contents* 1993–1996.

The following key words were used to search these databases: rainbow trout, *Oncorhynchus mykiss,* barramundi, *Lates calcarifer*, sea perch, silver perch, *Bidyanus bidyanus*, snapper, sea bream, flounder, prawn, *Penaeus monodon*, crustaceans, mussels, *Mytilus edulis*, oyster, *Crassostrea gigas*, pearl oyster, abalone, *Haliotis*, gastropod, Atlantic salmon, *Salmo salar, Salmo gairdneri*, yabbie, marron, redclaw, *Cherax destructor*, crayfish, NOEC, LC_{50}, EC_{50}, tolerance, water quality guidelines, toxicants, pesticides and biocides, toxicity, hazardous chemicals, effects, sublethal responses, turbidity, suspended solids, secchi, heavy metals, Australia, salinity, brackish water, freshwater, marine, pollution, pollutants, environmental pollutants, dissolved oxygen, temperature, ammonia, pH, acidity, alkalinity, survival.

This dataset review was relatively comprehensive, however, due to resource limitations, some of the sources which are more difficult to access were not included in the search strategy. Some unpublished data, research theses, governmental reports, internal reports, and scientific papers published in journals not listed in databases or published in languages other than English, were not accessible. Also some of the information included in the review is based on data from database abstracts only.

Due to the paucity of information for many water quality parameters, recommended ranges for culture were also used. Often, these data were obtained from 'personal communications' with practitioners in this field which were either based on experimental, but non-published, evidence, or on experience. Clearly, any information that has not undergone peer review must be considered with less confidence than information which has been subject to some level of external scrutiny. Further discussion on level of confidence that should be put in the guidelines for aquaculture and harvesting of aquatic foods is provided in Section 9.4.1.5.

Where possible, relevant data on tolerances and toxicity for one or two representative species were collated. These data were summarised for each species group and used to formulate the interim water quality guidelines in Section 9.4.2. A précis of relevant scientific and technical information, together with references, is provided as the rationale for each guideline. Where discrepancies in the data were identified, the more conservative data were generally used. If data for specific water quality parameters could not be found, appropriate data for other species were used to build a data resource for each group. The source information for the aquaculture guidelines was compiled into an aquaculture database which can be accessed on the Guidelines CD-Rom.

For the protection of human consumers of aquatic foods a search of the available data found insufficient information for deriving water quality guidelines that would ensure the Australian and New Zealand food standards would be met. Relevant food standards from the *Food Standards Code* (ANZFA 1996, and updates) established by the Australia New Zealand Food Authority have therefore been provided as guidance. Discussion is provided in Section 9.4.3.

9.4.1.5 Discussion on confidence levels

1. Protection of cultured fish, molluscs and crustaceans

To determine guidelines for each of these water quality parameters, a search of the data-set (Section 9.4.1.4/2) was undertaken for a number of measurements, including:

- no observed effect concentration (NOEC);
- lowest observed effect concentration (LOEC);
- effective concentration (EC_{50}) plus a description of effect;

- concentration to kill 50% of test population (LC_{50}) (the 96 hr exposure period was preferred).

NOEC and LOEC measurements were the most suitable to determine 'safe' levels for protection of aquaculture species. However, the 96 hr LC_{50} was also used, based on the recommendation by Boyd (1990) that an application factor of 0.1 or 0.05 times the lowest 96 hr LC_{50} value may be used to estimate a 'safe' concentration of a potential toxicant for aquaculture species. For example, if the 96 hr LC_{50} of a substance is 0.1 mg/L, a concentration of 0.01 mg/L or 0.005 mg/L may be considered safe for prolonged exposure. Although Boyd (1990) noted that this practice involved some uncertainties, this method has been used in the United States and Japan to establish water quality guidelines for the protection of aquatic animals and plants. However, there can be potential sub-lethal effects on growth or resistance to pathological organisms (R Cordover, pers. comm. 2000).

MATCs — maximum acceptable toxicant concentrations — are often used to indicate *safe* levels. A MATC is equal to the lowest concentration which have been reported to harm organisms in laboratory toxicity (e.g. 96 hr LC_{50}) tests multiplied by an application factor. The *safe* levels recommended by many overseas government agencies (e.g. USEPA, EIFAC, CCME, DWAF) are conservative estimates as they use application factors ranging from 1/10 to 1/100 (Boyd 1989).

In most cases there is a good data set for establishing physico-chemical water quality guidelines for the protection of aquacultural production (Section 9.4.2.1). However, the paucity of information on the effect of inorganic and organic toxicants and biological contaminants on aquaculture species has severely limited the number of water quality guidelines that could be established (Sections 9.4.2.2, 9.4.2.3 and 9.4.2.4, respectively). Where specific water quality guidelines are not available for the protection of aquaculture species, guidelines for the protection of aquatic life (Chapter 3, Volume 1) could be utilised but these are likely to provide a more conservative guideline value.

For those water quality guidelines protecting aquaculture production a high level of credence can be assumed where referenced sources have been used, particularly with review papers such as Boyd (1989, 1990), Meade (1989), Pillay (1990), Svobodova et al. (1993), Schlotfeldt and Alderman (1995), DWAF (1996) and Zweig et al. (1999).

Every care has been taken with the use of personal communications, which are sometimes based on scientific data (although un-referenced) and, at other times, anecdotal evidence. Although the advice is of high quality, attempts to find the required data through scientific experimentation should be made where possible.

The water quality guidelines listed in this Chapter can be used with reasonable confidence to assess ambient water quality for aquacultural uses. If ambient water quality exceeds the guidelines for any parameter then there is a high risk of an impact on aquacultural activities, and further work should be undertaken to better define the risks and potential impacts. However, even though there is a low risk if ambient water quality remains below the guidelines, this cannot be taken as a guarantee that problems will not occur in the future.

2. Protection of human consumers of aquatic foods

The ANZFA food standards for contamination of aquatic foods are legally binding and must be adhered to.

9.4.2 Water quality guidelines for the protection of cultured fish, molluscs and crustaceans

These water quality guidelines are provided as a general guide for aquaculture in Australia and New Zealand. Where specific water quality guidelines for the protection of aquaculture species cannot be made, guidelines for the protection of aquatic ecosystems (Chapter 3) can be used.

Given the large number of different aquaculture production systems and species utilised in Australia and New Zealand, across a wide range of environmental conditions, it should not be assumed that one set of specific values will apply equally in all situations. Local, site-specific information will be needed to supplement the broad information provided in this Chapter.

A decision tree for the determination of water quality guidelines for the protection of aquaculture species is provided in figure 4.4.1 in Volume 1. Specialist assistance may be required to complete those steps where chemical speciation/complexation must be taken into account (Section 3.4.3), and likewise to conduct toxicity tests should they become necessary. A user can make a decision on the risk-based framework and leave the process at any level, however, the further through the process one moves, the greater the confidence in the level of risk.

Tables 9.4.3–9.4.43 provide the water quality guidelines for general freshwater and saltwater (brackish and marine water) aquaculture uses. Where information is available on the specific water quality requirements for each of the species groups in table 9.4.1, it has been included in Section 9.4.2 and should be referenced where guidance is sought for particular species groups. Section 9.4.2 also contains a short discussion for many water quality parameters describing how the guidelines were formulated.

It is worthwhile considering a worked example to demonstrate how the decision tree can be used. An aquaculture company wishes to grow prawns (such as *Penaeus monodon*). They begin by testing the basic (physico-chemical) water quality parameters to obtain a characterisation of the site they wish to use for the prawn culture. Based on recommendations of a prawn farming consultant they test for alkalinity, dissolved oxygen, hardness, pH, salinity and suspended solids. These are provided in table 9.4.2 and when compared with the general saltwater and prawn specific guidelines the site characterisation appears adequate.

However, the company also has several decisions to make regarding the other parameters being outside the guidelines range:

- With respect to dissolved oxygen the site's water quality is below the recommended limit (table 9.4.7) and the farmer would have to undertake additional management to ensure the prawns will grow (e.g. use aerators). This become an economic decision, although tests may be required to determine why the source water is low in dissolved oxygen (may be a sign of organic pollution).
- Salinity at times is at the bottom end of the recommended range (table 9.4.11) and an assessment would need to be made if this would adversely affect production.
- Hardness is quite low compared to the guidelines (table 9.4.9), and again the decision on whether to take steps to alter this (i.e. the addition of limestone) has to be made.

Table 9.4.2 Site characterisation at proposed prawn farm site compared to general and species specific guidelines

	General Saltwater Guidelines (usually mg/L)	Prawn Specific Guidelines (usually mg/L)	Site characterisation (usually mg/L)
Physio-chemical stressors			
Alkalinity	>80	>80	86–89
Dissolved oxygen	>5	>5	>3
Hardness	>50	150–400	40
pH (pH units)	6.6–8.0	6–9	8.2
Salinity	0–36	>15–30	12–22
Suspended solids (Organic matter)	<75	<75	70
Inorganic toxicants			
Cadmium	<0.0005	<0.053–0.15	0.0003
Hydrogen sulphide	<0.002	<0.002	0.001
Organic toxicants			
Endosulfan	0.001	0.01	not detected in water or sediments
Malathion	None provided f/w <0.1	0.001	0.002 (water) 0.003 (sediments)

Overall, through discussions with the other prawn farmers in the region, the company decides that the additional work required to keep the oxygen, salinity and hardness levels within that required for prawns, can be maintained economically year round. The company determine that from the point of view of the physio-chemical parameters there is low risk (i.e. the water quality is acceptable) in utilising that particular site for prawn farming.

It becomes a lot more complicated with the various toxicants as site or regional specific environmental factors (such as water hardness, dissolved organic matter and turbidity) can significantly influence the availability and/or the effects of a contaminant to the culture organism. Given the large number of chemicals and biological contaminants (Sections 9.4.2.2, 9.4.2.3 and 9.4.2.4), and the high cost of measurement of many contaminants (particularly the pesticides and other organic toxicants), some assistance is required in selecting which ones to test for. The basis for measurement of inorganic chemicals, organic toxicants and organisms might be experience from other farms in the area, or a history of potential pollutants in the water source.

Again through discussions with other prawns farmers, consultants and local government authorities, the aquaculture company determines that there are a number of potential contaminants. A consideration of the factors affecting toxicity (hardness, metal bioavailability, bioaccumulation; refer to Section 8.3, Volume 2) shows that the main inorganic chemicals of concern are cadmium and hydrogen sulphide, whilst the pesticides Endosulfan and Malathion have been used in the area for banana farming so they could also be of concern.

Water and soil (sediment) samples were taken and analysed in a registered laboratory and the inorganic chemicals were found to be within the guidelines range, although the company was warned that cadmium can be of concern from a human food safety viewpoint (Section 4.4.5). No Endosulfanwas detected, however, the levels of Malathion in both the water and the sediments were higher than that recommended for black tiger prawns (table 9.4.41), however, no guidelines were available for saltwater.

A series of acute and chronic toxicity tests undertaken by the local university showed that the prawns were not adversely affected and there were no human food safety concerns.

Therefore the company found that the sources waters were of low risk for their planned prawn farm.

9.4.2.1 Physico-chemical parameters

A number of basic parameters need to be tested in all water sources used for aquaculture, including dissolved oxygen, hardness, salinity and temperature. Many of these parameters are also regularly monitored in the culture system to ensure that the aquatic organisms are being held in conditions conducive to survival and growth.

1. Alkalinity

Alkalinity relates to the capacity of the water to accept protons and is a measure of the water's buffering (acid neutralising) capacity when considered in conjunction with other water quality parameters (i.e. CO_2). The alkalinity of water is the amount of carbonates, bicarbonates, hydroxides and, to a lesser extent, silicates, borates, phosphates and organics (Klontz 1993). It is expressed as mg $CaCO_3$/L or as mEq/L — the number of milli-equivalents of hydrogen ions which are released by 1 kg of water when an excess of acid is added (Strickland & Parsons 1968).

The chemical composition of rocks and soils strongly influences the natural alkalinity of water, which can range from very low values to several hundred mg/L $CaCO_3$ (DWAF 1996, Zweig et al. 1999). Waters with moderate to high alkalinity tend to be more strongly buffered than waters with low alkalinity. Seawater has a mean total alkalinity of 116 mg/L (Lawson 1995).

Guideline notes

Zweig et al. (1999) state there are no direct effects of alkalinity on fish and shellfish, however, it is an important parameter due to its indirect effects, including the protection of aquatic organisms from major changes in pH. In addition, in low alkalinity waters, where CO_2 and dissolved carbonates are at low concentrations, photosynthesis may be inhibited, thus restricting phytoplankton growth (Lawson 1995). DWAF (1996) considers that alkalinity below 20 mg $CaCO_3$/L is less suitable for fish culture due to the associated unstable water chemistry, while levels above 175 mg $CaCO_3$/L reduces natural food production in ponds which, in turn, leads to below optimal production. Tucker and Robinson (1990) suggest that a range between 20 and 400 mg/L is sufficient for most aquaculture purposes, although the desirable level is ≥100 or 150 mg/L. Tyco (pers comm 1999) stated that many surface waters in Australia have alkalinities from 10 to 30 mg/L and support fish. Thus, a guideline level of ≥20 mg/L is recommended for freshwater species (table 9.4.3).

Salt water is slightly alkaline and has a strong buffering capacity (Kulle 1971) so alkalinity is not usually of concern for most seawater and brackish water aquaculturists. However, Meade (1989) suggested a range of 10 to 400 mg/L for saltwater species, so a guidelines level of >10 mg/L is recommended for all saltwater culture species (table 9.4.3).

See also discussion under pH (Section 9.4.2.1, no.8).

Table 9.4.3 Summary of the recommended water quality guidelines for alkalinity

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	≥20 >10	freshwater saltwater	DWAF (1996) based on Meade (1989)
General	≥20 >10–400 ≥100–150	freshwater saltwater most aquaculture purposes	DWAF (1996) Meade (1989) Tucker & Robinson (1990)
Freshwater fish	20–400 20–200 20–175 20–400 15–20	silver perch rainbow trout freshwater species silver perch salmonids	Rowland pers comm Forteath pers comm DWAF (1996) Rowland (1995a) SECL (1983)
Marine fish	>20 >100	 Atlantic salmon (in sw)	Swindlehurst pers comm Klontz (1993)
Brackish water fish	>5	barramundi	Curtis pers comm
Freshwater crustaceans	50–100 50–300 50–150	redclaw yabbies marron	Jones (1990) Wingfield pers comm Wingfield pers comm
Marine crustaceans	>80		Swindlehurst pers comm
Edible bivalves	>20		Swindlehurst pers comm
Non edible bivalves	>20		Swindlehurst pers comm
Gastropods	>20		Swindlehurst pers comm

2. Biochemical oxygen demand (and COD)

The biochemical oxygen demand (BOD) is a measure of the combined biological and chemical demand on dissolved oxygen in a system. It is a measure of the amount of oxygen required by bacteria, algae, sediments and chemicals over a set period of time. BOD is of importance in aquaculture because microbial degradation of organic matter is a major sink for dissolved oxygen, a highly important parameter for aquaculture (Zweig et al. 1999).

Aquaculture operations should not utilise waters which are polluted with chemicals and/or excessive nutrients. Thus, BOD becomes an important parameter for aquaculture. Increasing levels of BOD indicate organic pollution which is a cause of concern for aquaculturists (Schlotfeldt & Alderman 1995).

BOD is often measured as the five day BOD (BOD_5), defined as the amount of dissolved oxygen consumed by microorganisms in the biochemical oxidation of organic matter over a 5 day period at 20°C. However, for aquaculture operations, the time period and temperature conditions under which BOD is estimated can be modified, with the resultant value being expressed as a function of time (i.e. mg L^{-1} hr^{-1}) (Zweig et al. 1999).

Some regulatory authorities, e.g. Queensland's Environmental Protection Agency, are moving away from monitoring requirements for this parameter because of the difficulty of measuring and the availability of better indicators for aquaculture. According to Semple (pers comm) total organic carbon (TOC) is a more direct and effective measure of the environmental impact of an effluent stream than BOD_5 and it allows timely intervention in the operations. Recent research undertaken by Brisbane Caltex Refineries has demonstrated that BOD_5 can be effectively correlated to TOC with a 95% confidence level.

Chemical oxygen demand (COD) is a theoretical maximum measure of the amount of oxygen required by the chemicals in a water source. It is usually only significant where high concentrations of chemicals are in the water, e.g. effluent from factories.

Guideline notes

As most aquaculture activities can increase BOD, a low background level is preferred. Svobodova et al. (1993) noted that the BOD_5 for cyprinids is 8 to 15 mg/L while for salmonids the corresponding levels are up to 5 mg/L (both depend on the intensity of the culture system and the rates of aeration).

For freshwater species, the COD and BOD guidelines suggested by Schlotfeldt and Alderman (1995) are used as the recommended guideline (table 9.4.4). Little information is available for marine species, so no guideline is provided.

Svobodova et al. (1993) noted that the COD maximum level for cyprinid culture is 20–30 mg/L while for salmonids the corresponding levels are up to 10 mg/L (both depend on the intensity of the culture system and the rates of aeration). The COD level for saltwater is yet to be determined.

The guidelines can be used while taking into account factors such as dissolved oxygen requirements of the culture species, the degree of pond aeration, seasonal temperature fluctuations, expected photosynthetic activity, and oxygen solubility. A resultant judgement can be based on the appropriate BOD for the source water (Zweig et al. 1999).

See also discussions under Dissolved oxygen (9.4.2.1/5) and Suspended solids (9.4.2.1/10).

Table 9.4.4 Summary of the recommended water quality guidelines for biochemical oxygen demand

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<15	freshwater BOD_5	Schlotfeldt & Alderman (1995)
	<40	freshwater COD_5	Schlotfeldt & Alderman (1995)
	ND	saltwater BOD_5	
	ND	saltwater COD_5	
General	<15	freshwater BOD_5	Schlotfeldt & Alderman (1995)
	<40	freshwater COD_5	Schlotfeldt & Alderman (1995)
Freshwater fish	<10	rainbow trout BOD	Forteath pers comm
	<12	freshwater species BOD	DWAF (1996)
	<30	rainbow trout COD	Forteath pers comm
	<5	salmonids BOD	Svobodova et al. (1993)
	<10	salmonids COD	Svobodova et al. (1993)
Marine fish	<10	BOD_5	Swindlehurst pers comm
Brackish water fish	<20	BOD_5	Swindlehurst pers comm
Freshwater crustaceans	<10	BOD_5	Swindlehurst pers comm
Marine crustaceans	<10	BOD_5	Swindlehurst pers comm
Edible bivalves	<10	BOD_5	Swindlehurst pers comm
Non edible bivalves	<20	BOD_5	Swindlehurst pers comm
Gastropods	<10	BOD_5	Swindlehurst pers comm

ND Not determined — insufficient information

3. Carbon dioxide

Carbon dioxide is a natural component of surface water. It is dissolved in water in its molecular gaseous states; only 10% is in the form of carbonic acid, H_2CO_3. These two forms of carbon dioxide together constitute what is termed free CO_2. The ionic forms (i.e. fixed carbon dioxide) are represented by the bicarbonate and carbonate ions (HCO_3^- and CO_3^{2-} respectively). Their presence is important for the buffering capacity of the water (Svobodova et al. 1993).

The level of carbon dioxide in the water is related to photosynthetic activity of aquatic plants and respiration of these plants and aquatic animals, as well as bio-oxidation of organic compounds. Dissolved carbon dioxide forms carbonic acid, causing a drop in pH. Likewise, its removal during (algal) plant photosynthesis causes the pH to climb (Walker 1994). At equilibrium, freshwater contains about 2.0 mg/L CO_2 (Klontz 1993) and seldom rises above 20 to 30 mg/L (Svobodova et al. 1993). In waters used for intensive fish culture, free carbon dioxide levels typically fluctuate from 0.0 mg/L in the afternoon to 5 to 10 mg/L at daybreak (Boyd 1990). Zweig et al. (1999) warned that extraordinarily high (toxic) levels of CO_2 can be found in ground waters.

High concentrations of carbon dioxide have a narcotic effect on fish and even higher concentrations may cause death; however, such concentrations seldom occur in nature.

The direct adverse effects can occur when there is an excess of free CO_2, especially in waters low in dissolved oxygen. This latter situation can occur when too much free CO_2 is utilised for photosynthesis of phytoplankton, or when water is vigorously aerated with CO_2 free air. Free CO_2 concentrations below 1 mg/L affect the acid-base balance in fish blood and tissues and cause alkalosis (Svobodova et al. 1993). Fish suffering from free CO_2 deficiency gather close to the water surface and show symptoms of suffocation even though the concentration of oxygen in the water is adequate (Taege 1982).

The toxic action of carbon dioxide is either direct or indirect. The indirect action of both free and bound CO_2 is exerted on fish through its influence on water pH, especially where the values rise to toxic levels (Svobodova et al. 1993). Also, changes in pH affect the toxicity of those chemicals which exist in the dissociated and nondissociated forms of which only one is toxic, such as H_2S and ammonia.

Most aquaculture species will survive in waters containing up to 60 mg/L carbon dioxide provided that dissolved oxygen concentrations are high (Boyd 1989); however, SECL (1983) suggested the carbon dioxide levels should be kept below 20 mg/L for salmonid hatcheries. Unfortunately, carbon dioxide concentrations normally are high when dissolved oxygen concentrations are low.

Guideline notes

Meade (1989) suggested a range of 0 to 10 mg/L for aquaculture. Pillay (1990) recommended that levels should not be above 3 mg/L for most farmed finfish. For freshwater species DWAF (1996) recommended below 12 mg/L and Schlotfeldt and Alderman (1995) below 25 mg/L so a median level of <10 mg/L is recommended as the guideline for freshwater aquaculture (table 9.4.5). For saltwater species the guideline is recommended at <15 mg/L which is the lowest of those provided for the groups in table 9.4.5.

4. Colour and appearance of water

These are not highly objective measurements but many fish farmers and crustacean farmers attach a lot of significance to these two properties of pond water. Colour is a result of the interaction of incident light and impurities in the water (Lawson 1995). There are three common causes of water colouration and variations in water appearance:

- suspension of silt and clay particles
- significant growth of plankton, particularly microalgae
- suspension of humic acids and other organic acids

Table 9.4.5 Summary of the recommended water quality guidelines for carbon dioxide

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<10	freshwater	Professional judgement
	<15	saltwater	Professional judgement
General	<10	aquaculture	Meade (1989)
	<12	freshwater	DWAF (1996)
	<25	freshwater	Schlotfeldt and Alderman (1995)
Freshwater fish	<10	rainbow trout	Pillay (1990), Forteath pers comm
	<6	rainbow trout	Holliman (1993)
	0–15	silver perch	Rowland (1995a)
	<3	farmed fish	Pillay (1990)
Marine fish	<15		Swindlehurst pers comm
Brackish water fish	<15	barramundi	Curtis pers comm
Freshwater crustaceans	<15		Wingfield pers comm
Marine crustaceans	<25		Swindlehurst pers comm
	<20	prawns	Boyd & Fast (1992)
Edible bivalves	<25		Swindlehurst pers comm
Non edible bivalves	<25		Swindlehurst pers comm
Gastropods	<25		Swindlehurst pers comm

Generally, when farmers refer to the 'colour' of the water, they are actually referring to turbidity due to significant silt and clay particle accumulation, or growth of phytoplankton and zooplankton.

Colouration of surface water in rivers and creeks (e.g. humic acids and organic acids), although not due to suspended particles, acts in a similar way with regard to light penetration. This type of water colouration may be beneficial in tank and cage culture as it shades fish and prevents sunburn as well as reducing plant biofouling. However, it may cause difficulties for growers in observing their stock.

Lawson (1995) reported that impending oxygen shortages in the water can often be detected by changes in colour.

Although high colour may shade fish and impede algal growth, it is usually due to tannins. These are phenols which bind with protein and at high levels may affect fish respiration, particularly with sensitive fish species (such as rainbow trout).

Guideline notes

ANZECC 1992 recommended a less than 10% change in euphotic zone for freshwater and saltwater ecosystem protection. Measurement of colour is difficult and is not usually undertaken by farmers. O'Connor (pers comm) suggested 30–40 platinum-cobalt (Pt-Co) units (refer to APHA/AWWA/WEF 1995 for a description of this method) as a good starting point for a recommended guideline (table 9.4.6).

See also discussion under Suspended solids and turbidity (9.4.2.1/10).

Table 9.4.6 Summary of the recommended water quality guidelines for colour

Group	Guideline Pt-Co units	Comments	Reference
Recommended guideline	30–40	freshwater and saltwater	O'Conner pers comm

5. Dissolved oxygen

Dissolved oxygen (DO) is a very basic requirement for aquaculture species (Zweig et al. 1999). However, the amount of oxygen available to aquatic animals is approximately only 0.0015% (w/v maximum) compared with 21% available in air. Boyd (1989) considered that dissolved oxygen is the most critical water quality variable in aquaculture. Anoxia occurs when dissolved oxygen levels in the environment decrease to the point where aquatic life can no longer be supported. In suboptimal dissolved oxygen levels, growth is slowed. Dissolved oxygen is usually expressed in mg/L, ppm or partial pressure.

Some species are more resistant to low levels of oxygen than others. Boyd (1990) noted that the amount of oxygen required by aquatic animals is quite variable and depends on species, size, activity (levels increase with activity), water temperature (doubles with every increase of 10°C), condition (lean fish consume less than fat fish), DO concentration, etc. Other species are air breathers and are able to be farmed under intensive conditions with very low levels of dissolved oxygen and poor water quality (e.g. catfish, eels, aquatic reptiles).

Some species have a greater affinity for oxygen (higher levels of haemoglobin and similar complexes in blood) and, therefore, are more tolerant of low levels. This also relates to the partial pressure of dissolved oxygen in the water and its ability to exchange across gill membranes. This, in turn, governs the minimum oxygen concentration to survive, grow, etc, and is approximately the minimum recommended concentration (Purser 1996 a,b).

Daily fluctuations in impounded waters are higher than those in the open sea or running waters. The DO concentration can fluctuate in response to photosynthesis of aquatic plants and respiration of aquatic organisms. Daily fluctuations are such that the lowest DO concentrations occur soon after sunrise with levels higher in the late afternoon (Boyd 1990).

In ponds, tanks and other enclosed culture systems, mechanical aeration can be used to lift dissolved oxygen levels, while water movement from currents and tides assists in open culture systems. Pure oxygen (oxygenation) may be used to supplement dissolved oxygen levels, particularly in intensive culture systems.

The factor most frequently responsible for a significant reduction in the oxygen concentration of the water (oxygen deficiency) is pollution by biodegradable organic substances (including waste waters from agriculture, the food industry and public sewage). These substances are decomposed by bacteria which use oxygen for this process (Svobodova et al. 1993). The most common cause of low DO in an aquaculture operation is a high concentration of biodegradable organic matter in the water, resulting in a high BOD. This problem is further exacerbated at high temperatures (Zweig et al. 1999).

