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Water Quality for Irrigation and General Water Uses: Background Information

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Water Quality Guidelines is a joint initiative of the Australian and New Zealand governments, in partnership with the Australian states and territories.

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Contact

Australian Government Department of Climate Change, Energy, the Environment and Water
GPO Box 3090 Canberra ACT 2601
General enquiries: 1800 920 528
Email waterquality@dcceew.gov.au

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1 Introduction

1.1 Background to the review

This Chapter forms an update to an existing Chapter of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ARMCANZ and ANZECC 2000). In particular, this is the updated version of what is known as Chapter 9.2 of the abovementioned guidelines: Water Quality for Irrigation and General Uses. In order to undertake this review, the revised Water Quality Management Framework (WQMF) documentation was reviewed to inform the process. The first two steps of the WQMF involve gaining an understanding of the current knowledge and what it means in terms of the management aims. In this instance, what we know is documented in the current review. In terms of the management aims associated with irrigation water, the goal is to provide good quality water for successful crop production i.e. protect environmental, animal and consumer health by providing water that is safe for a crop/food produce to be irrigated with without adverse effects, and is free of levels of contaminants that will accumulate in plant edible tissues.

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 and ARMCANZ 2000), 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, 1996b), the Canadian Water Quality Guidelines (CCME 2018), the US Guidelines for Water Reuse (USEPA 2012), the EU recommendations on water quality for irrigation (Alcalde-Sanz and Gawlik 2017) and member states regulations, the FAO Water quality for agriculture (FAO 1985, also cited as Ayers and Westcot 1985), ISO Guidelines for treated wastewater use for irrigation projects (ISO, 2020), WHO Guidelines for the safe use of wastewater, excreta and grey water - Wastewater in agriculture (WHO 2006). Major scientific databases (Web of Science, CAB, Science direct, Google Scholar), were searched for current scientific information on most water quality issues for irrigation, covering the period from 2000 to 2021.

Methodologies used to develop specific guidelines are discussed further in relevant Sections. The primary emphasis in revising guidelines was on sustainability in agricultural practice, 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.

The guidelines should be used for irrigation water sourced primarily from rainfed natural sources for protection of the soil environment, crop growth and quality, and related natural resources that could be impacted by the use of the irrigation water.

Specific irrigation water quality guidelines for intensive horticultural activities (e.g. hydroponics and glass-house growing), and waste water reuse (recycled water) are not included in this document. The Australian Guideline for Water Recycling (NRMMC et al. 2006) should be considered for irrigation with recycled water.

The guidelines are values below which there should be minimal risk of adverse effects. Further investigation is recommended if a DGV, SGV or trigger value is exceeded, to determine the level of risk.

1.2 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 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 1 Key issues concerning irrigation water quality effects on soil, plants and water resources

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

1.2.1 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. Often irrigation (irrigation induced surface runoff and deep drainage, leaching practice) provides energy for offsite movements of applied fertilisers (typically phosphates and nitrates that are known as contaminants in aquatic systems) which can result in adverse environmental impacts such as eutrophication from these diffuse sources. Consideration should be given to an overall water cycle management strategy (including environmental flows which are required to maintain ecosystems downstream), with an emphasis on maintaining both on-site and downstream water quality.

1.2.2 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), salinity (EC), 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.

1.2.3 Crop tolerance and type

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 and Westcot 1985). Guidelines for crop tolerance to various toxicants are provided including salinity and certain inorganic and organic contaminants.

Climatic conditions

The effects on irrigation management of rainfall, temperature, evapotranspiration and deep drainage past the rootzone 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 properties and solute transport mechanisms within the soil profile. The leaching flux, plant available 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. Climate change associated variations in rainfall patterns and distributions will have a significant impact on irrigation in the future.

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, salt concentrations in the root zone need to be managed by altering the degree of leaching in the soil profile. Drainage is an integral part of sustainable irrigation and poor drainage in irrigated lands could lead to adverse environmental impacts such as rising groundwater, salinity.

Pathogens
The type of crop will also directly impact the required water quality and the potential risks related to microbial contamination.

2 Biological parameters

2.1 Algae

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

2.1.1 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.

2.1.2 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 (Chorus and Welker 2021; Miller and Russell 2017).

2.2 Cyanobacteria (blue-green algae)

Algal blooms should be treated as potentially toxic and the water should not be used for irrigation until the algae are identified and the level of toxin determined.