Guideline notes

As suggested by Zweig et al. (1999), setting DO guidelines for source water is difficult as DO can be affected by many processes independent of the initial source water DO. Thus, at the site selection stage, the initial DO and BOD can be used to assess the ability of the source water to maintain appropriate oxygen levels. Other factors affecting DO concentration in aquaculture operations can only be assessed and if necessary mitigated once the operation is running (Zweig et al. 1999).

Meade (1989) said that dissolved oxygen levels above 5 mg/L provide protection for most aquaculture species and this level is recommended as the guideline (table 9.4.7).

See also discussions under Biochemical oxygen demand (9.4.2.1/2) and Temperature (9.4.2.1/11).

Table 9.4.7 Summary of the recommended water quality guidelines for dissolved oxygen

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	>5	freshwater and saltwater	Meade (1989)
General	>5	freshwater and saltwater	Meade (1989)
Freshwater fish	>6	coldwater species, & warmwater species	DWAF (1996), Lawson (1995)
	>5	rainbow trout	DWAF (1996)
	>6	rainbow trout	Pillay (1990)
	>4.5 (afternoon)	silver perch, optimal	Rowland (1995a)
	>3.0 (dawn)	silver perch, optimal	Rowland (1995a)
Marine fish	>7		Alabaster & Lloyd (1982)
	>6		Huguenin & Colt (1989)
Freshwater crustaceans	>5	can tolerate lower levels	Wingfield pers comm
	>3		Swindlehurst pers comm
Marine crustaceans	>5	prawns	Boyd (1989), Lee & Wickins (1992)
Gastropods	>3	abalone	Fallu (1991)

6. Gas supersaturation (total gas pressure)

Supersaturation of dissolved gas occurs when the pressure of the dissolved gas (total gas pressure; TGP) exceeds the atmospheric pressure. TGP refers to the sum of the partial pressures of dissolved gases in the water (i.e. oxygen, nitrogen and carbon dioxide).

Supersaturation can occur via a range of processes including an increase in temperature, mixing waters of different temperatures, air entrainment (e.g. as in a waterfall), photosynthesis, and bacterial activity (Lawson 1995). Supersaturation (especially in well or spring water used in hatcheries) can also occur when physical processes such as pressurised air injections are improperly applied, when rapid temperature increases occur, or when air bubbles are carried to great depths (Tomasso 1993). It also can occur where heaters are used, especially if the water is in pipes and under pressure.

Gas supersaturation can be caused by entrainment of air bubbles when water falls over high dams and often results in air leaks in pipes (Nebeker & Brett 1976), or improper submergence of the intake of pumps (Kils 1977), highly efficient submerged aerators (Colt & Westers 1982) and high levels of photosynthesis in ponds (Takashi & Yoshihiro 1975). It may also occur during fish transport, especially in aeroplanes where the pressure falls at altitude, or in road tankers where oxygen is used and in systems with oxygenation. In Tasmania, freshwater fish kills have been reported due to supersaturation of waters flowing out of a hydro-electric plant.

Nitrogen supersaturation is the main problem as it is the major (78%) component of air. The maximum level is around 103% of atmospheric pressure before problems occur. Water supersaturated with nitrogen is unable to carry adequate oxygen for fish (Klontz 1993).

Oxygen saturation up to 200–300% can be tolerated if oxygen is used directly or during photosynthesis (when air is used, nitrogen becomes the main component and problems can occur). It can cause massive distension of the swim bladder of salmonids, although the mortality is usually low (Klontz 1993). This can occur if the water supply is from highly vegetated streams on bright sunny days.

Gas-bubble disease is a problem related to the supersaturation of gases in water. Changes in pressure may cause bubbles to form in the blood and tissues of aquatic animals. This

phenomenon is known as gas bubble trauma which may cause acute or chronic problems, especially in eggs, larvae and juveniles.

The signs of this problem are pop-eye (exophthalmia, which is not always evident in cases of gas bubble disease, can also be due to other causes) and the presence of bubbles under the skin (easily visible in the fins and on the head) and in the gills. Fish suffering from this condition usually leap vigorously from the water before they die (Nowak 1996).

High carbon dioxide levels in fish transport systems (where ventilation is absent) can inhibit oxygen uptake.

Guideline notes

Although Svobodova et al. (1993) recommended that the N_2 levels at existing atmospheric pressure should be below 300%, DWAF (1996), Meade (1989) and SECL (1983) claimed dissolved oxygen levels should be much lower at between 103 to 105%. Lawson's (1995) conservative suggestion of a level of <100% (N_2 existing atmospheric pressure) is recommended as the guideline for both freshwater and saltwater aquaculture (table 9.4.8).

See also discussions under Dissolved oxygen (9.4.2.1 No.5).

Table 9.4.8 Summary of the recommended water quality guidelines for gas supersaturation

Group	Guideline	Comments	Reference
Recommended guidelines	<100%	freshwater & saltwater	Lawson 1995
General	<100% <103–105	freshwater	Lawson 1995 SECL (1983), Meade (1989) DWAF (1996)
Freshwater fish	<105%		Swindlehurst pers comm
Marine fish	<105%		Swindlehurst pers comm
Brackish water fish	<105%		Swindlehurst pers comm
Freshwater crustaceans	<120%		Swindlehurst pers comm
Marine crustaceans	<120%		Swindlehurst pers comm

7. Hardness

Total hardness primarily measures the concentration of all metal cations (usually dominated by calcium and magnesium in freshwater) in the water, with the exception of alkali metals (Zweig et al. 1999). Hardness is normally expressed as the level of calcium carbonate ($CaCO_3$) in mg/L and can be divided (Sawyer & McCarty 1978) into four categories:

- soft water has the range 0 to 75 mg/L;
- moderately hard water ranges from 75 to 150 mg/L;
- hard water ranges from 150 to 300 mg/L;
- very hard water is >300 mg/L $CaCO_3$.

Soft water is usually acidic while hard water is generally alkaline. Most fresh surface waters in Australia and New Zealand have a hardness between 10 and 400 mg/L as $CaCO_3$.

In soft waters, carbonate and bicarbonate salts are in short supply, so large pH swings can be common place. Hard water has been found to reduce the toxicity of several heavy metals (e.g. cadmium, chromium(III), copper, lead, nickel and zinc; SECL 1983), as well as ammonia and the hydrogen ion (Zweig et al. 1999).

Some aquacultural species have a specific requirement for calcium, for bone formation in fish and exoskeleton formation in crustaceans. Calcium is also necessary for proper osmoregulation, and the calcium ion generally reduces the toxicity of hydrogen ions, ammonia and metal ions. High calcium levels in freshwater can inhibit phytoplankton growth; however, blue-green algae are known to thrive in harder water (high Ca^{2+}) which can influence productivity of the pond water.

Guideline notes

Hardness averages 600 mg/L in ocean water and therefore is not a problem in seawater or brackish water systems (Lawson 1995). Desirable hardness levels vary for different freshwater species and groups of species as summarised in table 9.4.9.

Although species requirements vary markedly (table 9.4.9), Meade (1989) recommended a range between 10 and 400 mg/L for aquaculture. The recommended guideline range for freshwater species is 20–100 mg/L as proposed by DWAF (1996). In saltwater, the hardness requirement is not of concern (Lawson 1995).

Table 9.4.9 Summary of the recommended water quality guidelines for total water hardness

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	20–100 NC	freshwater saltwater	DWAF (1996) Lawson (1995)
General	20–100 NC 10–400	freshwater saltwater aquaculture	DWAF (1996) Lawson (1995) Meade (1990)
Freshwater fish	20–300 50–100 10–160 10–200 20–175	 rainbow trout silver perch silver perch freshwater species	Boyd & Walley (1975), Romaire (1985) Forteath pers comm Rowland pers comm Rowland (1995a) DWAF (1996)
Brackish water fish	50–200	barramundi	Curtis pers comm
Freshwater crustaceans	>100 50–200 50–400 50–300 >50	crayfish crayfish and shrimp yabbies marron crayfish	De la Bretonne (1969) Lee & Wickins (1992) Wingfield pers comm Wingfield pers comm Boyd (1990)
Marine crustaceans	160–400	black tiger prawn	Lee & Wickins (1992)

NC: Not of concern

8. pH

The term pH refers to the hydrogen ion (H^+) concentration in water; more generally, pH refers to how acidic or basic a water is. pH is interdependent with a number of other water quality constituents, including carbon dioxide, alkalinity and hardness. It is known to influence the toxicity of hydrogen sulphide, cyanides and heavy metals, as well as having an indirect effect on ammonia levels; un-ionised NH_3 increases with pH (Klontz 1993).

In aquaculture, low pH is often a consequence of sulfuric acid formation by the oxidation of sulphide-containing sediments, as commonly occurs where iron pyrite is present (Lawson 1995, Zweig et al. 1999). The EIFAC (1969) noted that acidification of highly alkaline water can increase the free carbon dioxide concentration, resulting in CO_2 toxicity rather than pH imbalance. In addition, acid water tends to dissolve metals more readily. For example, aluminium concentrations are high in acid waters (Haines 1981). According to Nowak (1996), acidification of estuarine tributaries due to drainage of acid sulfate soils (which have

pH <3.5) can cause low pH by providing a long term source of dilute sulphuric acid and dissolved metals (iron, aluminium and manganese). High pH in aquaculture is commonly a result of excess photosynthesis in waters with high alkalinity and low calcium hardness (Zweig et al. 1999).

pH can indirectly affect aquaculture species through its effect on other chemical parameters (Zweig et al. 1999). For example, low pH reduces the amount of dissolved inorganic phosphorus and CO_2 available for phytoplankton photosynthesis. In addition, low pH can result in the solubilisation of potentially toxic metals from the sediments, while at high pH, the toxic form of ammonia becomes more prevalent. Phosphate, which is commonly added as a fertiliser, can precipitate at high pH (Boyd 1990, Zweig et al. 1999).

However, species tolerances can vary. For example, in comparison with cyprinids (especially carp and tench), salmonids are more vulnerable to high pH and more resistant to low pH (Svobodova et al. 1993).

During transfers, animals should be acclimatised slowly to waters of different pH.

Guideline notes

Meade (1989) recommended that pH be maintained at between 6.5 and 8.0 for all aquaculture species.

In freshwater, pH can change quickly due to the amount of carbon dioxide added or removed during plant growth. In culture systems, particularly recirculation systems, the pH may be reduced (more acidic) by the production of metabolites. Buffering is, therefore, important in such systems.

Most estuarine and freshwater species are tolerant of a relatively wide range of environmental pH (Tomasso 1993), around pH 5.5 to pH 8 (Schlotfeldt & Alderman 1995). Swingle (1969) claims that the desirable range for warmwater pond fish is 6.5 to 9.0 (measured at daybreak [Ellis 1937]). A range of 5.0 to 9.0 was considered safe by the European Inland Fisheries Advisory Commission (EIFAC 1969). Above and below this range results in slow growth and then death. However, these ranges may be too high when considering interactions with other environmental variables or during certain stages of the life cycle. For example, in water containing high levels of ammonia, a pH of 9.0 will cause a high percentage of the ammonia to exist in the toxic un-ionised form of ammonia (Emerson et al. 1975). This is a problem in poorly buffered pond water during the late afternoon hours when the natural pH rhythm peaks (pH increases through the day as photosynthesis increases). In fact pH has been known to exceed 10 to 11 in poorly buffered ponds in the late afternoon.

Therefore, the recommended guideline (table 9.4.10) is that pH be maintained at between 5.5 and 9.0 for freshwater.

However, seawater, in general, resists changes in the pH values (Poxton & Allhouse 1982) and usually has a pH around 8.2 (Walker 1994). The alkalinity of the seawater provides greater protection against carbon dioxide build-up, while in the well-buffered brackish water the pH is normally between 6.5 and 9.0 (Boyd 1989). For saltwater species the range of 6.0 to 9.0 pH units is recommended as the guideline (table 9.4.10).

It should be noted that pH can change by the hour as a function of photosynthesis which removes carbon dioxide. This is particularly the case in pond-based culture systems. Therefore, readings should be taken over the daylight hours to gain a better appreciation of the pH levels.

See also discussions under Alkalinity (9.4.2.1/1) and Temperature (9.4.2.1/11).

Table 9.4.10 Summary of the recommended water quality guidelines for pH

Group	Guideline (pH units)	Comments	Reference
Recommended guidelines	5.5–9.0 6.0–9.0	freshwater saltwater	Professional judgement Professional judgement
General	5.5–8.0 6.5–8.5	freshwater all aquaculture species	Schlotfeldt & Alderman (1995) Meade (1989)
Freshwater fish	6.5–9.0 7–7.5 7–7.5 6.5–9.0 5.0–9.0	silver perch, rainbow trout aquaculture species rainbow trout warmwater pond fish	Rowland (1995a), CCME (1993) DWAF (1996) Holliman (1993) Swingle (1969) EIFAC (1969)
Marine fish	6.7–8.6	optimal	Pillay (1990)
Brackish water fish	6.7–8.6	optimal	Pillay (1990)
Freshwater crustaceans	6.5–8.5	freshwater crustaceans	various
Marine crustaceans	7.8–8.3 6–9	prawns and crabs prawns	Lee & Wickins (1992) Boyd (1989)

9. Salinity (total dissolved solids)

Total dissolved solids is a composite measure of the total amount of material dissolved in water. This parameter can be represented in three ways: as total dissolved solids (TDS), as salinity or as conductivity. TDS and salinity are both measures of the mass of solutes in water; however, they differ in the components they measure (salinity only measures dissolved inorganic content whereas TDS is the mass of dissolved inorganic and organic compounds in water).

Salinity is the main measure used in aquaculture, as it influences the water and salt balance (osmoregulation) of aquatic animals. It usually is expressed in mg/L, but in aquaculture it is commonly expressed in parts per thousand (ppt or ‰). Most inland waters contain 0.05 to 1.0 ppt salinity, although in arid regions and with artesian water the salinity can be very high. Estuarine waters may range from 0.5 to more than 30 ppt often depending on the depth of the sample; marine waters range between 30.0 to 40.0 ppt, brine or hypersaline waters display salinities above 40 ppt.

As with pH (Section 9.4.2.1/8) salinity can vary significantly over a short time period (e.g. 5–6 hours), particularly in or near estuaries. It can also vary significantly with various weather events, particularly precipitation in the catchment of the water source. Therefore readings need to be made over the appropriate time periods (daily and seaonal).

Salinity directly affects the levels of dissolved oxygen: the higher the salinity, the lower the dissolved oxygen levels at a given water temperature.

Like temperature, salinity is an important limiting factor in the distribution of many aquatic animals. Diadromous fish (e.g. barramundi) and anadromous fish (e.g. salmonids) can move between full-strength seawater and freshwater as part of their reproductive activities. Brackish water species are more tolerant of rapid changes in salinity; however, even they can be limited in their distribution by salinity gradients. Euryhaline animals can tolerate wide changes in salinity, while those tolerating only limited ranges are referred to as stenohaline.

Some animals (e.g. fish) are osmoregulators and are able to regulate the concentration of their body salts despite changes in the salinity of their environment, whiles others (e.g. bivalves) are osmoconformers and alter their salt levels to that of the environment.

Salinity requirements can vary for particular species depending on their life cycle stage. Salinity also affects the temperature requirements of some species, although there is a lack of understanding of temperature-salinity interactions and the effects of changing the ionic ratios for many species (Tomasso 1993).

Freshwater organisms have body fluids more concentrated in ions than the surrounding water, meaning that they are hypersaline or hypertonic to the environment. These animals tend to accumulate water which they must excrete while retaining ions. Saltwater species have body fluids more dilute in ions than the surrounding water; they are hyposaline or hypotonic to their environment. They must excrete ions and uptake water continually. Outside of their natural salinity ranges, aquatic animals must expend considerable energy for osmoregulation at the expense of other processes, such as growth.

Many brackish water and marine animals can adjust to changes in salinity if the change is made gradually (i.e. no more than 10% change in an hour).

Guideline notes

Salinity tolerance varies significantly between species and some species have wider tolerances than others (particularly those which live in brackish water. However, the recommendations of Lawson (1995) are used for the recommended guidelines (table 9.4.11)

See also discussions under Suspended solids and turbidity (9.4.2.1/10).

Table 9.4.11 Summary of the recommended water quality guidelines for salinity

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<3	freshwater	Lawson (1995)
	3–35	brackish water	Lawson (1995)
	33–37	saltwater	Lawson (1995)
General	not applicable	See species requirements	
Freshwater fish	<3		Lawson (1995)
Marine fish	>30		Swindlehurst pers comm
	30–40		Zweig et al. (1999)
	33–37		Lawson (1995)
Brackish water fish	3–35		Lawson (1995)
Freshwater crustaceans	<6	best for yabbies	Mills & Geddes (1980)
	<7	best for marron	Morrissy (1976)
Marine crustaceans	15–30	*P. monodon*	Lee & Wickins (1992)
	8–35	crabs and lobsters	Lee & Wickins (1992)
Edible bivalves	27–39	Highest survival for Sydney rock oyster larvae	Nell & Holliday (1988)
	20–40	Good growth in sydney rock oyster adults	Nell & Holliday (1988)
	19–27	Best for Pacific rock oyster larvae	Nell & Holliday (1988)
	15–45	Good growth in Pacific oyster adults	Nell & Holliday (1988)
Non edible bivalves	>30	pearl oysters	FAO/UNDP (1991)
	20–35	*Pinctada fucata*	Namaguchi (1994)
	30–34	*Pinctada maxima* spat	Southgate pers comm
	30–35	*Pinctada margaritifera* spat	Southgate pers comm
Gastropods	>25	abalone	Hahn (1989)

10. Suspended solids and turbidity

There are three basic types of suspended solids:

- phytoplankton, zooplankton and bacterial blooms
- suspended organic and humic acids
- suspension of silt and clay particles

All influence the level of turbidity (turbidity increases with suspended solids) and scatter light, restricting penetration into water. In aquaculture ponds, less light penetrating to the bottom inhibits growth of troublesome filamentous algae and aquatic weeds.

Particularly in aquaculture ponds, the biological turbidity can vary significantly due to a number of management strategies (refer to Boyd 1989 and 1990 for further discussion). This turbidity is often measured in centimetres using a secchi disc (i.e. it is the distance (cm) into the water at which a black and white disc become visible to the naked eye). For silver perch, the preferred secchi disc reading is 30 to 45 cm (Rowland 1995a), <200 cm for snapper (Ogburn 1996), <30 cm for barramundi, 30 to 40 cm for freshwater crayfish (O'Sullivan 1992), and <20 cm for prawns (Anderson 1993).

Typically, if the secchi disk reading is below 10 cm water turbidity is excessive. If turbidity is due to the presence of phytoplankton, there is likely to be a problem with dissolved oxygen concentrations when the light level decreases below the photosynthetic compensation level. Conversely, if turbidity is due to silt/clay or organic matter, planktonic productivity will be low.

Duchrow and Everhart (1971) pointed out that the main concern with regard to the protection of sessile benthic aquatic fauna and flora is not the suspended particles (turbidity) per se, but the amount of solids in suspension that potentially can settle out (settleable or suspended solids).

The measure for suspended solids (sometimes called non filterable residue or NFR) is measured in mg/L. The opposite is filterable residue or total dissolved solids (refer to Salinity, Section 4.4.4.3/1 for more information).

Suspended solids can cause gill irritations and tissue damage, which increases the stress levels of aquatic animals. Cold water fish have been killed upon exposure for 3 to 4 weeks to 500 to 1000 mg/L of suspended solids (Alabaster & Lloyd 1982). Turbid waters can also shield food organisms and clog filters (Zweig et al. 1999). Although sediment accumulation may be troublesome, the oxygen demand of the sediment and of particulate and dissolved organic matter has more serious consequences (Klontz 1993).

The practice of mechanical aeration tends to create water currents which maintain soil particles in suspension and perpetuates the turbidity of the pond (Boyd 1990). Problems of off-flavours in fish and crayfish are less common in turbid ponds (Walker 1994) (except where algae cause the turbidity), although the blue-green algae *Microcystis* is known to exist in waters with high clay turbidity.

Guideline notes

The effect of this criteria varies considerably between species. Meade (1989) recommended a level below 80 mg/L for aquaculture species.

Klontz (1993) stated that levels below 80 mg/L were quite innocuous for freshwater fish. Alabaster and Lloyd (1982) recommended a level below 80 mg/L for freshwater aquaculture, however, some species (e.g. rainbow trout) require lower levels of suspended solids so a median level of <40 mg/L is recommended as the guideline (table 9.4.12).

Marine species (e.g. snapper) are generally less tolerant, so the recommended guideline is <10 mg/L based on the lowest species recommendation i.e. snapper (table 9.4.12). However, as brackish water species (e.g. prawns and barramundi) can tolerate higher levels the recommended guideline for such waters is <75 mg/L.

Table 9.4.12 Summary of the recommended water quality guidelines for suspended solids and turbidity

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<40	freshwater	Professional judgement
	<10	saltwater	Professional judgement
	<75	brackish water	Professional judgement
General	<80	freshwater	Alabaster & Lloyd (1982)
	<80	all aquaculture species	Meade (1989)
Freshwater fish	<80		Klontz (1993)
	<25	rainbow trout	SECL (1983), Lloyd (1992)
Marine fish	<25	Atlantic salmon	Klontz (1993)
	<10	snapper	Ogburn (1996)
Brackish water fish	<75	barramundi	Swindlehurst pers comm
Marine crustaceans	<14	crabs/lobsters	Lee & Wickins (1992)
	<75	black tiger prawns	Swindlehurst pers comm
Non edible bivalves	<25	pearl oysters	FAO/UNDP (1991)
	<40	*Pinctada maxima* spat	Mills pers comm

11. Temperature

Water temperature is a fundamental parameter that affects the health of aquatic organisms. These organisms all have specific temperature ranges within which they can live normally. The natural temperature range encountered in specific regions of the sea is small in comparison with that observed in freshwater, particularly impounded surface waters.

The availability of oxygen is directly affected by the temperature of a water body (salinity and the rate of oxidation of organic matter also affect oxygen availability). Water temperature affects metabolism (metabolic rate), feed intake, growth, reproduction, physiological processes (affects the function of enzymes), disease immunity, movements and respiration rate. It also influences the susceptibility to potential toxic compounds and ammonia levels (Klontz 1993), as well as the bioaccumulation and detoxification (Zweig et al. 1999) and solubility of fertilisers.

Water temperature tolerances are specific to each species and are difficult to group into categories. Rowland (1986) pointed out that many species suitable for aquaculture will survive and reproduce over a wide temperature range, but the optimum temperature range for maximum growth is more narrow. For example, a species might tolerate temperatures of 5 to 36°C, but the range for maximum growth might be from 25 to 30°C. It is useful to note that best growth often occurs when the water is close to lethal temperatures; care is required to prevent losses if temperatures rise.

In aquaculture, it is seldom economical to cool or heat large volumes of water. Sites should be selected in geographic regions that provide an ambient temperature conducive to the growth of market size products within a reasonable period of time (Lawson 1995). It is imperative that the temperature never deviates beyond lethal limits (Zweig et al. 1999). Therefore, species which exhibit maximum growth rates at prevailing water temperatures usually are selected for a particular location (Lawson 1995, Boyd 1999):

Tropical/subtropical	grow well above 26–28°C
Warm water	grow best at 20–28°C
Cool water	grow well between 15 and 20°C
Cold water	grow best below 15°C

If animals are transferred between waters with a greater temperature difference than 3 to 4°C, the sudden changes in metabolism may cause thermal shock and even death (Boyd 1990). Temperature change of 0.2°C/min usually can be tolerated for overall changes below 2°C over a one hour period (ANZECC 1992).

Tomasso (1993) noted that the temperature requirements of a given (fish) species will vary with several factors:

- estuarine species may exhibit more or less tolerance of extreme temperatures depending on the concentration of dissolved solids in their environment;
- acclimation to extreme temperature can occur in some species;
- differing stages of the life cycle may have different temperature optima;
- complex physiological changes occurring during reproduction very often are dependent on absolute temperatures, changing temperatures and interaction with other abiotic factors, such as photoperiod.

Guideline notes

This general water quality criteria varies significantly between species (see Lawson 1995 and Zweig et al. 1999 for species summaries). Consequently it is recommended that changes to water temperature be kept below 2°C over a one hour period (table 9.4.13) as provided by ANZECC (1992).

See also discussions under BOD (9.4.2.1/2) and Dissolved oxygen (9.4.2.1/5).

Table 9.4.13 Summary of the recommended water quality guidelines for temperature

Group	Guideline	Comments	Reference
Recommended guideline	<2.0ºC change	over 1 hour	ANZECC (1992)

9.4.2.2 Inorganic toxicants (heavy metals and others)

A number of chemicals can occur in surface waters as a result of human activities. These can be of inorganic (this Section) or organic (Section 9.4.2.3) origin.

A wide range of heavy metals can be a problem in freshwater, brackish water and inshore marine aquaculture, especially in areas of human habitation (pollution). Trace quantities of metals are present in natural waters; however, their concentrations are generally greater where pollution from industrial processes (ore mining and processing, smelting plants, rolling sheet metal mills, textile and leather industries) as well as exhaust gases of motor vehicles and burning of other fossil fuels occurs. The metals of greatest concern to fisheries (and aquaculture) include aluminium, arsenic, cadmium, chromium, copper, iron, lead, mercury, nickel and zinc (Svobodova et al. 1993). Other inorganic toxicants include ammonia, chlorine, cyanide, fluoride, hydrogen sulfide, nitrite, nitrate and phosphates.

Increasing hardness (9.4.2.1/7) reduces the uptake and toxicity of several metals, including cadmium, chromium (III), copper, lead, nickel and zinc, to freshwater organisms. Other physio-chemical parameters, especially pH and redox potential, will also influence metal bio-availability (refer to equations in Section 3.4.3.2 of Volume 1).

Speciation of metals is important in determining toxicity to aquatic organisms, as this influences their bio-availability. Water quality guidelines for metals in aquatic ecosystems have typically been based on total concentrations, yet it is now well established that bio-availability, i.e. the ability to penetrate a biological cell membrane, and toxicity of metals to aquatic organisms is critically dependent on the chemical form or speciation of these metals.

Most studies of the toxicity of heavy metals to fish and other aquatic organisms have shown that the free (hydrated) metal ion is the most toxic form, and that toxicity is related to the activity of the free metal ion rather than to total metal concentration (Florence & Batley 1988). Their toxicity also can be affected by pH, hardness, alkalinity, dissolved oxygen, temperature and turbidity (SECL 1983). Duration of exposure, interaction with other toxic agents and species can affect the biological response to these toxic metals significantly, e.g. mercury and methane give rise to methyl mercury.

A discussion of speciation considerations has been provided in Section 8.3.5.16 of Volume 2. It is only noted here that guidelines based on total concentrations may be over protective, since only a fraction of the total concentration will generally be bio-available, especially in samples containing appreciable concentrations of particulate matter. Measurement of the bio-available metal is required, but this is not a trivial exercise, and a hierarchy of measurements of increasing complexity must be prescribed.

The mechanisms of metal toxicity to fish are varied, although many act as enzyme poisons. Therefore, it is difficult to assess the probable effect of a measured concentration of a metal. In pond water heavy metals can be adsorbed onto clay particles and chelated by organic matter so that they remain in solution but may not have an adverse effect on fish or crustaceans (Boyd 1990). The toxicity of heavy metals is related primarily to the dissolved, ionic form of the metal, e.g. Cu^{2+} or Zn^{2+}, rather than to absorbed, chelated or complexed forms (Boyd 1989). Svobodova et al. (1993) note that the toxic action of metals is particularly pronounced in the early stages of development of the fish.

1. Aluminium

Aluminium (Al) is amongst the most abundant naturally occurring metals. The toxicity of aluminium varies with pH and other physico-chemical properties of water. Aluminium is soluble at pH values below 6.0; a number of chemical species can be formed, the most toxic occurring at pH 5.2 to 5.8. At higher pH values, an aluminium hydroxide precipitate is formed, which can flocculate in water. According to Svobodova et al. (1993), the fully flocculated hydroxide has a low toxicity, similar to that of suspended solids in general.

In freshwater, aluminium can cause problems for aquarium fish if town supply water is used.

The speciation and bio-availability of aluminium is discussed in Section 8.3.7.

Guideline notes

At pH greater than 6.5, an aluminium guideline of less than 0.03 mg/L is recommended as the guideline (DWAF 1996), while at lower pH the tolerance is reduced so a level of less than 0.01 mg/L is recommended. Meade (1989) suggested that for saltwater, aluminium should remain below 0.01 mg/L and this is used as the recommended guideline (table 9.4.14).

Table 9.4.14 Summary of the recommended water quality guidelines for aluminium

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.03	freshwater at pH >6.5	DWAF (1996)
	<0.01	freshwater at pH <6.5	DWAF (1996)
	<0.01	saltwater	Meade (1989)
Freshwater fish	0.05	rainbow trout	Svobodova et al. (1993)
	0.003	rainbow trout	Holliman (1993)
	0.1	freshwater species	Schlotfeldt & Alderman (1995)

2. Ammonia (and total ammonia nitrogen, TAN)

Ammonia (NH_3/NH^{4+}) reaches aquaculture waters as a by-product of metabolism (respiration) by animals and by decomposition of organic wastes by bacteria. Ammonia is one of the forms of the breakdown of nitrogenous waste products (excreted from aquatic organisms); the other forms produced under aerobic conditions by nitrifying bacteria are the toxic nitrite ($N0_2^-$-N) and relatively harmless nitrate ($N0_3^-$-N).

Ammonia concentration is an indicator of the pond's water quality: the greater the ammonia the poorer the water quality (Walker 1994). Most ammonia problems occur under intensive conditions where high feeding rates combined with low dissolved oxygen levels result in significantly higher ammonia levels. However, nitrate levels also need to be considered to determine the level of nitrification that is occurring in the culture water (Section 9.4.2.2/18).

The accumulation of ammonia in the water is known to be one of the major causes of functional and structural disorders in aquaculture (Poxton & Allhouse 1982). Ammonia can be a major problem for recirculating tank systems than for ponds because they do not often contain phytoplankton and macrophytes to assimilate ammonia unless an adequately sized nitrifying filter is installed (Zweig et al. 1999).

The major source of ammonia in aquaculture waters is the direct excretion of ammonia by molluscs, fish and crustaceans. In-pond sediments can also be a major source of ammonia (Burford, pers. comm. 1999, Hargraves, pers. comm. 1998). However, Svobodova et al. (1993) state that ammonia pollution may also be a result of domestic sewage, agricultural wastes or the reduction of nitrates and nitrites by bacteria in anoxic waters, or of inorganic origin, such as industrial effluents from gas works, coking plants and power generating stations.

Ammonia toxicity is greatly affected by the water chemistry. The toxicity of total ammonia nitrogen (TAN: being the sum of ammonium [NH_4^+] + unionised ammonia [NH_3]) depends on the fraction that is unionised (i.e. NH_3), since this is the most toxic form. The ionised form, NH_4^+, may also be toxic, but only at very high concentrations (Boyd 1990). Ionised and unionised ammonia exist at an equilibrium that depends on pH, temperature and salinity. Ammonia is usually measured as TAN, thus, the above modifying factors must be known to calculate the concentration of unionised ammonia (Zweig et al. 1999). According to Svobodova et al. (1993), the lower the oxygen concentration in water, the greater the toxicity of ammonia.

SECL (1983) noted that life stage, carbon dioxide concentrations, ionic strength and alkalinity all affect ammonia toxicity. Other factors are discussed by Zweig et al. (1999).

At lower temperatures and lower pHs, more of the relatively non-toxic ammonium is present. Ammonia is 30% less toxic in seawater than freshwater at the same pH and is also less at higher dissolved oxygen concentrations (Walker 1994).

High ammonia concentrations affect bodily functions and can damage gills. Chronic exposure to ammonia increases susceptibility to disease and reduces growth (Colt & Armstrong 1979).

Ammonia is more toxic when dissolved oxygen concentrations are low; however, the toxicity decreases with increasing oxygen levels. Thus, the effect is probably nullified in fish ponds because carbon dioxide concentrations are usually high when dissolved oxygen levels are low.

A combination of high total ammonia and high pH can cause ammonia toxicity in fish and crustaceans.

Guideline notes

Safe environmental ammonia concentrations are difficult to establish because of species differences and the complexity of evaluating low-level exposures (Tomasso 1993). As there is little consensus regarding permissible levels of ammonia (e.g. proposed guideline levels for marine crustaceans vary by a factor of ten; see table 9.4.15), Zweig et al. (1999) suggest it is best to be conservative.

Schlotfeldt and Alderman (1995) suggested that for freshwater aquaculture species at pH above 8.5, an ammonium (NH_4) level <0.05 mg/L should be used whilst below pH 8.5 it should be <1.0 mg/L.

Coche (1981) suggested a level below 0.1 mg/L for farm fish, molluscs and crustaceans. Meade (1989) suggested that un-ionised ammonia levels should be maintained at <0.02 mg/L. However, according to DWAF (1996), some species have lower un-ionised ammonia requirements depending on pH and temperature:

- <0.025 mg/L cold-water freshwater farm fish at pH >8.0, at lower pH to 0.0 mg/L;
- 0.0–0.3 mg/L warm-water freshwater farm fish.

Therefore, for freshwater species, the more conservative levels suggested by DWAF (1996) are used as the recommended guidelines (table 9.4.15) whilst for saltwater species a higher level of <0. 1 mg/L is used due to the reduced toxicity of ammonia in seawater.

The suggestion of Meade (1989) for a level for TAN at <1.0 mg/L for aquaculture species is used as the recommended guideline (table 9.4.15).

See also the discussions for Nitrate (9.4.2.2/18) and Nitrite (9.4.2.2/19).

3. Arsenic

The main sources of arsenic pollution in surface waters include byproducts of mineral ore processing, tanneries and dyestuff production plants, and the burning of crude oil and coal. Arsenic is commonly used in insecticides, herbicides and wood preservatives (Zweig et al. 1999). There are also natural groundwater sources of arsenic, derived from arsenic ores and volcanic activity that can reach concentrations sufficiently high to cause human health problems (Zweig et al. 1999). It is able to accumulate in large quantities in the sediments of ponds and in aquatic organisms (Svobodova et al. 1993).

Typically, the concentration of arsenic in freshwater is less than 1 µg/L and in seawater 4 µg/L (DWAF 1996).

According to Zweig et al. (1999) arsenic speciation in water is complex. It can exist in four oxidation states depending on whether it is in oxidising or reducing conditions. Arsenic binds strongly to particulate matter (a dominant from of arsenic in natural waters), can co-precipitate with iron oxides, and under reducing conditions, can precipitate as arsenic sulfide or elemental arsenic. Arsenic also forms methylated species through microbial action (Zweig et al. 1999).

Table 9.4.15 Summary of the recommended water quality guidelines for unionised ammonia and TAN

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.025	pH >8.0 cold freshwater	DWAF (1996)
	0.0	pH <8.0 cold freshwater	DWAF (1996)
	<0.3	warm freshwater	DWAF (1996)
	<0.1	saltwater	Professional judgement
	<1.0	TAN all species	Meade (1989)
General	<0.1	farm species	Coche (1981)
	<0.02	aquaculture species	Meade (1989)
	<1.0	TAN all species	Meade (1989)
Freshwater fish	<0.05	pH >8.5	Schlotfeldt & Alderman (1995)
	<1.0	pH <8.5	Schlotfeldt & Alderman (1995)
	<0.025	pH >8.0 coldwater	DWAF (1996)
	<0.3	warmwater fish	DWAF (1996)
	<0.1	silver perch	Rowland (1995a)
	<0.02	rainbow trout	Lloyd (1992)
	<1.0	freshwater species	Lloyd (1992)
	<1.0	TAN for freshwater fish	Lawson (1995)
Marine fish	<0.01	flounder	Hutchinson et al. (1992)
	0.0125	salmonids	Shepherd & Bromage (1988)
	<0.3	bream	Wajsbrot et al. (1993)
	<0.01	safe concentration	Huguenin & Colt (1989)
Brackish water fish	<0.1	barramundi	Rimmer (1995)
		many farm species	Boyd (1990)
Freshwater crustaceans	<0.1		Lee & Wickins (1992) Wingfield pers comm
Marine crustaceans	<0.1	all penaeids	Chin & Chen (1987)
	0.13	black tiger prawn	Chien (1992)
	<0.4	prawns	Boyd & Fast (1992)
	4.1	juvenile black tiger prawns	Allen et al. (1990)
Non edible bivalves	<0.001		Hahn (1989)
Gastropods	<0.003	abalone	Fallu (1991)

As a rule, arsenic occurs in the oxidation state V, but some of it also may be present in non-stable forms (i.e. in the oxidation state III) which can rapidly be absorbed into fish and are more toxic than those V forms. As with mercury (see 9.4.2.2/15) biological (particularly bacterial) activity may lead to the formation of organic methyl derivatives of arsenic (Svobodova et al. 1993).

To a large extent, pH and redox potential determine the inorganic arsenic species present in the aquatic environment. Metabolically, arsenic interacts with many elements, among them selenium and iodine (DWAF 1996). The speciation and bioavailability of arsenic are discussed in more detail in Section 8.3.7.

Information on the toxicity of arsenic to aquatic species is limited. Existing information indicates that arsenic is relatively non-toxic to aquatic organisms, with concentrations of ~1 mg/L required to cause mortality (Zweig et al. 1999). However, arsenic is more toxic to phytoplankton, with growth being affected at levels as low as five times the background concentration (Zweig et al. 1999).

Guideline notes

Meade (1989) suggested arsenic levels remain below <0.05 mg/L for both freshwater and marine species. DWAF (1996) also recommended a level of <0.05 mg/L for freshwaters. However, Eisler (1988a) recommended a higher level (<0.19 mg/L) for freshwater life than for marine species (0.036 mg/L), suggesting that freshwater species may be more tolerant of

arsenic than saltwater species. The dataset does not provide sufficient evidence to test this hypothesis further.

Thus, the values suggested by DWAF (1996) and Meade (1989) are used as the recommended guidelines (table 9.4.16).

Table 9.4.16 Summary of the recommended water quality guidelines for arsenic

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.05	freshwater & saltwater	DWAF (1996), Meade (1989)
General	<0.05	freshwater	DWAF (1996)
	<0.05	freshwater & saltwater	Meade (1989)
	<0.19	freshwater	Eisler (1988a)
	<0.036	saltwater	Eisler (1988a)
Freshwater fish	<0.05	freshwater species	Schlotfeldt & Alderman (1995)
	<1.0	salmonid hatchery	SECL (1983)
Edible bivalves	0.03	1/100th of 48 hr EC_{50} blue mussel larvae	Seed & Suchanek (1992)

4. Cadmium

Cadmium (Cd) is a highly toxic metal that is used in a variety of industrial processes including electroplating, nickel plating, smelting, engraving and battery manufacturing (Zweig et al. 1999). Inorganic (e.g. phosphate) fertilisers, reclaimed sewage sludge, municipal sewage effluents, and zinc (and other) mine tailings are also important sources of cadmium contamination (Zweig et al. 1999). Cadmium is usually associated with zinc in surface waters, but at much lower concentrations (Svobodova et al. 1993). The predominant form in the environment is the free ion (Cd^{2+}), although it will also complex with organic matter and particulates (Dojlido & Best 1993). Unlike mercury, it does not form organometallic complexes. In anoxic sediments, cadmium will precipitate as cadmium sulfide (Zweig et al. 1999). Background levels of cadmium in natural freshwaters are usually very low, generally ranging from 0.0 to 0.13 ppb (0.00013 mg/L), while saline water levels are typically less than 0.2 ppb in estuaries (<2.0 ppb in estuarine sediments) and less than 0.15 ppb in coastal areas (<1.5 ppb in marine sediments) (Zweig et al. 1999). The speciation and bioavailability of cadmium is discussed in more detail in Section 8.3.7.

According to Svobodova et al. (1993), of the dissolved forms, those which may be toxic to fish include the free ion and various inorganic and organic complex ions. Cadmium is of particular concern to aquaculture as it bioaccumulates (DWAF 1996). Apart from an acute toxic action which is similar to that of other toxic metals (damage to the nervous system), very small concentrations of cadmium may produce specific effects after a long exposure period, especially on the reproductive organs (Svobodova et al. 1993).

Cadmium toxicity is reduced with increasing levels of calcium and magnesium in the water (i.e. the harder the water the lower the toxicity). A similar relationship exists between cadmium and alkalinity. At high water temperatures, cadmium levels increase and fish survival decreases under low dissolved oxygen conditions. Additive (synergistic) effects have been found with cadmium and copper and cadmium and mercury, while cadmium toxicity is lowered in the presence of sub-lethal concentrations of zinc (DWAF 1996).

Guideline notes

The recommended guidelines for freshwater species vary with hardness as per DWAF (1996):

- at hardness 0–60 mg $CaCO_3$/L the guideline should be 0.0002 mg/L
- at hardness 60–120 mg $CaCO_3$/L the guideline should be 0.0008 mg/L
- at hardness 120–180 mg $CaCO_3$/L the guideline should be 0.0013 mg/L
- at hardness >180 mg $CaCO_3$/L the guideline should be 0.0018 mg/L.

These are more conservative than those suggested by Schlotfeldt and Alderman (1995).

The paucity of information on saltwater species makes the recommendation of guidelines difficult, however, Meade (1989) recommended a guideline of 0.005 mg/L for hardness >100 mg $CaCO_3$/L, and 0.0005 mg/L for hardness <100 mg $CaCO_3$/L.

To remain conservative, the suggestions by DWAF (1996) and Meade (1989) are used as the recommended guidelines (table 9.4.17). The values should be lowered if dissolved oxygen concentration is low or other metal toxicants are present.

Table 9.4.17 Summary of the recommended water quality guidelines for cadmium

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.0002–0.0018 <0.005–0.0005	freshwater (see above notes on hardness) saltwater and freshwater (see above notes on hardness)	DWAF (1996) Meade (1989)
General	<0.0002–0.0018 <0.005–0.0005	freshwater (see above notes on hardness) saltwater and freshwater (see above notes on hardness)	DWAF (1996) Meade (1989)
Freshwater fish	<0.0002	salmonids	Schreckenbach (1982), Svobodova et al. (1993)
	<0.001	rainbow trout	Holliman (1993)
	<0.003	silver perch	Rowland (1995a)
	<0.1	no effect limit for salmonids	Klontz (1993)
	0.004	all freshwater aquaculture species in softwater	Schlotfeldt & Alderman (1995)
	0.012	all freshwater aquaculture species in hard water	Schlotfeldt & Alderman (1995)
Freshwater crustaceans	<0.15 <0.0011		Wingfield pers comm US EPA (1986)
Marine crustaceans	<0.15 <0.053 <0.0093	black tiger prawn black tiger prawn marine crustaceans	Chen (1985) Smith (1996) US EPA (1996)
Molluscs	<0.0005	Regardless of hardness	Zweig et al. (1999)
Edible bivalves	<0.01	1/10th of level for 50% shell growth reduction in blue mussel juveniles	Seed & Suchanek (1992)

5. Chlorine

Chlorine (Cl) is a gas. Effluents containing chlorine can be discharged from municipal and agricultural water treatment plants, swimming pools, dairies and from various industrial plants. Chlorine is also used for destroying biofouling in the water cooling systems of power stations. In the early 1970s, failures of natural sets of Pacific oysters in the Tamar estuary of Tasmania were allegedly due to large quantities of chlorine which were used in the hydro-electric plant. Low concentration of chlorine can be absorbed naturally by organic matter in the water and in sediments (Svobodova et al. 1993).

Chlorine seldom occurs in nature, but is usually found as its anion, chloride. The chlorides of alkaline and alkaline earth metals are all highly soluble in water, e.g. sodium, potassium, calcium and magnesium. Whilst chlorine is a major constituent of seawater, it is in the stable form NaCl, so while there are usually 19 000 mg/L chloride as ionised salts in seawater, this form represents no danger. Chlorides are of concern in water supplies used for aquaculture because the anions of chloride are essential for osmotic, ionic and water balance in all fishes (DWAF 1996). Chlorine commonly reacts to form toxic chloramines in solution (Zweig et al. 1999).

Both free and combined chlorine residuals are extremely toxic to fish (Tompkins & Tsai 1976). If measurable concentrations (e.g. <0.08 mg/L) of residuals are present in the water, the water should not be considered safe for holding fish. Boyd (1990) noted that actual concentration of chlorine in city water supplies may be much greater than 1 mg/L.

The toxicity of the chlorine is increased with increasing water temperatures, while toxicity decreases with increasing pH.

Ammonia can combine readily with free chlorine to form the very toxic chloramines. This is particularly a problem in enclosed systems for aquarium fish using city water supplies.

Active chlorine may affect specific parts of the fish (e.g. the skins and gills) or the whole body (i.e. when chlorine is absorbed into the blood). The systemic effect manifests itself mainly as nervous disorders (Svobodova et al. 1993).

Prawn farmers are known to use post-harvest chlorination in an attempt to eliminate potential pathogens. Since the chlorinated water is exposed to sunlight for some time, the chlorine rapidly breaks down into its non-toxic derivatives during this procedure.

Guideline notes

Meade (1989) suggested levels below 0.003 mg/L for all aquaculture species, this was supported by Pillay (1990) who suggested a level less than 0.003 mg/L for all farmed fish species. Svobodova et al. (1993) consider that prolonged exposure to active chlorine concentrations above 0.04 mg/L will be toxic to the majority of fish species, whilst Schlotfeldt and Alderman (1995) suggested the range from 0.01 to 0.03 mg/L for freshwater species (these two suggestions are an order of magnitude higher than the first two suggestions).

The lower limit (Meade 1989) is recommended as the guideline for freshwater and saltwater species (table 9.4.18).

6. Chromium

Chromium (Cr) is mostly used in plating and chrome alloy production, but is also found in pigments, paints, ceramics, textile dyes, fungicides, fireproof bricks and catalysts (Zweig et al. 1999). Chromate compounds are also used for corrosion control in heating and cooling systems (Dojlido & Best 1993).

Table 9.4.18 Summary of the recommended water quality guidelines for chlorine

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.003	freshwater & saltwater	Meade (1989)
General	<0.003	all aquaculture species	Meade (1989)
	<0.01–0.03	freshwater species	Schlotfeldt & Alderman (1995)
	<0.04	most fish species	Svobodova et al. (1993)
Freshwater fish	<0.08	general	Tompkins & Tsai (1976)
	<0.03	silver perch	Rowland (1995a)
	<0.002	rainbow trout	Forteath pers comm
	<0.002	salmonid aquaculture	DWAF (1996)
	<0.003	farmed fish	Pillay (1990)
Marine fish	<0.04	optimal	Svobodova et al. (1993)
	<0.003	farmed fish	Pillay (1990)
Brackish water fish	<0.03	barramundi	Curtis pers comm
Freshwater crustaceans	<0.03	freshwater crayfish	Wingfield pers comm

NC: Not of concern

Under reducing conditions chromium is present as the free trivalent ion (Cr^{3+}), while in oxidising conditions such as those commonly found in aquaculture operations, it is found in the hexavalent form (Cr^{6+}). In natural waters a large proportion can also be bound to suspended solids and sediment (Zweig et al. 1999). Natural background concentrations are usually below 5 ppb (0.005 mg/L) and rarely exceed 20 ppb (Dojlido & Best 1993). In surface waters, the most stable forms of chromium are the oxidation states III and VI. Cr^{3+} is poorly soluble and is absorbed readily onto surfaces, while Cr^{6+} is far more soluble and the most common form in freshwater. For this reason, maximum admissible concentrations for chromium generally are based on toxicity data for the hexavalent ion (Svobodova et al. 1993). Chromium is also of concern for aquaculture due to its ability to bioaccumulate.

The speciation and bio-availability of chromium are discussed in more detail in Section 8.3.7.

The toxicity of the hexavalent ion is greater than that of the trivalent ion (Philips 1993, Zweig et al. 1999). Calcium and magnesium levels, and pH affect the toxicity of chromium compounds to fish; at a high pH and high concentration of calcium, the toxicity of chromium is reduced compared with that in soft, acidic waters.

Svobodova et al. (1993) note that with acute poisoning by chromium compounds, the body surface of the fish is covered with mucus, the respiratory epithelium of the gills is damaged and the fish die with symptoms of suffocation.

Guideline notes

It is assumed that when not specified, authors are referring to the more toxic hexavalent ion (VI). DWAF (1996) set its target water quality range at <0.02 mg/L for freshwater aquaculture. Boyd (1990) suggested a level of <0.1 mg/L for freshwater species, while Schlotfeldt and Alderman (1995) suggested 0.05 mg/L. As bioaccumulation is a problem with chromium, the more conservative level proposed by DWAF (1996) is recommended as the guideline for both freshwater and saltwater species (table 9.4.19). In acid soft waters, the recommended guideline can be reduced to <0.002 mg/L (DWAF 1996).

Table 9.4.19 Summary of the recommended water quality guidelines for chromium

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.02	freshwater	DWAF (1996)
	<0.02	saltwater	Professional judgement
General	<0.1	freshwater species	Boyd (1990)
	<0.02	freshwater	DWAF (1996)
	<0.05	aquaculture species	Schlotfeldt & Alderman (1995)
	<0.21 (III)	general	EU 1979, US EPA 1993
	<0.011 (VI)	freshwater	EU 1979, US EPA 1993
	<0.05 (VI)	saltwater	EU 1979, US EPA 1993
Freshwater fish	<0.05	rainbow trout	Holliman (1993)
	<0.1	no effect limit for salmonids	Klontz (1993)
Edible bivalves	0.045	1/100th of 48 hr EC_{50} blue mussel embryos	Seed & Suchanek (1992)

7. Copper

Copper (Cu) is used in antifouling paints, applied to boats and submerged structures. In addition, copper is used as fungicides and algacides. These uses, as well as copper mining activities are the major source of copper contamination in the aquatic environment (Zweig et al. 1999). The most common copper species in natural waters are the free (cupric) ion (Cu^{2+}), and copper hydroxide and carbonate complexes, while it also forms strong complexes with dissolved organic matter. The latter complexes usually control the aqueous copper and/or cupric ion concentration in freshwater systems (Zweig et al. 1999). At higher pH levels, the precipitation of copper carbonate complexes may also control the aqueous copper concentration. In seawater there is evidence that complexation to solids and organic matter is less due to the high concentration of ions competing for complexation sites. In bottom sediments, copper can precipitate out as sulphides, hydroxides and carbonates (Dojlido & Best 1993). Natural background concentrations of copper in water are typically around 2 µg/L (Dojlido & Best 1993).

Copper is a micronutrient, forming an essential component of many enzymes involved in redox reactions, and is an essential trace element for plants and animals. The DWAF (1996) states that the toxicity of copper depends on the solubility and chemical species of the copper present in the water. Free cupric copper ions (Cu^{2+}) are considered most toxic, and complex forms least toxic to aquatic organisms.

Its toxicity is strongly influenced by the physico-chemical properties of the water. In water with high dissolved organic content, copper can become bound in soluble and insoluble complexes, with reduced toxicities. Zinc exacerbates toxicity of copper. In very alkaline water copper forms hydroxides of low solubility, and in waters with a high bicarbonate/carbonate concentration copper precipitates as poorly soluble or insoluble cupris carbonate. Svobodova et al. (1993) note that compounds that are slow to dissolve or are insoluble are unlikely to be taken up to any extent into the fish body, so their toxicity to fish is low.

The speciation and bio-availability of copper is discussed in further detail in Section 8.3.7.

Although copper is highly toxic to aquatic organisms, its compounds are used in fish culture and fisheries as algacides and in the prevention and therapy of some fish diseases (Svobodova et al. 1993).

Guideline notes

To protect fish, the maximum admissible copper concentration in water is in the range of 0.001 to 0.01 mg/L depending on the species of fish and physico-chemical state of the water (Svobodova et al. 1993). Tebbutt (1977) reported LD_{50}s on fish at between 0.0001–0.0002 mg/L for copper sulphate. Chen et al. (1985) and Boyd (1990) suggested a level of <0.025 mg/L for no known adverse effects on aquaculture fish, while Post (1987) suggested <0.014 mg/L for fish hatcheries. DWAF (1996) and Pillay (1990) suggested <0.005 mg/L for freshwater aquaculture, and as a general guideline, respectively. Therefore, this is recommended as the guideline (table 9.4.20). With increasing hardness and alkalinity, the tolerance level should be increased as suggested by Meade (1989):

- hardness <100 mg/L (as $CaCO_3$), copper levels should be below 0.006 mg/L
- hardness >100 mg/L (as $CaCO_3$), copper levels should be less than 0.03 mg/L]

Table 9.4.20 Summary of the recommended water quality guidelines for copper

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.005	freshwater & saltwater (see above notes on hardness)	DWAF (1996), Pillay (1990)
General	<0.005	freshwater & saltwater	Pillay (1990)
	<0.005	freshwater	DWAF (1996)
	0.001–0.01	fish species	Svobodova et al. (1993)
	<0.025	aquaculture fish no effects	Chen et al. (1985), Boyd (1990)
	<0.014	fish hatcheries	Post (1987)
Freshwater fish	<0.006	silver perch	Rowland (1995a)
	<0.03	rainbow trout	Holliman (1993)
	<0.1	rainbow trout	Schlotfeldt & Alderman (1995)
	<0.1	no effect limit for salmonids	Klontz (1993)
Brackish water fish	<0.02	barramundi fingerlings	Nowak & Duda (1996)
Freshwater crustaceans	<0.03	hard water	Swindlehurst pers comm
	<0.006	soft water	Swindlehurst pers comm
Marine crustaceans	0.1	black tiger prawn	Chen (1985)
Edible bivalves	<0.008	recommended for Sydney rock oysters	Nell & Chvojka (1992)
	<0.005	1/10th of 15 d LC_{50}s for blue mussel	Seed & Suchanek (1992)
Gastropods	<0.006	1/10th of 96 hr LD_{50} abalone	Hahn (1989)

8. Cyanide

Cyanide (CN) is used in a variety of industrial processes, in particular, those involved with metal, petroleum and mineral processing. It is a non-cumulative biodegradable poison (SECL 1983) and can form a large number of complexes with metals, with varying toxicities according to their ability to dissociate into metal and hydrocyanic acid (HCN) which is the most toxic form of cyanide. For example, the toxicity of the iron cyanide complex is low to very low to fish, but the complex cyanides of zinc, cadmium, copper and mercury are highly toxic (Svobodova et al. 1993).