- **High risk crops**: ***Water contaminated with cyanobacteria should not be sprayed irrigated on vegetables and fruits, or come in contact with plants being grown for food, especially for food that are being eaten raw such as fruits (strawberries, grapes, apples, tomatoes) and leafy vegetables (salads, cabbages, lettuces) and should be avoided if the cell numbers are as follows:***
  - ***Microcystis aeruginosa*****: >50,000 cells/mL, or**
  - ***Dolichospermum circinale*****: >100,000 cells/mL**

  and

  - **10µg/L total microcystins or**
  - **20 µg/L total saxitoxins**

  Where there is no direct water contact with the edible parts of the plant, irrigating with blue-green algae contaminated water is unlikely to be a problem.

- **Pasture irrigation**: ***Spray irrigation of pasture should be avoided or minimised when the following levels are reached:***
  - ***Microcystis aeruginosa*****: >50,000 cells/mL, or**
  - ***Dolichospermum circinale*****: >100,000 cells/mL**

  Stock should be kept off pasture irrigated with blue-green algae-contaminated water for at least two weeks after irrigation.

- **Occupational exposure**: ***The guideline values proposed for various cyanotoxins are as follows:***
  - **Microcystin LR: 24 µg/L**
  - **Cylindrospermopsin: 6 µg/L**
  - **Saxitoxin: 30 µg/L**

2.2.1 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 planktonic genera of concern include *Microcystis*, *Dolichospermum, Chrysosporum, Nodularia* and *Raphidiopsis* (Jones and Orr 1994; McGregor and Fabbro 2000; Negri et al. 2003; Shaw et al. 1999). The benthic genera of concern include *Microseira, Heteroscytonema, Iningainema, Phormidium, Nostoc, Fischeriella* (McGregor and Sendall 2015; McGregor and Sendall 2017; Sendall and McGregor 2018; Gaget et al. 2017; Cirés et al. 2014). In New Zealand, a review of the literature published between 2009 and 2017 showed that the cyanotoxins of concern in freshwater environments included saxitoxins (from *Scytonema* cf. *crispum*), anatoxins (produced by the planktonic species *Microcoleus autumnalis*, previously *Phormidium autumnale*), nodularins and microcystins (from *Microcystis* and *Dolichospermum* genera). No new research had been published on cylindrospermopsins (Wood et al. 2018).

Cyanobacteria are a component of the natural plankton population in healthy and balanced surface water supplies. Cyanobacteria are generally hazardous when present in large numbers (blooms or mats). Planktonic 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). More information on cyanobacteria and cyanotoxins is provided in Section 9.3 of Chapter 3 of the NWQMS Water Quality Guidelines – Livestock drinking water.

Toxic blooms of cyanobacteria can consist of more than one species of cyanobacteria (Van Halderen et al. 1995). For benthic species, toxic conditions can arise even when the water body has temperatures >12°C (Mez et al. 1997). In South Australian conditions, benthic cyanobacterial growth has been shown to reach a peak in autumn and spring seasons (Gaget et al. 2020). Toxic conditions. 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. While no human deaths have been attributed to cyanotoxin ingestion, it can cause a range of other serious health effects (NHMRC and NRMMC 2011). Concerns regarding the effects of elevated levels of cyanobacteria and associated toxins in irrigation waters used on agricultural produce, crops and pastures continue to be raised (Cooper et al. 1996; Miller and Russell 2017; Saqrane and Oudra 2009; Spoof et al. 2020).

While cyanobacterial cells (and algal cells) can also cause problems through clogging of filters, sprays and other equipment, research has indicated that irrigation with cyanobacterial affected water can degrade plant health, growth and produce (Saqrane and Oudra 2009; Miller and Russell 2017; Spoof et al. 2020).

2.2.2 Effect on suitability of waters for irrigation

Uncertainty about the risks associated with low level toxin consumption through food have raised questions in relation to the use of irrigation waters potentially contaminated with toxic cyanobacteria. Existing research has typically focused on microcystins (MC) as it has been demonstrated to be the most common form of cyanotoxin, except in Queensland where cylindrospermopsins are much more common (McGregor, pers. comm.) and has led to livestock deaths (Crush et al. 2008; Chen et al. 2012; Corbel et al. 2014).

Several factors should be considered when assessing the suitability of water for irrigation:

- Exposure through food with the use of water contaminated with cyanobacteria for irrigation

Many cyanotoxins are extremely persistent in the environment, often being resistant to chemical or bacterial degradation (Cooper et al. 1996). Cyanotoxins can persist in soil and can be absorbed by plants long after irrigation with contaminated water ceases (Corbel et al. 2014; Lee et al. 2017). Furthermore, research has shown that many cyanobacteria toxins are persistent against washing (Codd et al. 1999) and even cooking at temperatures up to 300°C (Miller and Russell 2017; WHO 2003). 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.