Cyanide also can be present in water as simple compounds (non-dissociated HCN or simple CN^- ions). These can be very toxic or extremely toxic to fish species.

Cyanide toxicity is affected by the pH of the water: if pH is low the proportion of nondissociated HCN increases and so does the toxicity. Svobodova et al. (1993) note that

toxicity is also markedly enhanced by an increase in water temperature and a decrease in the concentration of dissolved oxygen in the water.

The mechanism of toxic action of cyanides is based on their inhibition of respiratory enzymes (Svobodova et al. 1993).

According to Klontz (1993), increased temperatures in pond water can enhance the growth of cyanogenic blue-green algae, the decomposition of which can release cyanide. It is particularly a problem in large reservoirs in which plant nutrients can flow (e.g. in agricultural run-off).

Guideline notes

Schlotfeldt and Alderman (1995) suggested a level below 0.1 mg/L for freshwater aquaculture, although the more conservative recommendation of Alabaster and Lloyd (1982) is used as the guideline. Meade (1989) suggested the hydrogen cyanide levels for all aquaculture should be below 0.005 mg/L. Published information suggests that 85% of cyanide is lost from seawater within 16 hours due to volatility of the chemical (Hefter & Longmore 1984), suggesting it may be of little concern to saltwater aquaculture species. However, to be conservative the suggestion of Meade (1989) is used as the recommended guideline for all aquaculture (table 9.4.21).

Table 9.4.21 Summary of the recommended water quality guidelines for cyanide

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.005	freshwater & saltwater	Meade (1989)
General	<0.005	freshwater	Alabaster & Lloyd (1982)
	<0.1	freshwater	Schlotfeldt & Alderman (1995)
	not of concern	saltwater	Hefter & Longmore (1984)
	<0.005	all aquaculture	Meade (1989)
Freshwater fish	0.03–0.5	freshwater species	Svobodova et al. (1993)
	<0.02	no known adverse effects	DWAF (1996)
	<0.005	salmonid hatchery	SECL (1983)
	<0.005	rainbow trout	Forteath pers comm

9. Fluorides

Very little reference was made in the scientifitc literature examined for this report on the effects of flourides on aquaculture species. It has been reported that city water supplies can cause problems for aquarium fish due to high levels of fluorides (Datodi, pers. comm.).

Guideline notes

Tebbutt (1977) suggested that the safe level for freshwater fish was 0.2 to 1.0 mg/L. This is recommended as the guideline for freshwater aquaculture (table 9.4.22). Insufficient information is available to set a recommended guideline for saltwater aquaculture. Therefore, the guidelines for ecosystem protection should be used.

Table 9.4.22 Summary of the recommended water quality guidelines for fluoride

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.2	freshwater	based on Tebbutt (1977)
	ND	saltwater	
Freshwater fish	0.2–1.0		Tebbutt (1977)
Edible bivalves	<0.025	blue mussel	Pankhurst et al. (1980)
	<30.0	Sydney rock oyster spat 20% growth reduction	Nell & Livanos (1988)

ND: Not determined — insufficient information

10. Hydrogen sulphide

Sulphide is the -II oxidation state of sulphur and can exist in solution as un-ionised hydrogen sulphide gas (H_2S) or as soluble sulphides (S_2^-). H_2S is produced by bacteria in oxygen depleted (anoxic) conditions. It can be found in source water taken from ground water, and anoxic areas of surface water. It is of great concern to aquaculture as it is very toxic to fish (Zweig et al. 1999). H_2S is also present in industrial effluents, including those from metallurgical and chemical works, pulp and paper plants and tanneries.

Under anaerobic conditions, certain heterotrophic bacteria can use sulphate and other oxidised sulphur compounds in metabolism which results in the release of hydrogen sulphide (Boyd 1989). It can escape (with other gases, e.g. methane and carbon dioxide) from rich organic mud and bubble into the overlying waters. Un-ionised hydrogen sulphide is a highly toxic gas, however, the ionic forms, have no appreciable toxicity. The pH regulates the proportion of total sulphides among its forms (H_2S, HS^- and S_2^-); as pH increases, the proportion of ionised species increases and the toxicity decreases (Svobodova et al. 1993).

H_2S is often found in mangrove muds, which when disturbed (e.g. during the building of fish or prawn ponds) will become oxidised. The consequent drop in pH can lead to the mobilistion of a range of heavy metals include Al and Fe.

Guideline notes

There is a wide variation in literature for the recommended levels. Meade (1989) suggested that for aquaculture, sulphate, hydrogen sulphide and sulphur concentrations should not exceed 50 mg/L, 0.003 mg/L and 1 mg/L, respectively. The recommendation for hydrogen sulphide by Schlotfeldt and Alderman (1995) of <1.0 mg/L for freshwater aquaculture is much higher than that of other authors and is possibly for the ionic forms. According to Boyd (1989), concentrations of 0.01 to 0.05 mg/L of H_2S may be lethal to aquatic organisms, and any detectable concentration of H_2S is considered undesirable. Zweig et al. (1999) recommend that source water found to have even low levels of H_2S should not be used for aquaculture.

The DWAF (1996) suggestion is recommended as the guideline for freshwater whilst a slightly higher one is used for saltwater species (table 9.4.23) based on data for marine species.

Table 9.4.23 Summary of the recommended water quality guidelines for hydrogen sulphide

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.001 <0.002	freshwater saltwater	DWAF (1996) Professional judgement
General	<0.001 <0.003 <0.01 <1.0	freshwater all aquaculture aquatic orgamisms freshwater	DWAF (1996) Meade (1989) Boyd (1990) Schlotfeldt & Alderman (1995)
Freshwater fish	<0.002 <0.002	silver perch salmonids	Rowland (1995a) SECL (1983)
Marine fish	<0.002	Atlantic salmon	Klontz (1993)
Brackish water fish	<0.3	barramundi	Rimmer (1995)
Freshwater crustaceans	<0.1	temperate water species	
Marine crustaceans	<0.002 <0.033	black tiger prawn black tiger prawn	Lee & Wickins (1992) Chen (1985)
Gastropods	<1.0	higher is toxic to abalone	Fallu (1991)

11. Iron

In natural systems, iron can be present in two oxidation states, either the reduced soluble ferrous ion (Fe^{2+}) or the oxidised insoluble ferric ion (Fe^{3+}). The ratio of these two ions depends on the oxygen concentration in the water, pH and other chemical properties of the water. Iron is a micro-nutrient that has been shown to be occassionally limiting in seawater. It is usually found as $Fe(OH)_3$.

Soluble ferrous iron can be oxidised to insoluble ferric compounds on the alkaline surfaces of fish gills. At a low water temperature and in the presence of iron, iron-depositing bacteria will multiply rapidly on the gills and further contribute to the oxidation of ferrous iron compounds. This can give the gills a brown colour. Fish can suffocate if these compounds build up and reduce the gill area available for respiration (Svobodova et al. 1993). Ferrous iron oxidation also can affect pond productivity by taking up phosphate and restricting plankton growth.

The soluble ions may be present in bore water (artesian) in high concentrations. Upon aeration, these oxidise (and precipitate) to ferric oxide which can form crystals on the gills of fish and crustaceans. Aeration prior to use may minimise the negative effects on culture species.

Guideline notes

The level suggested by Schlotfeldt and Alderman (1995) of <2.0 mg/L total iron for freshwater aquaculture is higher than the other guidelines (e.g. <0.1 mg/L in DWAF 1996) which are for the ionic (ferrous) state. The limit of <0.01 mg/L as given in Meade (1989) is recommended as the guideline for freshwater and saltwater aquaculture (table 9.4.24). This level was also recommended for saltwater by Huguenin and Colt (1989) and Svobodova et al. (1993). The data presented in table 9.4.24 suggest that the toxicity may be higher for finfish than crustaceans and molluscs, although additional data is required to confirm this hypothesis.

Table 9.4.24 Summary of the recommended water quality guidelines for ferrous iron

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.01	freshwater & saltwater	Meade (1989)
General	<0.01	aquaculture	Meade (1989)
	<0.01	saltwater	Huguenin & Colt (1989), Svobodova et al. (1993)
	<2.0	freshwater (total iron)	Schlotfeldt & Alderman (1995)
Freshwater fish	<0.1	rainbow trout	Holliman (1993)
	<0.5	silver perch	Rowland (1995a)
	<0.01	no known adverse effects	DWAF (1996)
	<0.1	no effect limit for salmonids	Klontz (1993), Svobodova et al. (1993)
	0.01	fish hatchery	Pillay (1990)
Brackish water fish	<0.02	barramundi	Curtis pers comm
Freshwater crustaceans	<0.1	temperate freshwater crayfish	
Marine crustaceans	<1.0	black tiger prawn	Chen (1985), Lee & Wickins (1992)

12. Lead

Major sources of lead (Pb) to aquatic systems include atmospheric deposition of exhaust emissions, improper disposal of batteries, lead ore mine wastes and lead smelters, sewage discharge, stormwater runoff, and agricultural runoff from fields fertilised with sewage sludge (Zweig et al. 1999).

The lead ion (Pb^{2+}) and hydroxide species dominate at pH ~6. At higher pH, lead hydroxide and carbonate species tend to dominate. Lead forms sulfate and carbonate precipitates, while it also complexes with organic and particulate matter (Zweig et al. 1999). Concentrations of dissolved lead are generally low due to either precipitation of carbonate species or adsorption to particulates, and natural background concentrations rarely exceed 20 ppb (0.020 mg/L) (Dojlido & Best 1995). Some evidence exists for the formation of lead organometallic compounds that can bioaccumulate (Schmidt & Huber 1976). Lead largely accumulates in the bottom sediments at concentrations about four orders of magnitude greater than in the water.

The solubility of lead compounds is reduced with increasing alkalinity and pH as well as with increasing calcium and magnesium concentrations (i.e. lead is more toxic in acid soft water).

The speciation and bio-availability of lead is discussed in more detail in Section 8.3.7.

Acute lead toxicity is characterised initially by damage to the gill epithelium, the affected fish die from suffocation (Svobodova et al. 1993).

Guideline notes

Effects vary with hardness of water. Post (1987) suggested a level of <0.01 mg/L in softwater and <4.0 mg/L in hardwater. The DWAF (1996) recommendation was for no known adverse effects in soft water. The levels provided by Eisler (1988b) for changing hardness are used for the recommended guidelines (table 9.4.25 & table 4.4.2, Vol. 1):

- <0.001 mg/L at 0–60 mg/L $CaCO_3$
- <0.002 mg/L at 60–120 mg/L $CaCO_3$
- <0.004 mg/L at 120–180 mg/L $CaCO_3$
- <0.007 mg/L at >180 mg/L $CaCO_3$

These are significantly lower than the suggestion by Meade (1989) for all aquaculture species as well as the recommendations of some other authors, however it was decided to take the more conservative figure (table 9.4.25).

Table 9.4.25 Summary of the recommended water quality guidelines for lead

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.001–0.007	freshwater & saltwater (see above notes on hardness)	Eisler (1988b)
General	<0.001	freshwater	DWAF (1996)
	<0.02	all aquaculture	Meade (1989)
	<0.01–4.0	depends on hardness	Post (1987)
	<0.001–0.007	depends on hardness	Eisler (1988b)
Freshwater fish	0.004–0.008	salmonids	Svobodova et al. (1993)
	<0.03	silver perch	Rowland (1995a)
	<0.03	rainbow trout	Forteath pers comm, Schlotfeldt & Alderman (1995)
	<0.01	rainbow trout	Holliman (1993)
	<0.1	no effect limit for salmonids	Klontz (1993)
Non edible bivalves	<0.02	1/10th of levels for 50% reduction in juvenile blue mussel shell growth	Seed & Suchanek (1992)

13. Magnesium

Magnesium is a major component in the hardness of water along with calcium (see 9.4.2.1/7) However, little data were found in the literature discussing its effects on aquaculture species.

Guideline notes

Meade (1989) recommended that magnesium not exceed 15 mg/L for all freshwater aquaculture species. No information is available for saltwater species, so no guideline is provided (table 9.4.26).

Table 9.4.26 Summary of the recommended water quality guidelines for magnesium

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<15 ND	freshwater saltwater	Meade (1989)
General	<15	freshwater only	Meade (1989)
Freshwater fish	<15–20	fish hatchery	Pillay (1990)

ND: Not determined — insufficient information

14. Manganese

Manganese is used in a number of industries, producing alloys, pigments, glass, fertilisers and herbicides. It can be found in several oxidation states, namely -III, -I, O, I, II, III, IV, V, VI and VII. It is an essential micronutrient for vertebrates but is neurotoxic in excessive amounts. At typical concentrations encountered in surface waters, manganese has aesthetic rather than toxic effects as it produces a slight green discolouration of the water (DWAF 1996). The oxidised form, Mn^{4+}, is far less soluble than the reduced form, Mn^{2+}. If high concentrations of reduced manganese are present in source water, it will oxidise and precipitate causing similar problems as iron (see 9.4.2.2/11; Zweig et al. 1999).

Typically, the median concentration of manganese in freshwater is 8 µg/L (range 0.02 to 130 µg/L) and 2 µg/L in sea water. However, DWAF (1996) notes that manganese concentrations in the mg/L range can be found in anaerobic bottom waters where manganese has been mobilised from the sediments.

Guideline notes

Tolerance to manganese depends on total water chemistry, such as pH. Schlotfeldt and Alderman (1995) suggested a range between 0.1 and 8.0 mg/L, while DWAF (1996) suggested <0.1 mg/L for freshwater aquaculture. Meade (1989) and Zweig et al. (1999) recommended that manganese not exceed 0.01 mg/L for all aquaculture species, and this is the guideline recommended here (table 9.4.27).

Table 9.4.27 Summary of the recommended water quality guidelines for manganese

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.01	freshwater & saltwater	Meade (1989), Zweig et al. (1999)
General	<0.01 0.1–8.0 <0.1	freshwater & saltwater freshwater freshwater	Meade (1989), Zweig et al. (1999) Schlotfeldt & Alderman (1995) DWAF (1996)
Freshwater fish	<0.01 <0.02 <5.0	silver perch rainbow trout fish hatchery	Rowland (1995a) Holliman (1993) Pillay (1990)

15. Mercury

Mercury (Hg) naturally occurs in the environment due to the volcanic degassing of the Earth's crust and weathering of mercury rich geology (Zweig et al. 1999). While naturally high background concentrations of mercury may occur in areas rich in mercury ores, the most significant causes of aquatic contamination occur through industrial processes, agriculture and the combustion of fossil fuels. Common sources include caustic soda, pulp and paper production and paint manufacturing (Zweig et al. 1999). In most cases, background levels in unpolluted waters will contain trace amounts of mercury which do not exceed 0.0001 mg/L (Svobodova et al. 1993, Zweig et al. 1999).

The bioavailability of mercury is discussed in more detail in Section 8.3.7.

As mercury readily accumulates in sediments, surface water concentrations are not a true representation of the actual total amount of mercury present. Elementary mercury and its organic and inorganic compounds can undergo methylation (a process induced by the activity of microorganisms) in the sediments. According to Svobodova et al. (1993), the toxic end-product of this methylation — methyl mercury — enters the food chains and bioconcentrates in increasing amounts in aquatic organisms up the food chain. They give a recommended safe level of 0.0003 mg/L with organic mercury compounds for fish in general.

Mercury can be taken up by fish from food via the alimentary tract; the other routes are through the gills and skin. Through the bioaccumulation process, carnivorous fish contain the highest amounts of mercury because they form the final link in the aquatic food chain (Svobodova et al. 1993). Aquatic invertebrates can also accumulate mercury to high concentrations (Zweig et al. 1999).

Mercury compounds may damage vital tissues and organs, including gills, liver, kidney, brain and skin in fish and also may have a harmful effect on reproduction.

Guideline notes

Recommendations vary significantly between authors. A low level of 0.00005 mg/L for freshwater is suggested by Schlotfeldt and Alderman (1995), while higher limits of <0.001 mg/L (Boyd 1990) and 0.02 mg/L (Meade 1989) have been suggested for all aquaculture species. For both freshwater and saltwater species, the median level of <0.001 mg/L is selected as the recommended guideline (table 9.4.28).

Table 9.4.28 Summary of the recommended water quality guidelines for mercury

Group	Guideline mg/L	Comments	Reference
Recommended guideline	<0.001	freshwater & saltwater	Professional judgement
General	<0.0005	freshwater	Schlotfeldt & Alderman (1995)
	<0.001	all aquaculture species	Boyd (1990)
	<0.02	all aquaculture species	Meade (1989)
Freshwater fish	0.001	salmonids	Svobodova et al. (1993)
	<0.002	silver perch	Rowland (1995a)
	<0.01	rainbow trout	Holliman (1993)
	<0.001	no known adverse effects	DWAF (1996)
Marine crustaceans	<0.0025	black tiger prawn	Chen (1985)
Edible bivalves	<0.00004	1/10th of level for 50% shell growth reduction in juvenile blue mussels	Seed & Suchanek (1992)

16. Methane

The reduction of organic matter under anaerobic conditions can cause the frequent release of bubbles which rise from sediments through the water column. This gas is mostly methane, although several other gases also can be formed including hydrogen sulphide, nitrogen, ammonia and carbon dioxide. Odourless and flammable, methane might be found in water taken from the bottom of lakes or reservoirs during summer (Zweig et al. 1999).

Guideline notes

McKee and Wolf (1963) and Boyd (1990) reported that a methane level of less than 65 mg/L had no effects on freshwater and marine fish so this level is recommended for freshwater and saltwater culture (table 9.4.29), although there is a paucity of information on the effects on different species.

Table 9.4.29 Summary of the recommended water quality guidelines for methane

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<65	freshwater & saltwater	McKee & Wolf (1963), Boyd (1990)
Freshwater fish	<65	no effects	McKee & Wolf (1963), Boyd (1990)
Marine fish	<65	no effects	McKee & Wolf (1963), Boyd (1990)

17. Nickel

Nickel (Ni) contaminates surface waters through effluents from metal plating industries and ore processing facilities, while it is also emitted by the combustion of petroleum products and used to manufacture batteries (Zweig et al. 1999).

The dominant form of nickel in aquatic systems is the free ion, Ni^{2+}. It forms strong complexes with humic acids and adsorbs well to particulate matter (Zweig et al. 1999). However, it natural waters it is predominantly in dissolved form (Dojlido & Best 1993). Typical background concentrations of nickel in surface waters range from 1–3 ppb (0.001–0.003 mg/L), with concentrations up to 50 ppb (0.05 mg/L) in industrialised areas (Dojlido & Best 1993).

Nickel compounds are of medium toxicity to fish according to Svobodova et al. (1993). Their toxicity is influenced markedly by the physico-chemical properties of the water, especially hardness (the toxicity is increased in soft waters).

The speciation and bioavailability of nickel is discussed in more detail in Section 8.3.7.

After toxic exposure to nickel compounds, the gill chambers of fish are filled with mucus and the lamellae are dark red in colour (Svobodova et al. 1993).

Guideline notes

The toxicity of nickel depends on hardness with the highest toxicity in soft waters. As the little information available varies markedly, the recommended guideline is that suggested by Meade (1989), of <0.1 mg/L for all aquaculture species (table 9.4.30).

18. Nitrate

Nitrate is the least toxic of the major inorganic nitrogen compounds (Zweig et al. 1999). As it is the end-product of the nitrification process, the concentration of nitrate is generally higher than both ammonia and nitrite (Zweig et al. 1999). The main sources of nitrate pollution in surface waters are the use of nitrogenous fertilisers and manures on arable land leading to diffuse inputs, and the discharge of sewage effluents from treatment works (Svobodova et al. 1993).

Table 9.4.30 Summary of the recommended water quality guidelines for nickel

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.1	freshwater & saltwater	Meade (1989)
General	<0.1	all aquaculture species	Meade (1989)
Freshwater fish	0.02	trout	Schlotfeldt & Alderman (1995)
	0.134	LOEC at 33 mg/L $CaCO_3$, pH 7	Atchinson et al. (1987)
	0.024	LOEC at 28 mg/L $CaCO_3$, pH 7.3	Atchinson et al. (1987)
	8	salmonid 96 h LC_{50} — softwater	EIFAC (1984)
	50	salmonid 96 h LC_{50} — hardwater	EIFAC (1984)
Edible bivalves	<0.02	1/10th of level for 50% reduction in shell growth in juvenile blue mussels	Seed & Suchanek (1992)

Nitrate is not recognised generally as being toxic to aquatic animals (SECL 1983). However, high nitrate concentrations (i.e. much higher than toxic concentrations of ammonia or nitrites) can impair osmoregulation and oxygen transport (Lawson 1995). As nitrate is the major plant-limiting nutrient in seawater (most phytoplankton grow well at a nitrogen:phospohorus ratio of 10:1), so high nitrate levels can result in eutrophication and excessive nuisance algal and plant growth (Zweig et al. 1999). This can have negative effects on culture species and can result in deaths due to changes in oxygen/carbon dioxide levels. CCME (1993) recommended that nitrate levels that stimulate prolific weed growth should be avoided.

Schlotfeldt and Alderman (1995) suggested that increasing nitrate levels signals organic pollution, and measures should be taken to reduce this input. However, high nitrate levels can be a sign that nitrification (conversion of ammonia to nitrate by certain bacteria) is occurring which is helping to reduce the levels of toxic ammonia (Burford, pers. comm. 2000).

Nitrate is known to accumulate to high levels in recirculation systems as an end-product of nitrification. Through the process of denitrification it can be converted to N_2 gas, so high nitrate levels can indicate that denitrification is not occurring.

High nitrate levels (e.g. >50 mg/L) could be a potential problem under conditions of low dissolved oxygen and high pH, both of which could be further lowered by an algal bloom stimulated by the excess nitrate.

Guideline notes

Coche (1981) and Pillay (1990) recommended a level of <100 mg/L for farmed fish, molluscs and crustaceans, and this level is used as the guideline for saltwater species (table 9.4.31). However, it should be noted that nitrate levels around 100 mg/L could be a danger under conditions of low oxygen and high pH since it could be reduced to ammonia (9.4.2.2/2).

Meade (1989) was much more conservative than all the species-specific levels, and suggested a level of 3.0 mg/L for aquaculture. However, a higher level of <50 mg/L is recommended for freshwater (table 9.4.31), as suggested by Schlotfeldt and Alderman (1995).

See also discussion under Ammonia (9.4.2.2/2), Nitrite (9.4.2.2/19) and Phosphates (9.4.2.2/20).

Table 9.4.31 Summary of the recommended water quality guidelines for nitrate

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<50 <100	freshwater saltwater	Schlotfeldt & Alderman (1995) Coche (1981), Pillay (1992)
General	<50 <100 <3.0	freshwater saltwater all aquaculture species	Schlotfeldt & Alderman (1995) Coche (1981), Pillay (1992) Meade (1989)
Freshwater fish	<100 <20 <300 <400	silver perch rainbow trout no known adverse effects on fish tolerated	Rowland (1995a) Svobodova et al. (1993) DWAF (1996) Muir (1982)
Brackish water fish	<100	barramundi	Curtis pers comm
Freshwater crustaceans	<100	freshwater crayfish	Wingfield pers comm
Marine crustaceans	100–200	black tiger prawns	Lee & Wickins (1992)

19. Nitrite

Nitrite is an intermediate product in the conversion of ammonia to nitrate, a process known as nitrification. Nitrite is usually rapidly converted to nitrate, thus, high concentrations are uncommon in most aquatic systems (Zweig et al. 1999). Nitrite is rarely a source water problem, and is of more concern during the operation of recirculating systems where the water is continually reused (Lawson 1995). However, in prawn ponds, nitrite levels may increase to quite high levels at times. This appears to be a problem, particularly in tropical regions, although the cause is unclear (Burford pers comm 2000).

Nitrite toxicity results in a reduction of the activity of haemoglobin; this can be toxic to finfish or crustaceans. The brown blood disorder in fish is where haemoglobin is converted into meta-haemoglobin.

According to Schwedler et al. (1985) the following factors affect nitrite toxicity: chloride concentration in the water, pH, animal size, previous exposure, nutritional status, infection and dissolved oxygen concentration. SECL (1983) suggest that the presence of calcium, size of fish and pH also affect nitrite toxicity.

Nitrites as a rule are found together with nitrates and ammonia nitrogen in surface waters, but their concentrations are usually low because of their instability (Svobodova et al. 1993, Zweig et al. 1999). They are readily oxidised to nitrate or reduced to ammonia, both chemically and biochemically by bacteria. If levels are increasing, it is a sign of organic pollution (Schlotfeldt & Alderman 1995).

The amount of nitrite tolerated by fish is related to the chloride content of the surrounding water (Tomasso et al. 1980). Brackish water has a higher concentration of calcium and chloride which tend to reduce nitrite toxicity, although high ammonia concentrations can increase the toxicity. Svobodova et al. (1993) claimed it was necessary to measure the ratio of chloride to nitrite when estimating the safe nitrite concentration for particular locations.

Most freshwater fish actively transport nitrite from the environment using the chloride uptake mechanism located on the chloride cells of the gills (Tomasso 1993).

Guideline notes

Coche (1981) and Meade (1989) both suggested a level of <0.1 mg/L for farmed fish, molluscs and crustaceans and this level is recommended as the guideline for both saltwater and freshwater species (table 9.4.32). However, a higher level of <0.2 mg/L for freshwater is

suggested by Schlotfeldt and Alderman (1995). With increasing temperature the tolerance levels should be decreased. Tolerance levels are lower in soft waters (see table 9.4.32).

See also discussion under Ammonia (9.4.2.2/2), Nitrate (9.4.2.2/18) and Phosphates (9.4.2.2/20).

Table 9.4.32 Summary of the recommended water quality guidelines for nitrite

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.1	freshwater & saltwater	Coche (1981), Meade (1989)
General	<0.1	freshwater & saltwater	Coche (1981), Meade (1989)
	<0.2	freshwater & saltwater	Schlotfeldt & Alderman (1995)
Freshwater fish	<0.1	rainbow trout soft water	Forteath pers comm
	<0.2	rainbow trout hard water	Forteath pers comm
	<4.0	silver perch	Francis pers comm
	<0.05	no known adverse effects	DWAF (1996)
	0.06–0.25	warmwater species	DWAF (1996)
	<0.01	salmonid - soft water	Pillay (1990)
	<0.1	salmonid - hard water	Pillay (1990)
Marine fish	<0.3	flounder	Hutchinson et al. (1992)
Brackish water fish	<0.1	barramundi	Curtis pers comm
Freshwater crustaceans	<0.5	all species	Lee & Wickins (1992)
Marine crustaceans	<0.2	black tiger prawn	Lee & Wickins (1992)
	<1.0	black tiger prawn	Chien (1992), Chen (1985)
	<4.5	black tiger prawn and post larvae	Boyd (1990)

20. Phosphates

Phosphate is a generic term for the oxy-anions of phosphorus, namely ortho-phosphate (PO_4^{3-}), hydrogen phosphate (HPO_4^{2-}) and dihydrogen phosphate ($H_2PO_4^-$). These three ions exist in equilibrium with each other, the position of the equilibria is governed by pH.

Phosphate is not generally recognised as toxic to aquatic organisms. However, it is an important plant nutrient which can assist in stimulating the growth of nuisance organisms, particularly algae in fresh and brackish waters. SECL (1983) recommend that levels in salmonid hatcheries should be kept below 0.025 mg/L.

In Australia, algal blooms are consistently recorded from freshwater when total phosphate levels are over 0.1 mg/L. Research in NSW has shown that local marine algae are nitrogen limited, so it would seem unlikely that phosphate levels would influence bloom culture (Semple pers comm 2000).

High levels of phosphates may result from the use of superphosphate and other fertilisers for agricultural purposes in the catchment. High levels may be present in ponds or tanks through the addition of inorganic fertilisers to assist in promoting microalgal growth for food for zooplankton which, in turn, acts as a feed source for larval fish, molluscs and crustaceans.

Guideline notes

Schlotfeldt and Alderman (1995) suggested a range of 0.6 to 1.0 mg/L for freshwater species. The lower level (<0.1 mg/L) suggested by DWAF (1996) for freshwater farm species is recommended as the guideline as it more closely matches the species specific data (table 9.4.33). For saltwater species the recommended guideline is <0.05 mg/L as this is the most sensitive level for marine fish.

Table 9.4.33 Summary of the recommended water quality guidelines for phosphates

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.1 <0.05	freshwater saltwater	DWAF (1996) Professional judgement
General	<0.1 0.6–1.0	freshwater freshwater species	DWAF (1996) Schlotfeldt & Alderman (1995)
Freshwater fish	<0.1 <0.2	in soft water hard water	Forteath pers comm
Marine fish	<0.05		Swindlehurst pers comm
Brackish water fish	<0.1	barramundi	Curtis pers comm
Freshwater crustaceans	<0.1–0.2		Wingfield pers comm
Marine crustaceans	<0.5 <0.1–0.2		Swindlehurst pers comm Burford pers comm (2000)

See also discussion under Ammonia (9.4.2.2/2), Nitrate (9.4.2.2/18) and Nitrite (9.4.2.2/19).

21. Selenium

Selenium (Se) is an essential element that can be very toxic at low concentrations. The principle sources of selenium in the environment are the burning of fossil fuels and cement production (Dojlido & Best 1993). It exists in a variety of oxidation states, and the most common forms in the environment are selenites and selenates. They possess similar behaviour as sulfites and sulfates (Zweig et al. 1999). The breakdown of organic matter containing selenium results in the formation of organoselenium compounds (Zweig et al. 1999). Natural background concentrations of selenium are typically 0.1 ppb (0.0001 mg/L) (Dojlido & Best 1993). Selenium is of little toxicological concern for marine organisms, and it has been suggested that it may even aid in detoxifying accumulated mercury (Philips 1993).

The speciation and bioavailability of selenium is discussed in more detail in Section 8.3.7.

Guideline notes

Very little data was found for this contaminant. Whilst the USEPA (1993) recommended more conservative values, the recommended guideline is that as suggested by Meade (1989) for all aquaculture species, below 0.01 mg/L (table 9.4.34).

Table 9.4.34 Summary of the recommended water quality guidelines for selenium

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.01	freshwater & saltwater	Meade (1989)
General	<0.01 0.005 0.071	all aquaculture species freshwater saltwater	Meade (1989) US EPA (1993) US EPA (1993)

22. Silver

The major sources of silver (Ag) include ore processing, photography, dentistry and electronics. It is associated with industrialised areas and wherever human beings are located, and is actually a reliable tracer for sewage (Zweig et al. 1999).

The common forms of aqueous silver under aerobic conditions are the free ion (Ag^+) in freshwater and silver chloride complexes in saltwater (Stumm & Morgan 1996). It can also precipitate as silver sulfide, silver oxide, silver chloride and silver nitrate (Dojlido & Best 1993).

Silver is highly toxic to aquatic life, however, the toxicity is dependent upon which salt is present (Zweig et al. 1999). Silver nitrate exhibits the greatest toxicity, followed by silver chloride and iodide, sulfide and thiosulfate (Zweig et al. 1999). Mortality and altered hatching of rainbow trout has been reported at silver concentrations as low as 0.0005 mg/L (Mance 1987). Molluscs (e.g. oysters) are known to accumulate silver rapidly but depurate slowly, and as such, should not be cultured in areas where elevated silver concentrations exist (Zweig et al. 1999).

Guideline notes

Very little data was found for this contaminant. Whilst Maryland DoE (1993) recommended different guidelines for freshwater and saltwater species, both of which were more conservative than that suggested by Meade (1989) for all aquaculture species, i.e. <0.003 mg/L, and this is used as the recommended guideline (table 9.4.35).

Table 9.4.35 Summary of the recommended water quality guidelines for silver

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.003	freshwater & saltwater	Meade (1989)
General	<0.003	freshwater & saltwater	Meade (1989)
	0.00012	freshwater	Maryland DoE (1993)
	0.0023	saltwater	Maryland DoE (1993)

23. Sulphide — see Hydrogen sulphide (9.4.2.2/10)

24. Total ammonia nitrogen (TAN) — see Ammonia (9.4.2.2/2)

25. Tin and tributyltin

The major sources of tin (Sn) include processing ore and manufacturing of paint and rubber products, while the major source of organotin is the use of tributyltin (TBT) as an antifouling agent for boats and submerged structures (Zweig et al. 1999). It can also derive from plastics industries where it is used as a catalyst, fungicide and disinfectant (Dojlido & Best 1993), as well as tin-based molluscicides (Acosta & Pullin 1991).

Tin hydroxide complexes predominate in natural waters under aerobic conditions (Mance et al. 1988). In natural waters, TBT remains in a slowly degrading form, retaining some of its toxic properties, which accumulates in sediments (Lloyd 1992). Typical natural background concentrations of tin rarely exceed 2 ppb (0.002 mg/L) (Durum & Haffty 1961), while levels of organotins should be negligible unless contamination exists. TBT contamination occurs largely in the marine environment, although will occur anywhere there exists significant boating activity (e.g. marinas; Lloyd 1992).

Tin is moderately toxic to aquatic organisms (Philips 1993), however, organotin compounds are very toxic and are of major concern to aquaculture (Zweig et al. 1999) As a result of their high toxicity and ability to bioaccumulate (Dojlido & Best 1993), organotins have been banned in most states of Australia for use as an antifoulants on vessels smaller than 20–25 m in length.

Guideline notes

Toxic effects of tin have been observed at a concentration of 2 mg/L for fish (Liebman 1958, as cited by Zweig et al. 1999).

For the highly toxic organotins, sediments containing TBT at a concentration of 1 ppb (0.001 mg/kg) were reportedly toxic to clams (Furness & Rainbow 1990).

An environmental quality standard for fish for organotins of 0.00002 mg/L (0.02 ppb) was recommended by Zabel et al. (1988, as cited by Zweig et al. 1999). Standards for TBT in source water have been proposed by Maryland DoE (1993), being <0.000026 mg/L (0.026 ppb) for freshwater and <0.00001 mg/L (0.01 ppb) for saltwater and these are used as the recommended guidelines (table 9.4.36).

Table 9.4.36 Summary of the recommended water quality guidelines for organotins/tributyltin

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.000026 <0.00001	TBT in freshwater TBT in saltwater	Maryland DoE (1993) Maryland DoE (1993)
General	<0.000026 <0.00001	freshwater saltwater	Maryland DoE (1993) Maryland DoE (1993)
Fish	<0.00002	organotins	Zabel et al. (1988, as cited by Zweig et al. 1999)
Edible bivalves	<0.000005 <0.0002 0.00002	Sydney rock oyster Pacific oyster (as TBT acetate) 1/10th of level for 50% reduction shell growth blue mussel juveniles	Nell & Chvojka (1992) Alzieu (1986) Seed & Suchanek (1992)

26. Vanadium

The speciation and bioavailability of vanadium is discussed in Section 8.3.7.

Guideline notes

No data for the species groups is available, so Meade (1989) suggested level of below 0.1 mg/L for aquaculture is used as the recommended guideline (table 9.4.37).

Table 9.4.37 Summary of the recommended water quality guidelines for vanadium

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.1	freshwater & saltwater	Meade (1989)

27. Zinc

Zinc (Zn) enters surface waters primarily as a result of discharges from metal treatment plants, chemical plants and foundries (Dojlido & Best 1993), while mining can also be a major source (Zweig et al. 1999).

In low alkalinity waters, the predominant forms of zinc are the free ion (Zn^{2+}) and hydroxide complexes, while carbonate and sulfate complexes dominate in high alkalinity waters (Zweig et al. 1999). At high pH Zinc can precipitate as zinc hydroxide and coprecipitate with calcium carbonate (Dojlido & Best 1993). Zinc also forms complexes with organic and particulate matter. Natural background concentrations of zinc are generally low, ranging from 5 to 15 ppb (0.005 to 0.015 mg/L)(Moore & Ramamoorthy 1984).

The speciation and bioavailability of zinc is discussed in more detail in Section 8.3.7.

Zinc toxicity is synergistic with copper, and zinc is more toxic in soft water (Lloyd 1992). Rainbow trout are specially sensitive to zinc toxicity, resistance increasing with age. Svobodova et al. (1993) considered that avoiding the use of galvanised pipes for the supply of water and avoiding the use of galvanised containers and equipment, especially in soft and acid waters, is the best remedy to avoid frequent occurrences of zinc toxicity in rainbow trout culture.

The clinical symptoms of zinc poisoning in fish are similar to those found for copper (i.e. gill damage, reduced growth and kidney damage).

Guideline notes

Post (1987) levels at less than 0.01 mg/L for softwater and <0.15 mg/L for hard water. The USEPA (1993) suggested for freshwater aquaculture a level of <0.11 mg/L, and <0.086 mg/L for saltwater aquaculture. Meade (1989), on the other hand, suggested a conservative level below 0.005 mg/L for aquaculture species and this is used as the recommended guideline (table 9.4.38).

9.4.2.3 Organic toxicants

A wide range of agricultural, industrial and domestic activities can result in organic compounds affecting aquaculture species. Organic compounds include antibiotics, oils (petroleum hydrocarbons), pesticides and polychlorinated biphenyls (PCBs).

Table 9.4.38 Summary of the recommended water quality guidelines for zinc

Group	Guideline mg/L	Comments	Reference
Recommended guideline	<0.005	freshwater & saltwater	Meade (1989)
General	<0.11	freshwater	US EPA (1993)
	<0.086	saltwater	US EPA (1993)
	<0.01	softwater	Post (1987)
	<0.15	hardwater	Post (1987)
	<0.005	aquaculture species	Meade (1989)
Freshwater fish	<0.01	rainbow trout	Svobodova et al. (1993)
	<0.1	salmonids	Klontz (1993)
	<0.01	rainbow trout	Holliman (1993)
	<0.05	silver perch	Rowland (1995a)
Marine crustaceans	<0.25	black tiger prawn	Chen (1985)
Edible bivalves	<0.006	1/10th of level for 50% reduction in shell growth blue mussel juveniles	Seed & Suchanek (1992)

1. Antibiotics and antimicrobial agents

Industries requiring the control of microbes (e.g. agriculture) may contaminate source water with unwanted antibiotics and antimicrobial agents (Zweig et al. 1999). For example, iodine is regularly used in veterinary drugs, agricultural chemicals and sanitising solutions (WHO 1989). The presence of such chemicals in source water may have adverse effects on the natural microbial communities that are essential for the health of culture species. In addition, disturbance of microbial communities can also provide ideal conditions for opportunistic pathogens (Zweig et al. 1999).

The effects of antibiotics and antimicrobials depends largely on their bioavailability. Molecules that are bound to sediments and other substrates are generally not bioavailable. Sensitive methods exist for the detection of very low levels of these agents, however, these levels may be representative of many antibiotic and antimicrobial chemical complexes that are not biologically active (Zweig et al. 1999).

Guideline notes

No data are available to provide guidelines for antibiotics and antimicrobials. However, it is recommended that due care should be taken when using such chemicals in aquaculture operations.

2. Detergents and surfactants

Surfactants are compounds which, by lowering the surface tension of water, can facilitate the formation of emulsions with otherwise immiscible liquids such as oils and fat. They are used widely in domestic and industrial operations, eg soaps, water softeners, perfumes, optical brighteners (Svobodova et al. 1993). Aquaculture species can be exposed to surfactants and the detergents that contain them through external and on-farm activities.

There are a large number of synthetic surfactants in production, and they span a wide range of chemical toxic actions for aquatic organisms. They all damage the lipid components of cell membranes and may impair gill respiratory epithelium. Surfactants are usually categorised into three groups, anionic, non-ionic and cationic. Anionic surfactants comprise such common groups as *linear alkylbenzene sulfonates* (LAS) and *alkyl ethoxylated sulfates* (AES). Non-ionic surfactants include *alcohol ethoxylates* (AE) and *alkylphenol ethoxylates* (APE). Cationic surfactants comprise quaternary ammonium compounds. Volume 2 (Section 8.3.7.21) provides further details on surfactants, including brief information on analytical methods.

The toxicity in fish is influenced by a number of biotic and, especially, abiotic factors including pH. According to Svobodova et al. (1993) older fish are more tolerant; however, the acute toxicity varies considerably between species.

Guideline notes

Due to the paucity of information is it difficult to set a suggested level for surfactants for the protection of aquaculture species. Therefore, it is recommended that the trigger values derived for the protection of aquatic ecocsystems (Vol 2, Section 8.3.7.21) are used for freshwater and saltwater farm species (table 9.4.39).

See also Section 9.4.3 for discussion on human health aspects.

Table 9.4.39 Summary of the recommended water quality guidelines for detergents and surfactants

Group	Guideline mg/L	Comments	Reference
Recommended guidelines		**LAS:**	Volume 2, Section 8.3.7.21
	0.28	freshwater	
	0.0001	saltwater*	
		AES:	
	0.65	freshwater	
	0.65	saltwater*	
		AE:	
	0.14	freshwater	
	0.14	saltwater*	

* Low reliability trigger value, for use only as an indicative interim working level.

3. Oils and greases (including petrochemicals)

As components of liquid and gaseous fuels, petroleum hydrocarbons are among the most widely processed and distributed chemical products in the world (Zweig et al. 1999). Primary sources in surface waters include runoff from roads and discharges fro industries using oil (Dojlido & Best 1993). At sea, spills from commercial or recreational shipping can cause problems for aquaculture. Mortalities and loss of production can occur, although

the major concern to aquaculture is the tainting of culture animals with off-flavours (Zweig et al. 1999).

It is generally agreed that the lighter oil fractions (kerosene, petrol, benzene, toluene and xylene) are much more toxic to fish than the heavy fractions (heavy paraffins and tars). Fish species can differ significantly in their sensitivity to these compounds — the fry of predatory fish (e.g. trout) show the greatest sensitivity to refined products. The naphthenic acids, which are acute nerve poisons, can kill fish at concentrations as low as 0.03 to 0.1 mg/L (Svobodova et al. 1993).

In general, oils of animal or vegetable origin are chemically non-toxic to aquatic life, although they can taint the flesh of food species, coat gills reducing oxygen uptake, increase BOD levels and increase maintenance of water treatment equipment in hatcheries (SECL 1983).

Guideline notes

Given the wide range of toxicities associated with the wide variety of petroleum derived oils, greases and other chemicals which can pollute aquaculture waters, SECL (1983) consider that it is difficult to develop meaningful criteria. They recommend that surface waters should be kept free of these contaminants. With regard to freshwater aquaculture species, Schlotfeldt and Alderman (1995) provided a level of <0.3 mg/L for petroleum, <0.004 mg/L for gasoil and <1.0 mg/L for benzine. A level below 0.3 mg/L is recommended as the guideline for petroleum products in freshwater aquaculture (table 9.4.40). Insufficient information was available to set a guideline for saltwater aquaculture.

See also Section 9.4.3 for a discussion on human health aspects.

4. Pesticides

Pesticide is the general term given to any chemical used to control unwanted nonpathogenic organisms (Zweig et al. 1999). Examples of pesticides include insecticides, acaricides, herbicides, algicides, and fungicides. They are used in a range industries, but predominantly in agriculture. Johnson and Finley (1980) summarised toxicity of 400 toxic chemicals to fish and aquatic invertebrates (see also Svobodova et al. 1993 and Zweig et al. 1999).

Table 9.4.40 Summary of the recommended water quality guidelines for oils and greases (including petrochemicals)

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.3 ND	freshwater (petroleum) saltwater	Schlotfeldt & Alderman (1995)
General	<0.3 <0.004 <1.0	freshwater for petroleum freshwater for gasoil freshwater for benzine	Schlotfeldt & Alderman (1995) Schlotfeldt & Alderman (1995) Schlotfeldt & Alderman (1995)
Freshwater fish	0.05 to 20	1/100th 48 hr LC_{50}	Svobodova et al. (1993)
Edible bivalves	<0.1	crude oil 1/10th 4 d LC_{50} (blue mussels)	Seed & Suchanek (1992)

ND Not determined — insufficient information

Many pesticides are highly toxic and persistent, and pose significant risks to fish and shellfish health. In addition, due to the bioaccumulative potential of many pesticides, there are risks to product quality and public health (Zweig et al. 1999). Many pesticides used today are less persistent, degrading to non-toxic forms within a few days; however, they are potentially harmful until they are degraded. Thus, the use of pesticides in aquatic farming areas should be discouraged.

Pesticides can be separated into seven categories: inorganic pesticides, organophosphorus pesticides, carbamates, urea pesticides, pyridinium pesticides, phenoxyacetic acid derivatives, and triazine derivatives (Dojlido & Best 1993). The chlorinated pesticides are usually of most concern due to their persistence and ability to bioaccumulate in fish and shellfish (Zweig et al. 1999).

Some herbicides and most pesticides have a broad action and are, therefore, also toxic to many non-target aquatic animals and micro-organisms. They can enter aquatic systems through direct spraying (to control the growth of aquatic weeds or insect pests) or indirectly through leaching and run-off from agricultural soils (DWAF 1996).

Guideline notes

A wide range of chemicals are used by primary industries in Australia and New Zealand to control animal and plant pests. There is very little information available on the effects of these chemicals on cultured species, and it was not possible to determine which chemicals pose the greatest threat to aquaculture. Available data on safe levels for aquaculture species are provided in table 9.4.41. Recommended guidelines are provided in bold, however, it should be noted that the effects vary considerably between species. It is also worthwhile consulting the guidelines for aquatic ecosystem protection (Chapter 3 of Volume 1, Volume 2).

See also Section 9.4.3 for discussion on human health aspects.

5. Phenols

Phenolic compounds include a wide variety of organic chemicals that arise from distillation of coal, wood, oil refineries, chemical plants, production of synthetic fibres, human and other organic sources, and degradation of pesticides. They also arise from naturally occurring sources and substances (SECL 1983).

Phenol is an organic compound consisting of a hydroxyl group attached to a benzene ring. Phenols are anaesthetics which affect the central nervous system of fish.

DWAF (1996) noted the following factors influence the lethal concentrations of phenols:

- with increasing temperature, the resistance of fish to phenols is increased;
- low dissolved oxygen concentrations decrease the lethal concentration of phenols;
- with increasing total hardness, phenol LC_{50} values increase substantially; and
- sensitivity to phenols increases with an increase in salinity.

These can be problems due to direct toxicity, increases in BOD and the tainting of flesh adversely affecting sales for human consumption, especially chlorophenols (which are formed from the chlorination of phenols). According to Svobodova et al. (1993), the maximum concentrations admissible for fish culture are 0.001 mg/L for chlorophenols, 0.003 mg/L for cresol, 0.004 mg/L for resorcine and 0.001 mg/L for hydroquinone.

Guideline notes

According to Svobodova et al. (1993), the maximum concentrations admissible for fish culture are 0.001 mg/L for chlorophenols, 0.003 mg/L for cresol, 0.004 mg/L for resorcine and 0.001 mg/L for hydroquinone. The more conservative suggestion by Schlotfeldt and Alderman (1995) for freshwater species is taken as the recommended guideline for phenols as a group (table 9.4.42).

Table 9.4.41 Water quality guidelines for 'safe levels' of pesticides, herbicides, etc

Chemical	Safe level (µg /L)	Species/Group	Source
2,4-D	4.0	fish	Pillay (1990)
	<0.004	**fish culture**	**Langdon (1988)**
	0.5	rainbow trout	Forteath pers comm
2,4-dichlorophenol	**<4.0**	**freshwater aquaculture**	**DWAF (1996)**
Acephate	**<4.7**	**rainbow trout**	**Forteath pers comm, Davies et al. (1994)**
Aldrin	0.003	pond aquaculture species	Lannan et al. (1986)
	<0.01	**freshwater aquaculture**	**DWAF (1996), Pillay (1990), Langdon (1988)**
Amitrole	**300.0**	**fish/salmon hatchery**	**Pillay (1990), SECL (1983)**
Atrazine	**<0.34**	**rainbow trout**	**Davies et al. (1994)**
Azinphos-methyl	**<0.01**	**freshwater aquaculture**	**DWAF (1996)**
Azodrin	**<0.01**	**black tiger prawn**	**Chen (1985)**
BP1100	**<0.2**	**black tiger prawn**	**Chen (1985)**
Butchor	**<1.0**	**black tiger prawn**	**Chen (1985)**
Carbaryl	**0.02**	**fish culture**	**Pillay (1990), Langdon (1988)**
Carbamate *	**<0.1**	**freshwater fish**	**Svobodova et al. (1993)**
Carboxylic acid derivatives	**<1.0–10.0**	**1/100th of 48 hr LC_{50}**	**Svobodova et al. (1993)**
Chlordane	**0.01**	**freshwater aquaculture**	**Lannan et al. (1986)**
	0.004	**marine aquaculture**	**Lannan et al. (1986)**
	0.010	fish culture	Boyd (1990)
	0.004	fish	Pillay (1990), Langdon (1988)
	<0.025	freshwater aquaculture	DWAF (1996)
	0.01	salmon hatchery	SECL (1983)
Chlordecone	**<0.001**	**fish**	**Langdon (1988)**
Chlorpyrifos	**<0.001**	**freshwater aquaculture**	**DWAF (1996)**
Chlorothalonil	**<0.0082**	**rainbow trout**	**Forteath pers comm**
Cyanazine	**0.0035**	**rainbow trout**	**Davies et al. (1994)**
Cypermethrin	**0.00147**	**rainbow trout**	**Davies et al. (1994)**
DDT	0.001	pond aquaculture species	Lannan et al. (1986)
	0.001	fish	Boyd (1990)
	0.003	fish	Pillay (1990), Langdon (1988)
	0.0001	freshwater aquaculture	Schlotfeldt & Alderman (1995)
	<0.0015	**freshwater aquaculture**	**DWAF (1996)**
	0.001	salmonid hatchery	SECL (1983)
	0.001	freshwater life	CCME (1993)
	0.001	rainbow trout	Forteath pers comm
Diazine †	**<1.0–10.0**	**freshwater fish**	**Svobodova et al. (1993)**
Demton	**0.01**	**pond aquaculture species**	**Lannan et al. (1986)**
	0.1	salmonid hatchery	SECL (1983)
Diazinon	**0.002**	**fish culture**	**Pillay (1990), Langdon (1988)**
	0.002	rainbow trout	Forteath pers comm
Dicamba	**200**	**salmonid hatchery**	**SECL (1983)**

Table 9.4.41 cont.

Chemical	Safe level (µg /L)	Species/Group	Source
Dieldrin	0.003	pond aquaculture species	Lannan et al. (1986)
	0.003	fish	Boyd (1990)
	<0.005	**freshwater aquaculture**	**DWAF (1996)**
	0.005	fish	Pillay (1990), Langdon (1988)
	0.003	salmon hatchery	SECL (1983)
Dalapon	**110**	**salmon hatchery**	**SECL (1983)**
Duthiocarbamates	**<0.0001**	**fish culture**	**Langdon (1988)**
Dunall OSE	**<0.1**	**black tiger prawn**	**Chen (1985)**
Diquat	**0.5**	**fish**	**Pillay (1990)**
	0.5	salmonid hatchery	SECL (1983)
	0.5	rainbow trout	Forteath pers comm
Diuron	**1.5**	**fish**	**Pillay (1990)**
Dursban	**0.001**	**fish**	**Pillay (1990)**
Endosulfan	**0.003**	**freshwater aquaculture**	**Lannan et al. (1986)**
	0.001	**marine aquaculture**	**Lannan et al. (1986)**
	0.01	black tiger prawn	Chen (1985)
	<0.003	freshwater aquaculture	DWAF (1996)
	<0.01	fish culture	Langdon (1988)
	0.003	salmonid hatcheries	SECL (1983)
Endrin	0.004	pond aquaculture species	Lannan et al. (1986)
	0.004	fish culture	Boyd (1990)
	0.003	fish	Pillay (1990), Langdon (1988)
	<0.002	**freshwater aquaculture**	**DWAF (1996)**
	0.004	salmonid hatcheries	SECL (1983)
	0.004	rainbow trout	Forteath pers comm
Fenitrothion	**<0.2**	**rainbow trout**	**Davies et al. (1994)**
Fenvalerate	**0.083**	**aquaculture**	**Eisler (1992)**
Gunthion (see also Azinphos-methyl)	**0.01**	**freshwater aquaculture**	**Lannan et al. (1986)**
	0.01	salmonid hatchery	SECL (1983)
Hexachlorobenzole	**0.00001**	**freshwater aquaculture**	**Schlotfeldt & Alderman (1995)**
Heptachlor	0.001	freshwater aquaculture	Lannan et al. (1986)
	0.001	aquaculture	Boyd (1990)
	<0.005	**freshwater aquaculture**	**DWAF (1996)**
	0.001	salmonid hatchery	SECL (1983)
Lindane	**0.01**	**freshwater aquaculture**	**Lannan et al. (1986)**
	0.004	**marine aquaculture**	**Lannan et al. (1986)**
	0.02	fish	Pillay (1990), Langdon (1988)
	4.0	fish	Boyd (1990)
	<0.015	freshwater aquaculture	DWAF (1996)
	0.08	freshwater aquaculture	Schlotfeldt & Alderman (1995)
	0.01	salmonid hatchery	SECL (1983)
Malathion	<0.1	freshwater aquaculture	Lannan et al. (1986)
	0.008	fish culture	Pillay (1990), Langdon (1988)
	0.001	black tiger prawn	Chen (1985)
	<0.1	**freshwater aquaculture**	**DWAF (1996)**
	0.1	salmonid hatchery	SECL (1983)
	0.1	rainbow trout	Forteath pers comm
Methoxychlor	**<0.03**	**freshwater aquaculture**	**Lannan et al. (1986)**
	0.03	salmonid hatchery	SECL (1983)
Mexacarbate	**0.1**	**fish**	**Pillay (1990)**
Mirex	<0.001	freshwater aquaculture	Lannan et al. (1986)
	<0.001	**freshwater aquaculture**	**DWAF (1996)**
	0.001	salmonid hatchery	SECL (1983)

Table 9.4.41 cont.