Spray irrigation of leafy vegetables such as lettuce and cabbages may represent a risk for accumulation of toxic residues (Crush et al. 2008; Jones et al. 1993) and all plants bioaccumulate cyanotoxins from irrigation water differently and at different rates (Chen et al. 2012; Crush et al. 2008).

Research has shown that even using irrigation water of low cyanobacteria toxin (MC) level concentrations, the toxins can bioaccumulate in food crop plants (Spoof et al. 2020).

Some plants bioaccumulate toxins at higher concentrations in their roots rather than leaves (Mohamed and Al Sheri 2009) which can increase human health risks for some vegetables such as carrots, radish and yams (Miller and Russell 2017). A summary of findings regarding bioaccumulation of microcystins by a range of plant species and compiled by Miller and Russell is presented in Table 2 Available data for cylindrospermopsins' accumulation in a range of vegetables or crops has been compiled in Table 3

No reliable studies on saxitoxins or nodularins in edible crops is available to date (Testai et al. 2016). Indeed, there is considerable uncertainty regarding extraction methods, sample cleanup and/or analytical methods, undermining the confidence in the data on toxin levels in food (Testai et al. 2016, cited by WHO 2020a).

Table 2 Accumulation of microcystins in food crops via irrigation with cyanobacteria-contaminated water.

Irrigation dose (µg/L)	Bioaccumulation concentrations (range, µg/kg)	Source
Lettuce		
Total MC 0.65, 2.5, 6.5, 13	Not detected at lowest doses, i.e. <0.65	Bittencourt-Oliveira et al. (2016)
MC-LR 0.5, 2, 5, 10	39.31 – 158.35 FW	
MC-RR 0.15, 0.5, 1.5, 3	32.00 – 143.35 FW	
	6.32 – 15.00 FW	
MC-LR 5, 10	~50 – 100 FW	Cordeiro-Araújo et al. (2016)
Total MC 0.62, 2.5, 6.23, 12.5	8.31 – 177.8 FW Greater accumulation occurred at higher irrigation doses	Hereman and Bittencourt-Oliveira (2012)
MC-LR 3.23±0.11 µg/mg* DW (bloom and scum)	94 – 2487 DW	Codd et al. (1999)
Total MC 2100	300-870 DW (roots) 740-840 DW (shoots)	Crush et al. (2008)
Carrot		
MC-LR 10, 50	5 FW	Machado et al. (2017)
Radish		
Probably MC-YR 0.3 – 1.8	<300 – 1200 FW	Mohamed and Al Shehri (2009)
Broccoli		
Total MC 1, 10	Only MC-LR detected, only detected in plant roots. 0.9 — 2.4 FW. Did not distinguish bioaccumulation concentration based on irrigation dose	Järvenpää et al. (2007)
Tomatoes		
MC-LR 100	FW	Gutierrez-Praena et al. (2014)
Roots	ND – 1635.2	
Leaves	ND – 12298.2	
Fruit – green tomato	<LD – 5.4	

Fruit – ripe tomato	<LD – 10.8	
MC-LR 100	1.6-3.9 µg MC-LR/kg DW in soil 4.5-8.1 DW (roots) 0.29-0.55 DW (shoots) Bioaccumulation: roots>shoots>stems	Corbel et al. (2016)
Rice		
MC-LR 24, 120, 600, 3000 µg/L	MC-LR (lowest dose) not detected (highest dose) 29.4 – 54 µg/kg FW	Chen et al. (2004)

Adapted from Miller and Russell, 2017

Table 3 Accumulation of cylindrospermopsins (CYN) in vegetables and herbs when experimentally grown in or irrigated with CYN contaminated water.

Concentration range for toxins (µg/L)	**Accumulation in tissues (range, µg/kg) and comments**	**Source**
Spinach, Lettuce		
10 and 50	up to 41.9µg/kg FW (CYN) in lettuce leaves up to 24µg/kg FW (CYN) in spinach roots up to 120µg/kg FW (CYN) in spinach leaves growth inhibition of leaves at 50µg/L of either treatment up to 24µg/kg FW (CYN) in spinach roots CYN translocated from roots to leaves	Llana-Ruiz-Cabello et al. (2019)
Lettuce, rocket		
3, 5, 10	Accumulation in both vegetables decreased with increasing concentration in water Bioaccumulation factor increased with CYN concentration in water in rocket (not in lettuce)	Cordeiro-Araujo et al. (2017)
Brassica species (kale, mustard species)		
30	2.71µg/kg FW (kale) 3.78µg/kg FW (mustard species)	Kittler et al. (2012)
Coriander, parsley		
0.1-1µg/ml	No accumulation at any of the concentrations used	Pereira et al. (2017)
Tomato, cucumber		
2, 5, 10	Root length and germination percentage altered by CYN in water Plant biomass significantly affected by CYN presence in water	Temsah et al. (2016)