Chemical	Safe level (µg /L)	Species/Group	Source
Paraquat	**<0.01**	**black tiger prawn**	**Chen (1985)**
Parathion	**0.04**	**freshwater aquaculture**	**Lannan et al. (1986)**
	0.001	fish culture	Pillay (1990), Langdon (1988)
	<0.004	black tiger prawn	Chen (1985)
	0.04	salmonid hatchery	SECL (1983)
	0.04	rainbow trout	Forteath pers comm
Pentachloraphenate	**<0.1**	**fish culture**	**Langdon (1988)**
Pyrethrin	**<0.001**	**fish culture**	**Langdon (1988)**
Pyrethrum	**0.01**	**fish**	**Pillay (1990)**
Rotenone	10.0	fish	Pillay (1990)
	<0.008	**black tiger prawn**	**Chen (1985)**
	10.0	**salmonid hatchery**	**SECL (1983)**
Saturn	**<0.033**	**black tiger prawn**	**Chen (1985)**
Seagreen	**<0.5**	**black tiger prawn**	**Chen (1985)**
Simazine	**10.0**	**fish culture**	**Pillay (1990), Langdon (1988)**
	10.0	salmonid hatchery	SECL (1983)
Silvex	**2.0**	**fish, salmonid hatchery**	**Pillay (1990), SECL (1983)**
TCDD (see Dioxin)			
TEPP (Tetraethyl Pyrosphosphate)	**0.3**	**fish**	**Pillay (1990)**
Trichlorphon	**<0.001**	**fish culture**	**Langdon (1988)**
Toxaphene	0.005	freshwater aquaculture	Lannan et al. (1986)
	0.005	fish	Boyd (1990)
	0.01	fish	Pillay (1990), Langdon (1988)
	<0.002	**freshwater aquaculture**	**DWAF (1996)**
	0.005	salmonid hatchery	SECL (1983)
	0.008	freshwater life	CCME (1993)
Zectran (see Mexacarbate)			

Note: Bolded text identifies those values recommended as water quality guidelines, * = 1/100th of 48 hr LC_{50}; † = 1/10th of 96 hr LC_{50}

Table 9.4.42 Summary of the recommended water quality guidelines for phenols

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.0006–0.0017 ND	freshwater saltwater	Schlotfeldt & Alderman (1995)
General	<0.0006–0.0017	freshwater	Schlotfeldt & Alderman (1995)
Freshwater fish	<0.5	fish hatchery	Pillay (1990)
	<0.001	chlorophenols in fish culture	Svobodova et al. (1993)
	<0.003	cresol in fish culture	Svobodova et al. (1993)
	<0.004	resorcine in fish culture	Svobodova et al. (1993)
	<0.001	hydroquinone in fish culture	Svobodova et al. (1993)

ND: Not determined - insufficient information

There is insufficient information to set the saltwater guideline so either the freshwater level can be used or consider the recommendations for Aquatic Ecosystem protection (Chapter 3 of Volume 1; Volume 2).

See also Section 9.4.3 for discussion on human health aspects.

6. Polychlorinated biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) were used widely as industrial chemicals and are recognised as very important environmental pollutants as they are among the most environmentally persistent of organic compounds (Svobodova et al. 1993). Although their solubility in water is very low, they are readily soluble in non polar solvents and can accumulate in fats. For these reasons and the fact that PCBs can accumulate in bottom sediments and in aquatic organisms, worldwide restrictions have been in place since the early 1970s (Zweig et al. 1999).

According to Svobodova et al. (1993), PCBs present a very difficult ecotoxicological problem: there are 209 individual PCBs, each one with different toxicological properties. They are all considered to be very toxic to extremely toxic to fish, especially in their early developmental stages. The solubility, and thus, toxicity of PCBs are enhanced by increases in temperature (Zweig et al. 1999).

Guideline notes

DWAF (1996) considers that there is no known quantitative information available on PCB levels that are *safe* and do not exert adverse effects on fish. Therefore, they say that the detection of any PCB levels should be regarded as serious. Meade (1989) suggested a level of <0.002 mg/L and this is used as the recommended guideline for both freshwater and saltwater (table 9.4.43).

See also Section 9.4.3 for discussion on human health aspects.

Table 9.4.43 Summary of the recommended water quality guidelines for PCBs

Group	Guideline mg/L	Comments	Reference
Recommended guidelines	<0.002	freshwater & saltwater	Meade (1989)
Freshwater species	0.001 <0.000014 0.0000011–0.0000051	freshwater aquaculture freshwater culture species salmonids	Lannan et al. (1986) Schlotfeldt & Alderman (1995) Svobodova et al. (1993)

9.4.2.4 Pathogens and biological contaminants

As noted by Zweig et al. (1999), high concentrations of pathogenic organisms are commonly found in waters polluted by human sewage and animal wastes. Thus, a major source of contamination is sewage outfalls in populated areas and livestock facilities. Other 'natural' sources of problem organisms, including pathogens and toxic microalgae, can occur within the culture environment.

1. Algal blooms and algal toxins

Algal blooms of all types are of growing importance as water resources are under increasing use, pressure and eutrophication (addition of nutrients), and as aquaculture industries develop.

According to Zweig et al. (1999), increasing eutrophication of surface waters can cause dramatic increases in phytoplankton and aquatic macrophytes. Such a bloom can cause the water pH to rise above 10, while the collapse of the bloom and subsequent decomposition of the organic matter can result in an oxygen deficit. In addition, some algal species produce toxic substances that may affect aquatic animals as well as domestic animals and humans (table 9.4.44). Algal toxins are released into the water during the period of algal bloom, particularly when the algal cells die and decompose. The toxins can enter the aquatic animal through the gills, body surface, or through ingestion.

Table 9.4.44 Problem microalgal species in Australia and New Zealand and their effects on aquatic organisms and human consumers of aquatic foods (refer to Section 9.4.3)

Species	Adverse effect *	Source
Miscellaneous		
freshwater & marine species which form algal blooms	Anoxia, fish appetite	Handlinger (1996a)
Dinophyceae (Dinoflagellates)		
Alexandrium angustitabulatum	Proven toxin producing species, possibly conspecific with *A. minutum*	MBMB (1996)
Alexandrium catanella	PSP Proven toxin producing species	Hallegraeff (1991) MBMB (1996)
Alexandrium minutum	PSP Proven toxin producing species	Hallegraeff (1991) MBMB (1996)
Alexandrium ostenfeldii	Proven toxin producing species	MBMB (1996)
Alexandrium spp.	Possible toxin producing species (at least 27 known strains of which some strains of at least 9 species are possibly PSP-toxin producers)	MBMB (1996)
Alexandrium tamarense	Some strains produce PSP and one bloom caused a fish kill Toxicity to other cells prawn mortality	Hallegraeff (1991) Handlinger (1996a) Su et al. (1991)
Cochlodinium spp.	Fish kills Ichthyotoxins	Hallegraeff (1991) Handlinger (1996a)
Dinophysis acuminata	Produces okadaic acid which can cause DSP	Hallegraeff (1991)
Dinophysis acuta	Produces okadaic acid and dinophysis toxin-1 which can cause DSP Proven toxin producing species	Hallegraeff (1991) MBMB (1996)
Dinophysis fortii	Produces okadaic acid and dinophysis toxin-1 which can cause DSP	Hallegraeff (1991)
Gambierdiscus toxicus	CFP	Hallegraeff (1991)
Gonyaulax polygramma	Fish kills	Hallegraeff (1991)
Gymnodinium catenatum	PSP	Hallegraeff (1991)
Gymnodinium (Fouveaux) sp.	Proven toxin producing species	MBMB (1996)
Gymnodinium mikimotoi	Massive kills of benthic invertebrates and fish Gill cell toxicity, Toxicity to other cells	Hallegraeff (1991) Handlinger (1996a)
Gymnodinium sanguinenum (*spendidens*)	Oyster kills Physical fish and shellfish gill obstruction	Hallegraeff (1991) Handlinger (1996a)
Gyrodinium aureolum (may actaully be *Gymnodinium mikimoto*)	Gill cell toxicity	Handlinger (1996a)
Noctiluca scintillans	Fish irritant and consumer of roe Gill irritation	Hallegraeff (1991) Handlinger (1996a)
Ostreopsis siamensis	Possible CFP	Hallegraeff (1991)
Pfiesteria piscimorte	Ichthyotoxins	Handlinger (1996a)
Phalacroma rotundatum	Some strains produce dinophysis toxin-1 which can cause DSP	Hallegraeff (1991)

Table 9.4.44 cont.

Species	Adverse effect *	Source
Prorocentrum lima	Produces okadaic acid and dinophysis toxin-1 which can cause DSP and possibly contributes to CFP problem Proven toxin producing species	Hallegraeff (1991)
Prorocentrum minmum	Possible human poisoning from eating shellfish but not DSP or PSP	Hallegraeff (1991)
Prorocentrum spp.	Toxicity to other cells	Handlinger (1996a)
Pyrodinium bahamense	PSP	Hallegraeff (1991)
Ptychodiscus brevis (formally *Gymnodinium breve*)	NSP	UC Davis (1997) AUST/NZ??
Scrippsiella trochoidia	Fish kills	Hallegraeff (1991)
Bacillariophyceae (Diatoms)		
Chaetoceros convolutus	Fish kills	Hallegraeff (1991)
Nitzchia pseudodelicatissima	Some strains produce domoic acid — causative agent of ASP	Hallegraeff (1991)
Nitzchia pungens	ASP	Hallegraeff (1991)
Pseudo-nitzchia australis	Proven toxin producing species	MBMB (1996)
Pseudo-nitzchia fraudulenta	Proven toxin producing species	MBMB (1996)
Pseudo-nitzchia pungens	Proven toxin producing species	MBMB (1996)
Pseudo-nitzchia turgidula	Proven toxin producing species	MBMB (1996)
Rhizosolenia cf. Chunni	Shellfish kills and tainting taste of seafood Toxicity to other cells	Hallegraeff (1991) Handlinger (1996a)
salicaceous diatoms	Gill irritation	Handlinger (1996a)
Thalassiosira mala	Oyster kills	Hallegraeff (1991)
Thalassiosira spp.	Physical gill obstruction	Handlinger (1996a)
Prymnesiophyceae (Golden-brown flagellates with haptonema)		
Chryochromulina polyepis	Fish kills Ichthyotoxins	Hallegraeff (1991) Handlinger (1996a)
Phaeocystis pouchetti	Fish kills Effect on fish migration, Gill irritation	Hallegraeff (1991) Handlinger (1996a)
Prymnesium parvum	Fish kills Toxicity to other cells, Gill cell toxicity	Hallegraeff (1991) Handlinger (1996a)
Chrysophyceae (Golden-brown algae)		
Pelagococcus subviridis	Lower abundance, feedings & fecundity of crustaceans and bivalves	Hallegraeff (1991)
Raphidophyceae (Chloromonads)		
Heterosigma akashiwo	Fish kills Toxicity to other cells	Hallegraeff (1991) Handlinger (1996a)
Dictyochophyceae (Silicoflagellates)		
Dictocha speculum	Fish kills	Hallegraeff (1991)

Table 9.4.44 cont.

Species	Adverse effect *	Source
Cyanophyceae (Blue-green algae)		
Anabaena spp.	Toxicity to other cells	Handlinger (1996a)
Aphanizomenon spp.	Toxicity to other cells	Handlinger (1996a)
Microcystis	Toxicity to other cells	Handlinger (1996a)
Nodularia	Toxicity to other cells	Handlinger (1996a)
Trichodesmium erythraeum	Nuisance organism	Hallegraeff (1991)

* Human consumers may be affected by the following : ASP = Amnestic shellfish poisoning , CFP = Ciguatoxin fish poisoning, DSP = Diarrhetic shellfish poisoning, NSP = Neurotoxic shellfish poisoning, PSP = Paralytic shellfish poisoning

Note : Several other possible toxin producing species known to be present in New Zealand coastal waters are listed in MBMB (1996)

Generally it is the health of finfish which is most affected by algal blooms, although there have been some instances of kills of invertebrates by them (Hallegraeff 1991, Handlinger 1996a). Handlinger (1996a) reported seven mechanisms for algal effects on fish:

- **Anoxia**: Oxygen depletion resulting from excessive abundance of phytoplankton algae is a common cause of mortality of fish and crustaceans in aquaculture ponds (Boyd 1990). Algal blooms can readily occur in water bodies with high levels of nutrients, coupled with conducive environmental conditions (e.g. no cloud and high temperatures) (DWAF 1996). High concentrations of nutrients in aquaculture waters can result from overfeeding, agricultural fertiliser run-off and effluents from sewage treatment plants. Large scale aquatic animal kills from this problem have occurred in both freshwater and marine waters.
- **Physical gill obstruction**: Mucus producing species can clog the gills of fish and shellfish. Examples in Australia include *Thalassiosira* and the non-toxic *Gymnodinium sanguinenum* (*spendidens*).
- **Gill irritation**: Examples include *Phaeocystis pouchetti, Nitzschia* sp., *Noctiluca scintillans* and salicaceous diatoms.
- **Gill cell toxicity**: Death from the destruction of the thin gill epithelium has been caused by a number of algal species including *Gyrodinium aureolum*, *Gymnodinium mikimotoi* and *Prymnesium parvum*.
- **Toxicity to other cells**: **These** algae affect other cells after ingestion or absorption and are inherently more likely to affect other species following ingestion of the contaminated animals. This group includes the algae producing hepatotoxins, neurotoxins, haemolysins and digestive cell necrotoxins. Examples from Australia and New Zealand (table 9.4.44) include the blue-green algae of the genera *Microcystis*, *Aphanizomenon* and *Anabaena* (all freshwater) and *Nodularia* (generally brackish water), and the marine species *Alexandrium tamarenses, Gymnodinium mikimotoi, Prymnesium parvum, Rhizosolenia chunni, Prorocentrum* spp and *Heterosigma akashiwo*.

  The extent and effects of blue-green algae blooms have been summarised by Johnstone (1994). However, fish kills caused directly by the blue-green algae appear to be rare. The DWAF (1996) considers that water-borne toxins produced by blue-green algae are unable to cross the gill membranes of fish and, therefore, do not enter the circulatory system. Toxic effects can be induced when the toxin is ingested by fish or when they eat the toxin-containing algal cells.

- **Ichthyotoxins**: Includes some of the above species as well as *Cochlodinium* spp, *Chryochromulina polyepis* and *Pfiesteria piscimorte*.
- **Reduced appetite**: Virtually all algal blooms, including non-toxic species, may affect fish appetite.

The effects of these toxins on the use of aquaculture products, particularly molluscs which have bioaccumulated the toxins, and the shellfish sanitation programs to overcome these problems are covered in Section 4.3. Hallegraeff (1987, 1991) provides excellent guides and detailed descriptions of many of the marine toxic species.

Handlinger (1996a) noted that, in general, fish which swim in waters affected by algal blooms ingest or absorb very little of these algae compared with filter feeding shellfish. In summary, she wrote there was very little overlap between the marine algal toxins accumulated by shellfish, and screened in shellfish sanitation programs, and those algal blooms causing fish kills.

Guideline notes

No guidelines can be recommended as the effects vary considerably between species of microalgae and the particular culture species.

See also Section 9.4.3 for discussions on human health aspects.

2. Bacteria, viruses and parasites

Handlinger (1996b) suggested that kills due to disease pose a more direct threat to the long term future of fish (cultured animal) stocks than pollutants. Water used for aquaculture always will contain a certain number of bacteria, viruses, fungi, parasites (both Protozoan and Metazoan) and other organisms which may be harmful to aquatic organisms. Even normally harmless bacteria and viruses, under adverse environmental conditions, can contribute to impaired health of the culture species. The maintenance of optimal water quality appears to be the best defence against infections by these organisms (DWAF 1996).

In artificial environs, there are means to reduce the amount of incoming potential pathogens by, for example, inflow filters that retain particles (to which most of the bacteria will be attached). In hatcheries, inflowing water may be UV-treated or ozonised to reduce the level of infective organisms. During the design of hatcheries, nurseries and growout farms, it is very important to incorporate the ability to isolate outbreaks quickly and for procedures to correct the problem.

There are many overseas examples of fish diseases which have decimated stocks rapidly after accidental introduction to native stocks or new susceptible species. Thus unusual sudden losses due to disease are more likely to represent new diseases or introductions, and rapid diagnosis of such diseases is important if they are to be controlled (Handlinger 1996b).

Australian examples of major diseases causing fish kills noted in Handlinger (1996b) include:

- Epizootic Haematopoietic Necrosis (EHN) is the classic Australian disease cause of fish kills. It is an internationally significant disease (OIE List B), only known to occur in Australia, with the very name indicating sudden mortalities. Most outbreaks are in Redfin perch, but has also been diagnosed in small rainbow trout. The distribution within Australia is limited, which makes knowledge of the distribution at any time important for control of fish movements, to prevent the spread of EHN to uninfected areas.
- Another serious disease of limited distribution in Australia is the Goldfish atypical strain of *Aeromonas salmonicida*, which was introduced into Australian Goldfish breeding

stocks in 1970s, and has since spread in to wild Goldfish, and to some other fish species. Usually a low death rate, occasionally high level of skin lesions.

- Epizootic ulcer syndrome (EUS), Red Spot disease, Bundaberg disease etc, from a range of species. Low mortality, high morbidity (ulcers), with a wide geographic range.

Handlinger (1996b) also noted that disease may be only one component in a complex cause of death, often acting in conjunction with environmental or physiological stress factors. Examples include:

- The gill protozoan *Chilodonella* or the skin fungus *Saprolegnia sp* cause deaths in Bony Bream (*Nematalosa erebi*) when winter temperatures fall below 10°C *Chilodonella* species are also thought to be introduced to local wild stocks through imported fish.
- Winter deaths with Saprolegniasis (fungus infection of skin) in brown trout with spawning stress associated with crowded spawning grounds in Tasmania.
- Eel Saprolegniasis deaths in post capture holding facilities following capture (crowding stress).
- Septicaemia in stressed migrating lampreys after trauma from obstacles to migration.
- Large number of digenean flukes in the gills and peritoneal cavity of Galaxids dying in saline lakes in 1984. Major cause of death probably a bloom of the dinoflagellate *Glenodinium*. Dying Redfin perch from the same area showed large numbers of the bacterium *Aeromonas hydrophila.*

The identification and treatment of pathogenic problems is outside the scope of this Chapter. A wide range of literature is available on this subject, some of which is listed below:

- finfish — PGVSUS (1988, 1992, 1996), Wolf (1988), Sindermann (1990), Austin & Austin (1993), Schlotfeldt & Alderman (1995)
- molluscs — Elston (1990), Sindermann (1990)
- prawns — Lightner (1996)
- freshwater crayfish — Huner (1994)

Guideline notes

No guidelines can be recommended as the effects vary considerably between species of pathogen and the particular culture species.

In summary, a reduced level of infectious organisms will contribute to a better overall health of aquaculture animals, a reduced need to treat animals with chemicals and drugs and, thus, to lower production costs as well as a residue-free product.

See also Section 9.4.3 for discussions on human health aspects.

9.4.3 Water quality guidelines for the protection of human consumers of aquatic foods

Most aquaculture products and recreationally and commercially harvested aquatic species are destined for consumption by humans. Generally aquaculture and commercially harvested aquatic foods are considered gourmet items and attract a premium price. To maintain demand, the aquaculture and fishing industries must ensure the highest quality of these products, both from a visual and, most importantly, from a human health point of

view. The guidelines contained in this Section are intended to protect the health of human consumers of aquatic foods.

A range of chemical and biological contaminants (including bacterial and viral pathogens) are of concern (table 9.4.45). These may accumulate in the soft tissues of aquatic species through ingestion. Other toxicants can be taken up by the animals directly from the water source through passive diffusion or active uptake. While these contaminants may not be deleterious to the health of the organisms concerned, many can adversely affect human health if consumed above certain levels. Others can taint, or cause 'off-flavour', which affects the palatability of aquatic foods and lower their market acceptability.

Table 9.4.45 Chemicals and biological contaminants important for the protection of human consumers of fish and other aquatic organisms (based on University of California, Davis, website, 1997, Cunliffe pers comm and Jackson pers comm)

Contaminants	Types
Chemical contaminants	Inorganic chemicals (heavy metals, etc.)
	Organic chemicals (pesticides, etc.)
	Radionuclides (radioactive elements)
Viral contaminants	Hepatitis A
	Norwalk virus
	Parvo-virus
	Poliovirus
	Rotavirus
Bacterial contaminants	*Listeria monocytogenes*
	01 *Vibrio cholerae*
	Non 01 *Vibrio cholerae*
	Vibrio parahaemolyticus
	Vibrio vulnificus
	Vibrio mimicus
	Vibrio hollisae
	Salmonella sp
	shiga toxin producing *E. coli*
Natural Toxins	Ciguatera
	Paralytic shellfish poisoning
	Neurotoxic shellfish poisoning
	Diarrhetic shellfish poisoning
	Puffer fish toxicity (tetrodotoxins)
Parasites	*Anisakis simplex* or herring worm
	Clostridium perfringerns
	Shigella
	Enterotoxic E. coli
	Cryptosporidium sp
	Giardia sp

The food standards developed by the Australian and New Zealand Food Authority (ANZFA) and published in the *Food Standards Code* (ANZFA 1996) aim to protect consumers from eating chemically contaminated foods, including aquatic species ((also see Australian web site (www.anzfa.gov.au) and New Zealand web site (www.anzfa.govt.nz) for updated information). These are based on the notion of acceptable daily intake (ADI) or acceptable weekly intake (AWI) — see Zweig et al. (1999) for the World Health Organization (WHO) provisional tolerable weekly intake for selected elements as well as import regulations for residues.

The food standards apply to the edible portion of the organisms, so the flesh levels, not the levels in the liver, kidney or other organs which are usually higher, are specified for finfish, while the hepatopancreas levels are not included for crustaceans (although this organ is eaten by some consumers). With molluscs, the parts consumed varies from the whole animal (e.g. oysters, clams and mussels) to specific parts (e.g. abalone, scallops and cephalopods).

9.4.3.1 Physio-chemical parameters

The basic physio-chemical properties of waters, whether natural or in artificial environs, generally do not have any direct effects on the safety of human aquatic foods during the culture (growing) or harvesting processing. However, post harvest activities need to be undertaken at the appropriate temperatures to avoid spoilage of the end product.

9.4.3.2 Chemical contaminants

As detailed in table 9.4.45, chemical contaminants may be categorised into three broad groups:

- **Inorganic chemicals (mostly heavy metals)**: These are a potential problem for human health, particularly in the case of bivalve molluscs where bioaccumulation increases the concentrations of toxicants. The rate of accumulation is species specific and depends on the mechanism of absorption and tissue distribution.
- **Organic chemicals (e.g. pesticides and herbicides)**: This broad group includes synthetic compounds which through either bioaccumulation or residue concentrations are potentially toxic to human consumers of contaminated aquatic foods.
- **Radionuclides (radioactive elements)**: At present, ANZFA do not give any maximum permitted concentrations (MPCs) for radionuclides in edible tissues. Many countries have limits set on imported foods, particularly for cesium-137 (Cs-137). Environmental levels of Cs-137 are considerably lower in the southern hemisphere than in the northern hemisphere, and exporters in Australia and New Zealand should not generally experience difficulty in meeting such limits.

The ANZFA food standards should be considered as the default standards for chemical contaminants. As the standards are currently under review, readers are referred to the relevant Australian (www.anzfa.gov.au) and New Zealand (www.anzfa.govt.nz) ANZFA web sites for updated information.

Zweig et al. (1999) provide an excellent summary of the guidelines used by the United States, Canada, Japan and the European Union for a wide range of chemical contaminants residues in imported aquatic foods.

9.4.3.3 Biological contaminants

There are a number of biological contaminants which can affect human consumers of aquatic foods, including:

- bacteria;
- viruses;
- parasites;
- micro-algae (biotoxins).

For each of these groups, further discussion is provided in the next four Sections (9.4.3.3/1–4). Various approaches for prevention and management of these potential contaminations are provided in Section 9.4.3.5.

The flesh of fish and crustaceans is less susceptible to contaminations by microorganisms and biotoxins. However, filter-feeding shellfish (bivalves) can concentrate these potential contaminants to levels higher than that in the water source. Thus shellfish are considered to be a higher risk for consumers of aquatic foods, although there can be secondary problems associated with fish or crustaceans, for example poisoning from wounds inflicted when handling these animals.

Table 9.4.46 specifies guidelines on safety for human consumption for the micro-algal biotoxins which use levels of toxins in edible flesh, note that in Australia there are no standards for NSP or DSP. For bacterial organisms the guidelines for commercially harvested fish species are based on risk management programs and vary between countries (Section 9.4.3.5/2). A water quality guideline for minimising the exposure of human consumers to bacterial diseases caused by ingesting contaminated wild fish species is provided in Section 4.4.5.3 of the Guidelines (Volume 1).

Table 9.4.46 Guidelines for the protection of human consumers of shellfish and finfish from contamination by microalgal biotoxins

Toxicant	Guideline *in water*	Standard *in edible tissue*
Neurotoxic shellfish poison (shellfish only)	No guideline. Toxins may be present in the microalgae and may be accumulated in other aquatic organisms.	<20 mouse units/100 g of edible shellfish flesh [New Zealand only]
Diarrhetic shellfish poison (shellfish only)	No guideline. Toxins may be present in the microalgae and may be accumulated in other aquatic organisms.	<20 µg/100 g of edible shellfish flesh (~5 mouse units) [New Zealand only]
Paralytic shellfish poison (shellfish only)	No guideline. Toxins may be present in the microalgae and may be accumulated in other aquatic organisms.	<80 µg of saxitoxin equivalent/100 g of edible shellfish flesh (~400 mouse units) [Australia & New Zealand only]
Amnestic shellfish poison (shellfish only)	No guideline. Toxins may be present in the microalgae and may be accumulated in other aquatic organisms.	<20 µg/g of domoic acid in edible shellfish flesh [Australia & New Zealand only]
Ciguatera-like toxins (finfish only)	No guideline. Toxins may be present in the microalgae and may be accumulated in other aquatic organisms.	<20 mouse units/100 g shellfish [New Zealand only]

Source: MBMB (1996) and Jackson (pers comm).

1. Bacteria

Bacterial aquatic food borne diseases in humans can be grouped according to where the bacteria originate:

- bacteria that are present in water/sediments (e.g. *Clostridium botulinum*; *Vibrio parahaemolyticus*; other *Vibrio* spp);
- bacteria from pollution of aquatic environments with human and/or animals faeces (e.g. *E. coli* and enterotoxic species, *Clostridium perfringens*, *Vibrio cholerae*; *Salmonella typhi*; other *Salmonella* spp; *Shigella flexneri*).