Jones and Orr, (2000, cited by EPA Tasmania 2011) therefore recommend that water used for irrigation of crops or plants that are eaten raw or where water directly comes in contact with the fruit/vegetable do not contain more than:

- *M. aeruginosa*: >50,000 cells/mL, or
- *Dolichospermum circinale*: >100,000 cells/mL,

and

- 10 µg/L total microcystins or
- 20 µg/L total saxitoxins

WHO (2020a) concludes, as a preliminary assessment considering all types of cyanobacterial toxins, that data so far does not point to a high level of short-term exposure to cyanotoxins **in** crop tissues. If, however, crops are sprayed or irrigated with water contaminated with cyanobacteria and not sufficiently washed, risks may be higher.

- Exposure through pasture and silage feed

Best practices are that spray irrigation of pasture should be avoided or minimised when cyanotoxin cell concentration reaches 50,000 cells/ mL for *M. aeruginosa* or 100,000 for *D. circinale* (Jones and Orr 2000, adopted in the SEQ Water Guidelines 2006)
Stock should be kept off pasture irrigated with blue-green algae-contaminated water for at least two weeks after irrigation (DPI NSW 2017; Agriculture Victoria 2020; SEQ Water 2006) recommend at least a two-week withholding period if sunny conditions prevail during this time and up to four weeks if moderately sunny. More guidance on guideline values for livestock is available in Chapters related to Livestock drinking water quality of the Water Quality Guidelines (ANZECC and ARMCANZ 2000).

- Occupational exposure

Cyanotoxins themselves are not volatile but exposure via inhalation of water droplets and aerosols or direct contact between ears, eyes, mouth or throat with affected waster can happen when using overhead irrigation for instance.

Even though they are not specifically derived for occupational health and safety purposes, the Guidelines for managing risks in recreational waters (NHMRC 2008) specify that fresh recreational water bodies should not contain:

- • ≥10 µg/L total microcystins; or > 50 000 cells/mL toxic Microcystis aeruginosa; or biovolume equivalent of > 4 mm^3 /L for the combined total of all cyanobacteria where a known toxin producer is dominant in the total biovolume; or
- • ≥ 10 mm^3 /L for total biovolume of all cyanobacterial material where known toxins are not present; or
- • cyanobacterial scums consistently present.

Additionally, some provisional guideline values have been established for cyanotoxins other than microcystins, such as cylindrospermopsins and saxitoxins (WHO 2020a,b) (Table 4).

Table 4 Guideline values for selected cyanotoxins in recreational waters

Toxin	Value (µg/L)	Value type
Microcystin LR	24	Provisional guideline value
Cylindrospermopsin	6	Provisional guideline value
Saxitoxin	30	Guideline value

Source: WHO (2020a and b)

Furthermore, the following practices should be adopted as much as possible (Jones and Orr 2000):

- Restrict spray irrigation when spray drift can be minimal
- Where spray drift is likely, workers should be made aware of the risks and avoid working in the vicinity and wear appropriate face masks

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. Should concerns be raised for potential contamination of crops from cyanotoxins, leaf and root tissue and soil tests should also be carried out (Corbel et al. 2014; Malakar et al. 2019).

A major constraint to the management and control of cyanobacterial blooms is the risk of releasing toxins into surrounding waters. Additionally, it is recommended that treated water sources (with simazine for instance) not be used for irrigation for at least ten days to two weeks to ensure that the effects of both the pesticide and the toxins have dissipated (DPIRD 2020; EPA Tasmania 2011). However, in some cases a longer withholding period may be needed; for example, microcystins have been known to survive for longer than three weeks (Jones and Orr 1994).

However, drinking water guidelines and a total daily intake (TDI) value are, at this stage, unable to determine a safe cyanotoxin concentration for irrigation waters given the diverse range of cyanotoxins and the differing bioaccumulation within different plant species (Corbel et al. 2014).