Bacterial contamination of aquatic foods can occur from exposure within the aquatic environment and/or after harvest and during processing. The latter is not within the scope of this document. However, most cases of human disease (gastroenteritis) are associated with consumption of raw or undercooked molluscs which have been contaminated either immediately or shortly prior to harvest.

For commercial harvesting of shellfish the usual approach to reduce the bacterial load to control this human health hazard is two-tiered:

- risk based (i.e. it is the risk level that defines the classification status) classification of waters (Section 9.4.3.5 No.2) to allow only certain waters/times for rearing/harvesting of molluscs for human consumption, based on results of detailed sanitary surveys and an ongoing strategic monitoring program which assesses growing water and shellfish quality (this approach includes the relaying of contaminated stock to clean waters and depuration, see below);
- treatment of shellfish to render safe for consumption e.g. heat treatment or irradiation of molluscs if necessary.

Depuration was formally introduced in NSW following a food poisoning outbreak in 1978, involving over 2000 clinical cases of viral gastroenteritis which was attributed to the consumption of contaminated shellfish farmed in the Georges River (Linco & Grohmann 1980, Murphy et al. 1979). Prior to this outbreak there was little information available regarding the sanitary status of NSW estuaries. From 1978 to 1981, estuaries where oysters were farmed in NSW were sampled by regulatory authorities in an attempt to ascertain the levels of faecal contamination. On the basis of these bacteriological findings estuaries were ranked according to risk, however, the methodology used in the sampling regime resulted in errors in the ranking of estuaries, with some estuaries being sampled at low frequency and consequently ranked incorrectly (Ayres 1991). Depuration was initially introduced to estuaries identified as high risk and by 1983 depuration of all shellfish sold in NSW became a statutory requirement, regardless of the sanitary status of the estuary from where the shellfish were harvested (Jackson & Ogburn 1998). Currently depuration remains compulsory for all oysters harvested in NSW for human consumption, however, this requirement is currently being reviewed as oyster harvest areas are assessed in terms of risk and formally classified.

Depuration is a process which exploits the natural physiological mechanisms of shellfish to promote purging of the gastrointestinal tract. Shellfish are depurated in order to reduce the likelihood of transmitting infectious or other injurious agents to consumers. Depuration involves live animals and the success of the process is dependent on the well being of these animals. The efficacy of depuration may be defined as the extent to which microbial and other contaminating agents are eliminated from shellfish during the process.

According to Jackson and Ogburn (1998) the current status of depuration and the factors which affect the efficacy of the process for bacterial species (refer to 9.4.3.3 No.2 for viruses) include:

- Depuration, under appropriate operating conditions, is capable of removing many bacterial species from shellfish, including faecal coliforms.
- The water temperature, salinity and turbidity all influence the efficacy of depuration. These factors must be optimised to maintain the health status of the shellfish in order to maximise the efficacy of depuration.
- It is likely that the optimal conditions for depuration will vary between shellfish species and within a species which has been acclimatised to different environments.
- The initial pathogen load, length of exposure to the pathogen and pathogen distribution within shellfish tissues will each influence the efficacy of depuration. Generally, the efficacy of depuration is decreased when the initial pathogen load is high.
- Ultraviolet radiation as a means of water disinfection during depuration, is relatively efficient and cost-effective. Further research is required to assess methods to enhance UV disinfection and to investigate alternate methods of disinfection.
- Not all bacterial species are removed from shellfish at the same rate during depuration. It is apparent that bacteria that constitute part of the natural microbiota of the shellfish (e.g. *Vibrio* spp.) are less readily removed than introduced bacteria (e.g. *E. coli)*.

Zweig et al. (1999) provide an excellent summary of the guidelines used by the United States, Canada, Japan and the European Union for a wide range of bacteriological standards in imported aquatic foods. For further discussion on *Listeria monocytogenes* and the various *Vibrio* spp, refer to UC Davis (1997).

2. Viruses

Viruses that infect human consumers of aquatic food or diffuse sources such as on-site wastewater systems (e.g. septic tanks) are of human origin (i.e. these viruses have been shed in human faeces via sewage outlets into waters where aquatic organisms are cultured or harvested). There are more than 110 different viruses known to be excreted in human faeces, collectively known as the 'enteric viruses' (Goyal 1984). They can remain in seawater for long periods of time and have been shown to survive as long as 17 months in marine sediments (Goyal et al. 1984). UC Davis 1997 suggests that viruses that are associated with sediments are as infectious to animals as those that are freely suspended, however, the potential exposure routes were not identified. However, there is a question regarding the infectivity of these agents whilst in waters and sediments, as pieces of viral RNA/DNA detected by some methods might not actually infer the presence of viable cells, see Richards (1999) for a discussion of this issue.

Viruses have been isolated from a wide range of bivalve molluscs whose filter-feeding activities can concentrate the viruses at levels much higher than the surrounding waters. The viruses do not multiply in bivalves, but accumulate in the gastrointestinal tract, liver-like digestive gland and other tissues. The behaviour of these agents is very complex and the accumulation rate is dependent on the viral species and the species of mollusc. Crustaceans, such as crabs and lobsters, can accumulate viruses by contact with contaminated seawater and/or by consuming contaminated bivalves (Hejkal & Gerba 1981). Whilst the highest concentration of viruses are found in the inedible portions of crabs (Goyal et al. 1984), they are usually present at a level below that of the water (UC Davis 1997).

Many cases of human food poisoning outbreaks have been associated with the consumption of contaminated raw oysters. In 1978, 1989 and 1990 Norwalk virus and Parvo-virus were responsible for three major food poisoning outbreaks in Australia, while the cause of another outbreak in 1996 was unconfirmed but could have been Norwalk virus. In 1997 an outbreak of Hepatitis A was linked to the consumption of contaminated oysters.

The presence or absence of viruses is even more difficult to detect than bacteria, so indicator species are also used. Since the viruses of concern to human health are derived mainly from sewage, *E. coli* and other faecal coliforms are used as the indicator species. However, the correlation between the presence of faecal coliform and viruses is unreliable. It is also now thought that shellfish may eliminate *E. coli* from their systems without eliminating viruses, so the absence of *E. coli* in the flesh is not a satisfactory predictor of absence of viruses. Nevertheless, the use of sanitary surveys are still relevant and are used in Australia and New Zealand as well as the USA and European Union (see Section 9.4.3.5 No.2).

Heat and depuration — which work well to reduce bacterial contamination of molluscs — are not equally efficient in reducing viral loads. Heat treatment may need to take place at higher temperatures than required for bacteria. UC Davis (1997) indicate that most viruses (excluding Hepatitis A) are inactivated when the internal temperature of molluscs reaches 60°C (140°F), which requires some 4 to 6 minutes of steaming. A common cooking practice is to steam molluscs only until the shell opens, however, as this may occur after only 1 minute of steaming (UC Davis 1997), this is not sufficient time to inactivate all of the viruses.

The ability of depuration to effectively eliminate viral agents from shellfish is uncertain. It is apparent that viral agents are capable of remaining in shellfish after the depuration process, and that viral agents generally take a longer period of time compared to bacteria, to be effectively removed from shellfish (Jackson & Ogburn 1998). Further research is required in this area.

Jackson and Ogburn (1998) provide a good review on the subject. For further discussion on Hepatitis A, Norwalk virus and Poliovirus, refer to UC Davis (1997).

3. Parasites

To date there is no evidence in Australia or New Zealand of any parasites which can be passed from aquatic organisms to humans, therefore no guidelines are provided. There is little epidemiological evidence indicating an important role of shellfish in the dissemination of protozoan infections, but *Giardia* sp. and *Cryptosporidium* sp. remain possibilities (Stelma & McCabe 1992). Fayer et al. (1998) indicated that oysters can serve as mechanical vectors of the human pathogen *Cryptosporidium parvum* oocysts.

Furthermore, it should be noted that the presence of parasites, cysts and necrotic tissue resulting from parasitic infections is likely to make the product unmarketable.

4. Marine biotoxins

There are a number of marine biotoxins which represent a significant threat to human consumers of aquatic foods — they are mostly associated with microalgae, although there are some toxins which occur in species which do not involve marine algae.

Microalgal-associated toxins

A comprehensive review of this topic is provided by Hallegraeff (1991). There are five recognised types of toxins (see table 9.4.46), which are all associated with naturally occurring marine microalgae (table 9.4.44 in Section 9.4.2.4/1 provides a list of problem

species in Australia and New Zealand). The toxins can accumulate in aquatic animals when they feed on the algae or on other animals which have fed on the algae. They include:

Paralytic shellfish poisoning (PSP)

- A number of toxic dinoflagellates can be concentrated by filter feeding bivalves and become poisonous to humans, these include species of *Gonyaulax, Gymnodinium*, *Alexandrium* and *Pyrodinium.* These are often described as the 'red tide' species due to the colouring of the water when they occur in blooms, although the colour is not always red. They are found in a wide range of environments from tropical to temperate waters.
- PSP can be caused by a combination of any of 18 toxin analogues, depending on the species of dinoflagellate and geographic area (UC Davis 1997). The primary toxins include saxitoxins, gonyautoxins and derivatives.
- All filter-feeding molluscs accumulate and depurate paralytic shellfish toxins (UC Davis 1997). In the Northern Hemisphere (USA), PSP has been reported in the viscera of mackerel, lobsters and crabs (UC Davis 1997).
- In 1986 a bloom of *Gymnodinium catenatum* caused two cases of poisoning from consumption of wild mussels and oysters in Tasmania (Jackson pers comm). Routine monitoring of this biotoxin group is undertaken in New Zealand.

Diarrhetic shellfish poisoning (DSP)

- Several dinoflagellates of the *Dinophysis* and *Prorocentrum* genera have been associated with DSP (refer UC Davis 1997).
- To date eight lipid soluble toxins have been isolated, including okadaic acid, dinophysistoxins, pectenotoxins, yessotoxins and derivatives. Filter-feeding molluscs can accumulate these toxins in their hepatopancreas even at dinoflagellate concentrations below that necessary to discolour the water (UC Davis 1997).
- DSP cases have been reported from commercial harvests of wild pipis in south Ballina Beach and Stockton Beach in NSW (Jackson, pers. comm.). Routine monitoring of this biotoxin group is undertaken in New Zealand.

Amnesic shellfish poisoning (ASP)

- Diatoms from the genus *Pseudonitzschia* produce the neurotoxin known as domoic acid (an amino acid) which also accumulates in filter-feeding shellfish (Handlinger 1996b). In the Northern Hemisphere (USA) PSP has been reported in the viscera of anchovies and crabs (UC Davis 1997).
- No evidence to date of occurrence in Australia, however, routine monitoring for this biotoxin group is undertaken in New Zealand.

Neurotoxic shellfish poisoning (NSP)

- *Ptychodiscus brevis* (formally *Gymnodinium breve*) can produce three known toxins called brevetoxins: brevetoxin B, brevetoxin C and GB-3 (Yasumoto 1985).
- No evidence to date of occurrence in Australia, however, routine monitoring for this biotoxin group is undertaken in New Zealand.

Ciguatera fish poisoning (CFP)

- By ingesting toxic dinoflagellates, certain species of tropical and subtropical fish can become toxic to humans.

- The species most often associated with ciguateric fish is *Gambierdiscus toxicus*. Other algal species include *Ostreopsis* spp. and *Prorocentrum* spp.
- There are at least four known toxins: ciguatoxin (the principal toxin), scaritoxin, ciguaterin and maitotoxin (UC Davis 1997).
- The toxins appear to be concentrated in the viscera, head or central nervous system of affected fish (Tosteson et al. 1988).
- Both herbivorous and carnivorous fish can become toxic, the first group by eating the algae itself, the second group by consuming toxic herbivorous fish. Generally larger fish are more poisonous than small fish as they consume greater amounts of toxins (Graig 1980). In Australia the fish most implicated in cases of ciguatera include mackerel and barracuda. The harvest waters in New Zealand are too cold for this biotoxin group.

Other toxins

According to UC Davis (1997), there are three naturally occurring species which are found in species that do not involve marine algae:

- **Gempylotoxin**: this is found in the escolars or pelagic mackerel, a small group of fish-eating oceanic fish as well as the snoek *Thyrsites atun*. In Australia the mackerel species of concern include *Lepidocybium flavobrunneum* and *Ruvettus pretiosus*. They produce an oil which has a purgative effect. No problems have been reported for this biotoxin group in New Zealand.
- **Tetramine**: this toxin is found in the salivary glands of the welk *Neptunia*, which is not found in Australia or New Zealand.
- **Tetrodotoxins**: there are 80 species of puffer (also called fugu or blowfish) fish that are know to contain the neurotoxin tetrodotoxin, some occur in Australian waters. It is unclear whether the fish itself produces the toxin, or like ciguatoxin, it is introduced to the fish by eating toxic algae.

Further information

For a detailed discussion of the biotoxin situation in New Zealand refer to MBMB (1996) whilst ASSAC (1997) provides a some brief notes for Australia. Details on symptoms and treatment, detection and prevention and selected bibliography for each of these toxins refer to UC Davis (1997).

9.4.3.4 Off-flavour compounds

Off-flavour compounds, otherwise known as tainting substances, can seriously affect the palatability of fish, crustaceans and molluscs and therefore have a large deleterious impact on the aquaculture and wild-capture fishing industries (both commercial and recreational). According to Zweig et al. (1999) odorous organic compounds, such as those from petroleum distillates and paper processing and other industrial effluents, are a common source of off-flavours in fish.

Table 4.4.5 in Volume 1 identifies a variety of off-flavour compounds together with the threshold concentration at which tainting will occur.

Svobodova (1993) suggested the admissible concentrations for a range of off-flavour causing contaminants:

- oils range between 2 and 25 µg/L

- chlorophenol is 1µg/L
- cresol is 3 µg/L
- resorcine is 4 µg/L
- hydroquine is 1 µg/L

According to Zweig et al. (1999), the simplest test for off-flavour producing organics requires neither equipment nor reagents: water which tastes or smells unusual may result in off-flavour. Therefore a sensory assessment can often be preferable to chemical analysis in assessment of the source water.

In addition to the chemical contaminants, a number of freshwater blue-green microalgae (cyanophycaea) and bacteria (actinomycetes) can cause off-flavours in native fish. The most common is the earthy/musty flavour, often referred to as 'muddy' taste, which often occurs in silver perch (*Bidyanus bidyanus*). Rowland (1995b) reported that the majority of off-flavour episodes are caused by geosmin and 2-methylisoborneaol compounds which are rapidly absorbed by fish and stored predominantly in fat tissue. Decaying organic matter can also cause off-flavour. The incidence of these off-flavours is highest in warmer months, during blooms of blue-green algae and in ponds with high stocking and feeding rates. Most off-flavours can be readily purged by placing fish in clean water such as underground or spring water, domestic (dechlorinated) or rainwater. Rowland (1995b) recommended that fish be purged in a solution of 3 g/L NaCl for at least 7 days.

9.4.3.5 Preventative and management approaches

There are usually high costs associated with detecting the levels of chemical and/or biological contaminants, either in the flesh of the aquatic organisms or in the waters in which they occur. It is generally accepted that food species should not be grown in, or harvested from, waters likely to be exposed to contamination. If a contamination should occur, the aquatic organisms should be regularly analysed to ensure that the ANZFA standards are not exceeded in harvested product.

Excluding filter-feeding shellfish where testing generally takes place prior to harvest (see below and Section 9.4.3.5 No.2), a problem with other types of aquaculture/fishery product testing is that it is retrospective. For planning purposes a method of product quality prediction would be preferable. This problem may be illustrated by the following examples:

- The viability of the setup of an aquaculture business is being investigated. How can the investors predict whether, on harvesting, the product will be suitable for sale for human consumption?
- It is proposed to start up an industrial/sewage plant upstream of a commercial fishery. How can we predict whether effluent from the plant will have a significant adverse effect on the fishery product quality?

Section 9.4.3.5 No.1 gives a simplified approach to making predictions of this nature using the *bioconcentration factor* approach. Since circumstances will vary enormously from case to case, this approach is only intended as a general guide, not as a set of prescriptive rules. In addition, because of the complexities involved, uncertainties will be associated with any prediction. Predictions cannot replace product testing. However, they may enable problems to be identified and resolved before they impact on an industry.

The testing of flesh samples (particularly for filter feeding species) to monitor growing area conditions provides a better indicator of long-term growing area water quality than an instantaneous grab water sample — water samples are useful for tracking pollution or for monitoring trends over a long period of time, however, the value of this type of sampling is of course related to the number of samples collected (i.e. spatial and temporal). This *area classification* approach is discussed in Section 9.4.3.5/2. The use of routine *monitoring for phytoplankton* is outlined in Section 9.4.3.5/3. Another preventative or management option is the *three-phased screening* approach, suggested by Zweig et al. (1999), is provided in Section 9.4.3.5/4.

1. Bioconcentration factor approach

One of the simplest methods of predicting bioaccumulation is the *bioconcentration factor* approach. Numerous terms are used in the literature for the same or related concepts, including *concentration factors*, *bioaccumulation factors* and *concentration ratios.*

Basic principles

Organisms obtain chemicals from a variety of sources in their environment, such as water, food, sediment, etc. The uptake of many chemicals is not homeostatically controlled by the organism's metabolism. If this is the case, then a higher concentration of the chemical in the source should result in a higher concentration in the organism. In fact, the bioconcentration factor approach assumes that the concentration in the organism which is attributable to a given source is *proportional* to the concentration in the source. In this case, the constant of proportionality is called the bioconcentration factor. For example, if C^w_f is the concentration of a chemical in a fish's flesh due to uptake of that chemical from the water in which it lives, and if C_w is the concentration of the chemical in the water, then the bioconcentration factor F^w may be calculated as follows:

$$F^w = \frac{C^w_f}{C_w}$$

If C^w_f and C_w are in the same units (for example, mg/kg), then F^w will be a simple number without units.

In some cases, it may be possible to make the further simplification that the concentration in an aquatic organism is directly related to only one source: the water in which it lives. This may be the case if the organism is known to take up the chemical almost exclusively from the water, or if it can be assumed that any changes in concentrations in the water will also result in proportional changes in concentrations in other sources such as food, sediment, etc.

Example 1

A fish used in aquaculture is known to bioaccumulate a chemical from two sources: water and food. The bioconcentration factors have been determined to be 10 and 30, respectively. If the concentration in the water[2] (C_w) is 0.01 mg/kg, and the concentration in the feedstock (C_F) is 0.005 mg/kg, then we can predict the concentration in the fish to be:

$$C_f = C^w_f + C^F_f = \left(F^w \times C_w\right) + \left(F^F \times C_F\right) = \left(10 \times 0.01\right) + \left(30 \times 0.005\right) = 0.25 \text{ mg/kg}$$

2 For water, concentration units of mg/kg and mg/L are equivalent.

Example 2

A fishery has been harvesting from a river floodplain area, and has collected water and product quality data over a long period. The harvested fish are restricted to the floodplain area over their lifetime. A factory is proposed to be built upstream, resulting in discharges of lead to the river. It is predicted that other water quality parameters will not be significantly affected by the factory, and that the *increase* in lead concentrations in floodplain waters resulting from the factory will be 0.01 mg/L (total water).

Based on their historical data, the fishery determines that the average bioconcentration factor F^w for lead in harvested product from the floodplain is 200 relative to total water, and that the average concentration of lead in the water (without the factory being present) is 0.003 mg/L. Using the bioconcentration factor, we can predict that the average concentration in product (C_p) after the factory is operating will be:

$$C_p = F^w \times C_w = 200 \times (0.003 + 0.01) = 2.6 \text{ mg/kg}$$

Since this is higher than the limits for lead in fish (1.5 mg/kg, table 9.4.46), the restrictions on effluent release from the factory may need to be significantly tighter than those proposed.

Some difficulties with using the bioconcentration factor approach

The bioconcentration factor approach attempts to model complicated bioaccumulation processes using a simple ratio. Caution must be exercised when using the approach, particularly when some of the basic assumptions of the method may not apply to the specific case being investigated. Some of the potential difficulties and limitations of the approach will be discussed in the following.

Assumption that the chemical is not homeostatically controlled

The bioconcentration factor approach should not be used for chemicals which are homeostatically controlled by the organism. In particular, it should not be used for essential elements (e.g. Co, Cu, Fe, Mn, Mo). For these elements, an organism's metabolism can be expected to maintain a constant flesh concentration regardless of the concentration in the water, up to the point at which an overload occurs (Chapman et al. 1996).

Identification of the appropriate source for calculation of bioconcentration factor

The bioconcentration factor used should relate the concentration in the aquatic organism to that source (e.g. total or filtered water concentration) which is the best indicator of uptake of the constituent in question.

Tissue distribution and the use of a bioconcentration factor for the appropriate product

Because most contaminants will bioaccumulate to a different extent in different tissues, the bioconcentration factor must relate to the tissue which is to be sold for consumption (muscle flesh, whole fish, etc).

Effect of water quality on bioconcentration factor

General water quality parameters, such as temperature, pH and major ion and suspended solids concentrations, can affect the bioconcentration factor. For example, in one study of polychlorinated biphenyl (PCB) in sunfish, the bioconcentration factor increased from 6000 to 50 000 between 5 and 15°C (Barron 1990).

Increasing concentrations of major ions with similar chemistries to that of the trace contaminant may lead to lower bioconcentration factors. For example, for freshwater fish

higher potassium concentrations can result in lower bioconcentration factors for cesium, while higher calcium concentrations can result in lower bioconcentration factors for strontium (NCRP 1984).

Assumption of steady state between tissue and water concentrations

The bioconcentration factor approach assumes that a steady state condition has been reached between concentrations in the edible tissue of the organism and concentrations in the source. Once this steady state is attained, the rate of uptake of a contaminant by the tissue equals the rate of loss (i.e. excretion) by that tissue. In reality, this only applies when the rates of uptake and loss are fast, so that a steady state condition is reached in a short time relative to both the lifetime of the aquatic organism and to the time scales over which changes in concentrations in the source occur.

On the other hand, when the uptake and loss rates are slow, then the aquatic organism will accumulate the contaminant gradually over its lifetime. In such a case, the bioconcentration factor approach may still prove useful *provided* that the bioconcentration factor has been determined using concentrations in aquatic organisms which were of the same age as those which are to be harvested, and that the water concentrations used are average concentrations determined over the lifetime of the aquatic organism.

Captive/non-mobile vs. mobile populations

Bioaccumulation predictions are likely to be most reliable in situations where the organism is captive or non-mobile (e.g. as for aquaculture and for sedentary species such as mussels), because the water quality to which they are exposed may be more accurately and reliably determined than for more mobile populations.

Obtaining concentration factors

Given the above complications, locally-derived bioconcentration factors are to be preferred where they are available. Unfortunately, collecting the necessary data can be a time-consuming and expensive exercise.

In the case of organic chemicals, measurement of the chemical partitioning between water and the organic chemical octanol is commonly used to estimate bioconcentration factors. Although there are some complications with this approach (Barron 1990, Meylan et al. 1999), it is less expensive that the measurement of the bioconcentration factor itself.

Where a locally-derived factor is not available, relevant literature values will need to be obtained. Some databases of bioconcentration factors exist, such as the USEPA's AQUIRE database (USEPA 1995). Generic guideline factors for a number of elements are also available from IAEA (1994).

Uncertainties in bioconcentration estimates

The accuracy of any prediction using bioconcentration factors will depend upon a large number of parameters. In general, accuracy better than an order of magnitude (i.e. a factor of ten) should not be expected, and the situation may be considerably worse than this where, for example, only generic guideline bioconcentration factors are available.

Further information

Barron (1990) discusses factors affecting bioconcentration of organic chemicals.

Walker and Gobas (1999) discuss the use of bioconcentration factors to derive water quality guidelines, particularly for organic chemicals.

Chapman et al. (1996) discuss bioaccumulation estimates for essential metals.

IAEA (1994), ICRP (1978) and NCRP (1984) give general information on methods of bioaccumulation estimation, with an emphasis on bioaccumulation of radioactive elements.

2. Area classification approach

The Australian and New Zealand Area Classification Approaches (described below) are heavily based on the USFDA program (also described below).

Australia

The Australian Shellfish Quality Assurance Program (ASQAP), formerly called the Australian Shellfish Sanitation Control Program (ASSCP), was introduced in 1988 in response to needs of the emerging Tasmanian oyster industry and AQIS (Australian Quarantine Inspection Service). In addition it was recognised that many Australian shellfish growing areas were under increasing pressure from a range of human activities including discharge of untreated or poorly treated human wastes, direct industrial waste discharge and runoff from urban and agricultural areas. to comply with export requirements.

The objectives of the ASQAP include:

- control the harvesting of contaminated shellfish by identifying and evaluating the impact of pollution of shellfish growing waters;
- protect shellfish from contamination after harvesting (post-harvesting controls).

A major component of the ASQAP is the identification of safe shellfish growing areas to permit commercial harvesting for the domestic market and/or for export. It should be noted that this program is not compulsory in any way and the degree to which the program is implemented varies amongst the states. While most states do not differentiate between domestic and export product, some have no legislative force behind their domestic sales. In addition, there is difficulty in applying the program to non-farmed shellfish in some states. For these reasons, the ASQAP is currently under review.

The ASQAP Operations Manual (ASSAC 1997) provides authorities interested in shellfish sanitation with a risk based system of procedures and guidelines to be used when regulating shellfish growing areas, harvesting, processing and distribution of shellfish. It covers:

- classification and survey of growing areas;
- relaying (relocation) and harvesting controls;
- post-harvest handling, storage, processing and transportation.

The shellfish harvesting area classification systems used in the ASQAP rely on the Sanitary Survey approach to ensure that molluscan shellfish harvested for human consumption are safe. The Sanitary Survey consists of:

- the identification and evaluation of all potential and actual pollution sources (Shoreline Survey) — this describes the studies required to identify and quantify pollution sources and estimate the movement, dilution and dispersion of pollutants in the receiving environment;
- the monitoring of growing waters and shellfish to determine the most suitable classification for the shellfish harvesting area (Bacteriological Survey) — this refers to the measurement of faecal indicator levels in the growing areas.

Resurveys are conducted regularly to determine if sanitary conditions have undergone significant change. They provide the basis for the classification of coastal and estuarine areas for the harvesting of clams, oysters, scallops, mussels and other bivalve molluscs.

As *Escherichia coli* (*E. coli)* is present in faeces and is not a normal constituent of the environment, the presence of *E. coli* (nonpathogenic strains) is used as an indicator of faecal contamination of food and water. The presence of *E. coli* in food or water suggests that enteric pathogenic bacteria may be also present. According to Jackson and Ogburn (1998) bacteria commonly used as indicators of faecal contamination include:

1. *Escherichia coli*.
2. Faecal coliforms. A less restrictive test that is quicker to perform and includes intestinal bacteria including *E. coli*.
3. Total coliforms. The least restrictive test that demonstrates the presence of bacteria from the intestine, as well as some related species of bacteria normally found in the environment.

It is pertinent to note that these microorganisms, including *E. coli***,** are derived not only from human sources of faecal pollution, but also from wild and domestic animals, including birds (Kator & Rhodes 1991). Enteric viruses are also a major problem, and are not detected by the use of normal bacterial indicators. In addition, some marine bacteria (e.g. *Vibrio* sp.) can also cause illness in consumers. Thus, it is important that techniques to monitor these types of organisms are developed and implemented. Some states are currently investigating other indicators for enteric viruses (e.g. coliphage) (K Lee pers comm 2000).

The ASQAP categories of classification are based on levels of contamination from sewage, poisonous or deleterious substances, other pathogenic organisms of non-faecal origin and biotoxin-producing organisms, radionuclides, and toxic wastes. The criteria for each classification are contained in the ASQAP Operations Manual (ASSAC 1997). The classifications that can result from the analysis of sanitary surveys are as follows:

- **Approved**: Shellfish harvesting areas which as a result of a sanitary survey and marine biotoxin monitoring have been found not to contain faecal material, pathogenic organisms or toxic or deleterious substances in levels that may affect public health should be classified as approved. Shellfish harvested from harvesting areas classified as approved can be sold directly for human consumption (direct marketing).
- **Conditionally approved**: This classification has the same sanitary quality as the Approved classification for most of the time. However these area may be subject to intermittent pollution from events which may be a potential threat to public health (e.g. failure of waste water treatment plant, seasonal increase in the human population, high rainfall causing run-off of pollutants and seasonal anchorage of a fishing fleet). These intermittent pollution sources must be predictable to allow appropriate management plans to be developed. The development and monitoring of these management plans require substantial resources. Conditional approved areas are closed to harvesting when the coliform concentrations exceed the approved area classification standards.
- **Conditionally restricted**: This classification is the same as for the conditional approved classification, except that the area is closed when the coliform concentrations exceed the restricted area classification criteria but are open to harvesting for relaying or depuration when the coliform concentrations meet the restricted area classification bacteriological standard. As for conditionally approved area.

- **Restricted area**: A restricted area classification might be considered where the harvesting area does not meet the approved area classification criteria but is not grossly polluted. Shellfish may be harvested if subjected to a suitable, effective purification process before being sold for consumption (e.g. depuration or relaying). A common situation where this classification might be appropriate is for harvesting areas affected by non-point source pollution from either urban or rural sources which cause the water quality to fluctuate unpredictably or of sufficient frequency that a conditional approved area classification is not feasible.
- **Prohibited area**: These are areas that are not properly surveyed, and hence of undetermined quality, or which contain excessive contaminants (i.e. human sewage, industrial and agricultural chemicals) or toxic substances (i.e. toxic algal species). Harvesting of shellfish from these areas is prohibited.

Following classification, routine monitoring is implemented.

New Zealand

New Zealand is an active member of the Australian Shellfish Quality Assurance Program.

New Zealand operates a mandatory shellfish quality assurance program for all commercial bivalve shellfish areas. The New Zealand Shellfish Quality Assurance Program (NZSQAP) is overseen by the Ministry of Agriculture (MAF) Food Assurance Authority, but involves a partnership with the Ministry of Health. This program is based on the United States Food and Drug Administration program but has been further developed to manage conditions that are unique to the New Zealand environment and aquaculture industry.

The program requires that a full sanitary survey of each growing area catchment be undertaken on public health grounds to assess the risks of the growing waters being contaminated. Areas highly susceptible to microbiological (including viruses) or chemical contamination would not be approved for harvest. Shellfish growing waters can be classified as:

- **Approved** areas when, under the most unfavourable meteorological, hydrographic, seasonal or point-source conditions, the total coliform median or geometric mean MPN of the water does not exceed 70 per 100 mL and fewer than 10% of the samples exceed a five-tube MPN of 230 per 100 mL (or a three-tube MPN of 330 per 100 mL). In addition, faecal coliforms do not exceed 14 per 100 mL and fewer than 10% of samples exceed a five-tube MPN of 43 per 100 mL (or a three-tube MPN of 49 per 100 mL). At least 15 samples must be analysed. Failure to meet the standards results in temporary closure of the waters.
- **Remote approved** areas have no human habitation in the growing area catchment and not impacted by any actual or potential pollution sources. The area shall meet the approved area requirements specified above, except that the number of samples for adverse pollution condition sampling may be varied at the discretion of the authorised health officer.
- **Conditionally approved** areas when the waters are subject to bacterial contamination events, such as from heavy rainfall in the catchment or discharge of sewage. If such an event occurs the State Shellfish Control Agency (SSCA) will conduct a sanitary survey and either approve harvesting if sanitary standards (as above for Approved Waters) are maintained, or close the area until further surveys demonstrate that the sanitary standards have been attained again.
- **Restricted** areas when the waters are subject to limited amounts of pollution such that shellfish must be depurated or relayed prior to sale. Under the most unfavourable

meteorological, hydrographic, seasonal or point-source conditions, water samples should not have total coliform levels in excess of 700 per 100 mL with fewer than 10% of samples exceeding 2300 per 100 mL for a five-tube MPN. In addition, faecal coliforms must not exceed 88 per 100 mL, with fewer than 10% of samples exceeding 260 per 100 mL for a five-tube MPN, or 300 per 100 mL for a three-tube MPN.

- **Conditionally restricted** areas when the waters are subject to intermittent pollution which makes them temporarily unsuitable as a source of shellfish for depuration or relaying. The waters are closed for harvesting until they can meet the sanitary criteria for restricted waters.
- **Prohibited** areas when the level of pollution is such that shellfish are likely to be unfit for human consumption even after depuration or relaying. The harvesting of shellfish is banned from such waters.
- **Unclassified** areas when no sanitary survey has been conducted. Harvesting of shellfish from such areas is banned.

After the sanitary survey has been completed a routine water/flesh sampling program is implemented to monitor for microbiological, chemical and marine biotoxin contamination. If the water quality does not meet the minimum standards for microbiological, marine biotoxin or potential chemical parameters, harvesting from those areas effected is prohibited until monitoring shows that the standards are being met again.

Sampling, testing and monitoring of shellfish growing waters is at the expense of individual industries and is regulated by quality control centres which arrange regular testing and inspection of shellfish growing sites. The mandatory marine biotoxin program includes both phytoplankton and shellfish monitoring. The program is approved by the Marine Biotoxin Technical Committee and may be found in the National Marine Biotoxin Management Plan.

Further details on the program may be found in Industry Agreed Implementation Standard 005.1: Shellfish Quality Assurance Circular. A copy of this standard may be found on the following web address — www.maf.govt.nz/Standards/seafood/iais/5/005.pdf.

For further explanation contact Phil Busby, National Manager Seafood, MAF Food Assurance Authority, PO Box 2526, Wellington, New Zealand, tel: +644 474-4167, fax: +644 474-4239.

European Union

European shellfish growing area classification is based on faecal coliform levels in shellfish meat. Annual classifications of growing areas are performed by regulatory agencies in each country. The European Council Directive (1992) sets the standards for each of the four growing area classifications:

- *Class A* areas are approved for harvesting shellfish that can be sold directly to the public, with no purification required. Shellfish harvested from Class A areas must contain <300 faecal coliforms or <230 *E. coli* per 100 g of mollusc flesh and intravalvular fluid based on a five-tube three-dilution MPN test or other acceptable method. *Salmonella* must also be absent from 25 g of mollusc flesh. In addition, there must be no positive results for Diarrhetic Shellfish Toxin and the amount of Paralytic Shellfish Toxin must be <80 micrograms per 100 g of mollusc flesh. Radionuclide levels are also specified.
- *Class B* areas are approved for harvesting, but all shellfish must be purified (by relaying or depuration) or cooked by an approved method prior to sale to the public. Shellfish in

Class B areas must have <6000 faecal coliforms or <4600 *E. coli* per 100 g of mollusc flesh in 90% of samples.

- *Class C* areas are not approved for immediate harvesting. Instead shellfish from these areas must be relayed for a prolonged period (at least two months). This process may also be combined with purification to ensure shellfish meet microbiological end-product standards. Alternatively, shellfish may be harvested and cooked by an approved method prior to sale for human consumption. Shellfish from Class C areas must have <60 000 faecal coliforms per 100 g of mollusc flesh.
- *Class D* areas are those from which harvesting of shellfish is totally prohibited. Shellfish in these areas have >60 000 faecal coliforms per 100 g of mollusc flesh. In addition, areas may be designated as prohibited at the discretion of the state.

Any of the above classified areas may be subject to closure if routine monitoring indicates that sanitary standards are being exceeded. In addition, the EC Directive specifies criteria that must be met for all aspects of shellfish processing (e.g. the treatment of shellfish during harvesting, transport and storage). The level of continued monitoring required to maintain the growing area classifications, varies between countries.

USA and Canada

In the USA, the National Shellfish Sanitation Program (NSSP) of the Food and Drug Administration classifies waterways for shellfish harvesting on the basis of a sanitary survey of the growing area, in addition to an ongoing strategic water sampling program. A protocol for depuration has also been established (NSSP 1990 a,b, NSSP 1995 a,b). A similar classification system operates in Canada. The NSSP emphasises the importance of the sanitary survey in determining acceptable and unacceptable growing areas and requires that the survey of the waterway be updated annually. The NSSP also establishes contingency plans for marine biotoxins and other deleterious substances (e.g. pesticides and heavy metals). Shellfish growing waters are then annually classified as:

- **Approved** areas when, under the most unfavourable meteorological, hydrographic, seasonal or point-source conditions, the total coliform median or geometric mean MPN of the water does not exceed 70 per 100 mL and fewer than 10% of the samples exceed a five-tube MPN of 230 per 100 mL (or a three-tube MPN of 330 per 100 mL). In addition, faecal coliforms do not exceed 14 per 100 mL and fewer than 10% of samples exceed a five-tube MPN of 43 per 100 mL (or a three-tube MPN of 49 per 100 mL). At least 15 samples must be analysed. Failure to meet the standards results in temporary closure of the waters.
- **Conditionally approved** areas when the waters are subject to bacterial contamination events, such as from heavy rainfall in the catchment or discharge of sewage. If such an event occurs the State Shellfish Control Agency (SSCA) will conduct a sanitary survey and either approve harvesting if sanitary standards (as above for Approved Waters) are maintained, or close the area until further surveys demonstrate that the sanitary standards have been attained again.
- **Restricted** areas when the waters are subject to limited amounts of pollution such that shellfish must be depurated or relayed prior to sale. Under the most unfavourable meteorological, hydrographic, seasonal or point-source conditions, water samples should not have total coliform levels in excess of 700 per 100 mL with fewer than 10% of samples exceeding 2300 per 100 mL for a five-tube MPN. In addition, faecal coliforms must not exceed 88 per 100 mL, with fewer than 10% of samples exceeding 260 per 100 mL for a five-tube MPN, or 300 per 100 mL for a three-tube MPN.

- **Conditionally restricted** areas when the waters are subject to intermittent pollution which makes them temporarily unsuitable as a source of shellfish for depuration or relaying. The waters are closed for harvesting until they can meet the sanitary criteria for restricted waters.
- **Prohibited** areas when the level of pollution is such that shellfish are likely to be unfit for human consumption even after depuration or relaying. The harvesting of shellfish is banned from such waters.
- **Unclassified** areas when no sanitary survey has been conducted. Harvesting of shellfish from such areas is banned.

Shellfish harvested from approved or conditionally approved waterways that meet approved area criteria may be harvested and sold directly. Depuration or relay is required for shellfish harvested from conditionally approved areas not meeting approved criteria, and for shellfish harvested from restricted areas or from conditionally restricted areas that meet restricted area classification.

The practice of depuration in the USA is strictly controlled by the SSCA. A scheduled depuration process (SDA) is established for each depuration facility (NSSP 1995b). This process evaluates the effectiveness of the plant to reduce the number of microorganisms in shellfish harvested from restricted waters on the basis of experimental data. In addition the SDA assesses plant design and construction and process variables such as environmental parameters. This process of verification results in the determination of a maximum initial level of faecal coliforms for each plant. Each batch of shellfish to be depurated must be sampled from the harvest lot and also after the depuration process. All samples are analysed for the presence of faecal coliforms by the MPN method. Rigid sampling regimes specify the number of samples which are required from each batch and the number of samples is dependent on the number of areas harvested and the variability of pollution in each area. End-product standards have been established for each shellfish species commercially harvested. Shellfish are depurated for at least 48 hours.

3. Phytoplankton monitoring

Phytoplankton monitoring may prove useful as a predictor of marine biotoxins appearing in shellfish. However, phytoplankton monitoring is undertaken only in certain parts of Tasmania, South Australia and Western Australia as part of a routine marine biotoxin monitoring program. This shortcoming is being examined as part of the National Biotoxin Strategy Project, a FRDC funded project. Cost-effective tests to monitor for the presence of biotoxins are urgently needed, although monitoring is probably hindered by high costs and the current principal of cost recovery from industry (K Lee pers comm 2000).

New Zealand uses phytoplankton within the comprehensive marine biotoxin management program that is mandatory for all commercial harvest areas. A similar program is operated by the Ministry of Health for all recreational shellfish harvesting sites. A combination of phytoplankton and flesh tests are used to monitor for biotoxin activity. Commercial areas are sampled weekly for biotoxin activity and should mandated trigger levels be reached for a number of species, flesh testing is invoked immediately. The trigger levels are those listed in table 9.4.47.

Further specific details on the mandatory monitoring requirement and regulatory test methods may be obtained by contacting Phil Busby, National Manager Seafood, MAF Food Assurance Authority, PO Box 2526, Wellington, New Zealand, tel: +644 474-4167, fax: +644 474-4239.

Table 9.4.47 Levels of risk assessment with regard to phytoplankton cell numbers (MBMB 1996)

Species	Toxin *	Risk	Cell/litre
Alexandrium spp.	PSP	Low Moderate High Very high	1–200 201–1000 1001–5000 >5001
Dinophysis acuminata	DSP	Low Moderate High Very high	1–1000 1001–2000 2001–10 000 >10 001
Dinophysis acuta	DSP	Low Moderate High Very high	1–500 501–1000 1001–5000 >5001
Gymnodinium sp.	NSP	Low Moderate High Very high	1–1000 1001–2000 2001–10 000 >10 001
Prorocentrum lima	DSP	Low Moderate High Very high	1–500 501–1000 1001–5000 >5001
Pseudonitzchia sp.	ASP	Low Moderate High Very high	1–50 000 50 001–200 000 200 001–500 000 >500 001
Rhizosolenia sp.	*can cause bad taste & shellfish deaths*	Low Moderate High Very high	1–50 000 50 001–200 000 200 001–500 000 >500 001

* ASP = Amnestic shellfish poisoning , DSP = Diarrhetic shellfish poisoning, PSP = Paralytic shellfish poisoning

In New Zealand, the shellfish industry have requested that they be notified where levels of certain phytoplankton species (table 9.4.48) are exceeded so that harvesting decision can be made.

Table 9.4.48 Notification levels for phytoplankton cell numbers (MBMB 1996)

Species	Cell/litre
Alexandrium sp.	200
Dinophysis spp.	500
Gymnodinium cf. *breve*	1000
Prorocentrum lima	500
Pseudonitzchia sp.	100 000 (below 50% of total phytoplankton)
Pseudonitzchia spp.	50 000 (below 50% of total phytoplankton)

4. Three-phased screening approach

This is recommended by Zweig et al. (1999) and utilises expert water quality analysis laboratories to do the assay for the water quality. It is designed for aquaculture operations to evaluate source water quality in a step-by-step process to minimise costs to the degree possible.

For Phase I the water quality criteria of the source water for the basic physio-chemical properties necessary to sustain the cultured organisms are measured. This provides a simple means of screening the source water without going through the more expensive tests for the

chemical or natural contaminants. Zweig et al. (1999) suggest that if chemical or natural contaminants are not suspected, and Phase I criteria are met, then the source water can be considered acceptable. If the Phase I criteria are not met, there are three options:

- water source is rejected (look for another site);
- undertake a Phase III field trial; or
- assess the technical and economic feasibility of treating the source water to bring it within acceptable Phase I criteria.

Phase II is designed to screen criteria on anthropogenic (of human origin) pollutants and biological contaminants. Because it is neither feasible nor desirable to test for every possible pollutant, only pollutants typical of current and historical industrial, municipal and agricultural activities in the catchment should be tested. If the Phase II criteria are not met, the feasibility of pre-treating the source water could be considered as in Phase I. A decision as to whether to pursue a Phase III field trial or reject the source water can then be made.

If both Phase I and Phase II criteria are met, it is not mandatory to pursue Phase III. However, Zweig et al. (1999) advise that Phase III be pursued, if possible, as a means of minimising the risk of project failure.

Phase III involves a pilot study or field test in which the culture species are grown in the selected source water, using similar management techniques as a those of the proposed project. They would then be tested for bioaccumulated pollutants and off-flavour. The pilot study could also be replaced by sampling cultured animals from an existing aquaculture facility, if available, which is using the same source water and the planned technology.

Following Phase III where implemented, a final decision can be made on the use of the source water.

9.4.4 Some precautionary comments

These guidelines have been developed on the basis of information currently available (to the middle of 1996; see comments in Section 9.4.5 and 4.4.7 regarding future work) for Australian and New Zealand aquaculture species. The approach (detailed in Section 9.4.1.4) was to concentrate the information search on one or two representative species from each of eight species groups, or categories. While the focus of the search allowed data on the commonly cultured species to be collated, it soon became apparent that despite increases in aquatic toxicology research in Australia and New Zealand in the past ten years, no information is available for some important aquaculture species. This is particularly the case for non finfish species. Section 9.4.5 details many of the deficiencies and makes suggestions for future research needs.

A review of the data presented in Section 9.4.2 shows that the toxicological data are occasionally contradictory. When carefully viewing the data it becomes apparent that the toxicity values sometimes differ by several orders of magnitude, i.e. one source may recommend a value of 1 mg/L for a toxicant, whereas another lists 0.01 mg/L for the same toxicant. This is highly confusing to the reader and certainly makes the guidelines appear less credible. Differences in data for given species were most likely due to different methods of exposure — i.e. time and duration of exposure, size and age of fish, and test conditions: temperature, pH etc.

However, the stringency of the various researchers' methods is unknown, thus when using the guidelines the following should be considered:

- More recent data may be more reliable than data previously published due to improved technology and methods.
- In cases where differences in acceptable/tolerated concentrations are extreme between different researchers, it is suggested to use the general guideline and proceed with caution, i.e. monitor fish for signs of avoidance behaviour, stress, etc.
- In the case of organic toxicants, these compounds are extremely persistent in the environment and thus have the potential to accumulate in the sediments. This is particularly important for bottom feeders, so sediments should also be monitored for levels of these compounds. Additionally, other adverse effects such as stress and immunosuppression can occur at levels much lower than those causing clinical toxicity.

Additionally, much of the data are traced from other databases or compilations, i.e. from secondary sources. Thus, much of the original literature is not based on recent research, but dates back more than ten or twenty years. Analytical methods applied at that time may not have been as sophisticated as they are today, and this may be one reason that the guidelines sometimes differ by several orders of magnitude: for example, the lowest level detectable for a toxicant 'x' twenty years ago may have been 1 mg/L. Toxic effects may have been observed at any level above this detection limit. Where toxicity was occurring below the detection limit, either it would not have been picked up or it was attributed to background. Necessarily, the *safe* level would therefore have been determined as <1 mg/L. Today, refined analysis may be able to show that, in fact, even at 0.1 mg/L a toxical effect can occur, and thus the *safe* level has to be refined.

The definition of toxicity itself is not straightforward, nor is the set-up of toxicity tests consistent, so there will always be dispute about the applicability of published values and derived guidelines. A number of other drawbacks to this approach can be identified:

1 In aquaculture, the culture species are in an artificial farm environment where: avoidance of pollutants is impossible due to physical constraints of the culture structures; the feed is usually not derived from the immediate environment (except in the case of bivalves and some freshwater fish and crayfish) so oral exposure to pollutants is unrelated to ambient concentrations; and there are additional stresses due to farming procedures (e.g. any procedure requiring handling, higher stocking densities).

2 Aquatic toxicological studies often involve the use of surrogate species of no commercial value. While this information cannot be extrapolated easily to aquaculture species under farming conditions, it can be a good starting point for future considerations.

3 Tolerance to individual pollutants is very variable between species, even within the species groups selected in table 9.4.1. For example, Davies et al. (1994) noted that safe levels of contamination for several types of pesticides calculated from *Oncorhynchus mykiss* toxicity data were not suitable for the protection of juvenile and adult *Galaxias maculatus* and *Pseudaphritis urvilli* from physiological stress in at least three of seven cases of exposure.

4 Many chemicals break down into a number of isomers and metabolites. Some forms of a chemical compound or an element are more biologically available and, thus, often more toxic. For example, metal speciation (Section 3.4.3) is very important but rarely reported,

with free ions usually being more toxic. For this reason a multidisciplinary research approach, combining toxicology and chemistry is required.

5 The physiological effect of a substance (e.g. on reproductive potential and growth) may be just as important as the direct toxicity of the substance. Unfortunately, most studies only include data on direct toxicity.

6 In aquaculture it is desirable to maintain the concentration of a potential toxicant below the level that is known to have any adverse effect on the culture species. Yet, the toxicity of most substances is proportional to the time of exposure. The toxicity tests are usually undertaken over a short period (24 hr to 96 hr) and undesirable sublethal effects of a substance may not be revealed. For aquaculture, chronic long-term effects are important. For example, fish may tolerate 3 mg/L DO_2 for several days without apparent harm, but over a longer period the fish growth could be lowered and the fish could become more susceptible to diseases. Not only would mortality affect farm production but also reduced food conversion ratio (FCR), reduced growth, and increased sensitivity to pathogens could occur. Unfortunately, chronic long-term data are not available for most toxicants and aquaculture species.

7 Toxicity data are reported as nominal or measured values; nominal values are less accurate than measured ones and can result in an underestimation of toxicity due to unknown loss of toxicant during the test.

8 Toxicity data are based on static, semi-static or flow-through tests: flow-through tests tend to give the lower toxicity values since the toxicant is always present in the experimental system at a constant level. Toxicant levels in static or semi-static tests may decrease during the whole test (static tests) or until renewal (in semi-static tests). Unless exposure to the toxicant in the environment is pulse rather than constant, it would be more appropriate to use flow-through test results.

9 Toxicity of chemicals is related to the physiological state of fish or other aquatic organisms affected by water quality, most importantly temperature, but also other variables such as water hardness and pH. Fish metabolic rate may double for every 10°C rise in temperature, resulting in increased uptake of toxicants. Additionally, temperature or low oxygen related stress could increase susceptibility to toxic effects.

10 Due to logistics, toxicity tests often use small size animals or early life stages. Toxicity, however, may change with life stage (often larvae are most sensitive) and size (usually smaller fish are affected earlier). This is because the weight-specific metabolic rate of fish decreases in larger fish and, thus, they may take up toxicants more slowly than smaller fish.

11 The source of test animals (wild population or cultured specimens) and the environment in which the animals lived previously may affect results of toxicity tests and, as a consequence, lead to overestimation or underestimation of toxicity of the tested compound.

12 Aquaculture animals may sometimes be exposed to a mixture of toxicants or other water quality parameters at suboptimal levels. For example in Macquarie Harbour, Tasmania, fish may be exposed to increased copper levels in association with low pH and low salinity. The results of standard toxicity tests provide little information about potential synergistic and antagonistic effects of combinations of pollutants.

13 Although no observed effects concentrations (NOECs) have been used to determine many of the water quality guidelines in this document, there is some opposition to their use in the scientific community because they are not a valid statistical endpoint.

14 For much of the data presented in the tables, no information is given on the type of endpoint measured in the test. If the result of the test is expressed as LC_{50}, the result will be much greater than NOEC values.

9.4.5 Priorities for research and development

Although there has been an increase in research in aquatic toxicology in Australia and New Zealand, information is still lacking on the effects of toxicants and water quality parameters on Australian and New Zealand aquaculture species, particularly invertebrates and marine/brackish water finfish. The majority of information utilised in this report concerned freshwater finfish species. However, not many data are presented for the species that contribute the majority of aquaculture production in Australia and New Zealand. This is of concern because those particular marine and coastal enterprises have to exist in waters that are likely to be impacted by other activities and they would be well-advised to monitor their water quality closely.

The other most obvious deficiency is the absence of guidelines for juvenile life forms. This is unfortunate, as in most cases the juvenile forms will be more susceptible. Thus, while the guidelines provided may be of use to grow-out aquaculture enterprises, they are of little use to hatcheries or closed-cycle operations. However, this potential shortcoming is also evident in other major reviews, including CCME (1993), Svobodova et al. (1993), DWAF (1996), and Zweig et al. (1999).

Deficiencies in the data did not allow a critical or statistical analysis of the data to be made before deriving the water quality guidelines in Sections 9.4.2 and 9.4.3. With the rapidly increasing aquaculture industry in Australia and New Zealand, there is an urgent need to determine water quality guidelines with higher confidence levels. Whilst some new information has become available (particularly Zweig et al. 1999) and has been incorporated into the current report, the search of appropriate literature and databases was originally completed in June, 1996. Thus, the data can be considered incomplete.

The priority for future work (R&D) should be the collation of all available water quality toxicity data relevant to Australian and New Zealand aquaculture species (both adult and juvenile life forms) onto a database so that guidelines can be developed from a critical analysis of the data. Where appropriate, overseas data for the same species can also be incorporated into the database. It would then be possible to identify the species and compounds for which there are significant gaps in toxicity data.

Further research should concentrate on developing data for juvenile forms of important aquatic species, and for aquatic invertebrates, marine finfish and brackish water finfish. In the case of species logistically difficult to test (e.g. southern bluefin tuna) surrogate species should be evaluated or the use of in vitro tests and modelling further investigated.

A specific area of research may be to explore the use of bio-indicator species, be they animals or plants. With funds becoming ever more restricted, it is foreseeable that expensive analytical apparatus will not be widely available, and cheaper methods will be necessary. As well as reduced costs, another advantage of using bio-indicators is that they may be sensitive to a wide range of toxicants, indicating a more general unsuitability of the water for culture. While this approach may sound unacceptable to the analytical scientist, from a farmer's point

of view it seems to be the more sensible approach for on-going monitoring. After all, the farmer is not desperately concerned about the exact ppm level of any one particular agent, but needs to know whether his/her aquatic animals will thrive.

The experiments should be long term, and non-lethal effects such as health, reproduction and growth should be investigated. Issues around bioaccumulation and assimilation also need to be addressed, as do the movement of contaminants or toxicants out of disturbed sediments and into the water column.

Interactions between toxicants and deteriorating or changing water quality should be determined. The effects of mixtures of toxicants should also be investigated. These studies should be undertaken under realistic environmental conditions (i.e. tested concentrations should reflect environmental pollution, in order to realise true farming situations).

Multidisciplinary studies investigating toxicity, chemistry and biology should be undertaken. Long-term monitoring studies on farms should be encouraged to define environmental conditions affecting production. There is need for a computer database combining toxicity data (from both Australia and overseas) with environmental data from culture areas and results of case studies from Australian aquaculture. Responsible agencies/organisations for the establishment and maintenance of this database is yet to be determined.

More research is needed to determine the effects of chemical speciation and different isomers on toxicity of an element or compound. All experiments and monitoring should be properly designed to allow for statistical evaluation of the results.

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