2.3 Human and animal pathogens

Default guideline values for Escherichia coli (E.coli) in irrigation waters are provided in **Table 5.**

Table 5 Default guideline values for *Escherichia coli* in irrigation waters sourced from rainfed natural sources used for food and non-food crops

Intended Use	Site preventative measures	DGV * (*E. coli* / 100mL)
Commercial food crops consumed raw or unprocessed	None required, water can be used for all crop applications, including spray irrigation of salad crops	1
Commercial food crops **	If spray irrigation a minimum of 2 days between final irrigation and harvest (no harvesting of wet produce). Above-ground crops with subsurface irrigation. Crops with no ground contact with skins removed before consumption (e.g. Citrus).	100
	No contact of irrigation water or irrigated soil with harvestable part of crop (e.g. apples, apricots, grapes, citrus, nuts), or Crops processed and cooked before consumption (e.g. wine production, potatoes, beetroot).	1000
Livestock drinking water	Water not to be used for consumption by cattle under 12 months of age if the source of water contains animal (e.g. abattoir or saleyard) waste. Monitor and manage algae blooms.	100
Pasture or fodder crop irrigation	Limited withholding periods: Exclude lactating dairy cattle from pasture for four hours or until pasture is dry. Suitable for direct grazing by animals with consideration of notes below *	100

(including hay, silage, and commercial fodder production).	5 day withholding period: Exclude grazing animals for 5 days after irrigation. Fodder dried and ensiled (not for human consumption)	1000
Non-food crops (e.g. trees, woodlots, flowers, cotton)	no access during irrigation. If public potentially in vicinity and spray irrigation a minimum of 25 m buffer to nearest public access is required.	10000

Adapted from Australian Guidelines for Water Recycling (AGWR) (NRMMC, EPHC and AHMC 2006)
Median values (refer to discussion on derivation of guidelines below)
* water sourced from human sewage requires the log reduction values as defined in the AGWR (NRMMC, EPHC and AHMC 2006), helminth controls may also be required as specified for livestock exposure, and supply to pigs is prohibited.
**Production of all food crops should meet the relevant industry food production water quality requirements

2.3.1 Description

The presence of human and animal pathogens in irrigation waters is an important issue in agricultural water quality management, particularly with the overall trend towards decreasing water quality and the increasing recycling of water from 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 and groundwater.

Pathogens are capable of survival (and at times multiplication) in the environment for long enough periods to allow for transmission to humans, with several factors influencing their die off; temperature, moisture, exposure to ultraviolet radiation, time, absence of appropriate intermediate hosts, type of plant, soil characteristics, indigenous microflora, etc (WHO 2006; Murphy 2015; HI 2020; Tran et al. 2020). More specifically, factors that affect pathogen survival in the environment include:

- type of pathogen;
- water quality;
- soil characteristics;
- temperature, humidity, precipitation, solar radiation;
- foliage/plat type
- the nature of agriculture and animal husbandry;
- method of transportation of irrigation water;
- irrigation method;
- treatment and storage for pathogen die-off and
- competition with native flora and fauna

Water-borne pathogens of concern to human and animal health in Australia comprise a range of micro-organisms including bacteria, viruses, protozoa and helminths (NRMMC et al. 2006). 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 with the water remaining on the surface of irrigated land (e.g. playground) or edible produce. Prevention of exposure of the pathogen to the plant is a well accepted approach to minimise infection risk to crops (FAO/WHO 2008; Alegbeleye et al. 2018). Pathogens transported via droplets and 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 and 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 and Bole 1976; Alegbeleye et al. 2018). Bacteria are responsible for most reported outbreaks.

Viruses consist of a strand of genetic material with a lipoprotein capsid and an envelope in the case of enveloped viruses. They act by invading the host cell, subsequently modifying its behaviour to produce more viral particles (Metcalf and Eddy 1991). Pathogenic viruses occur in natural waters largely as a result of contamination with sewage and animal excreta and should not normally be present in irrigation waters in large numbers unless the irrigation source is contaminated.

Problems can occur if viruses are present in irrigation waters applied to crops which are directly for consumption without further treatment, as they have been known to persist on vegetation for several weeks or months (WHO 2006; Alegbeleye et al. 2018). 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 (Alegbeleye et al. 2018). Survival is also influenced by temperature, solar radiation, wind, rainfall, humidity, concentration in water and irrigation method. There is evidence that viruses may persist for longer in soil than on aerial vegetable surfaces (WHO 2006; Murphy 2015).

Protozoa are single-celled micro-organisms without cell walls and include amoebas, flagellates and ciliates. This group is responsible for the majority of diarrhoea related illnesses in humans and includes two pathogenic organisms of particular concern, *Giardia* and *Cryptosporidium*. Cattle and sheep (and to a lesser extent marsupials and rabbits) are likely to constitute the main public health threat from Cryptosporidium due to their prevalence in catchments (Ryan et al. 2014). Whereas Giardia can be sourced from a range of domestic and wild animals (Henderson 2009; Balderrama-Carmona et al. 2017).

As part of their lifecycle, most protozoa form cysts or oocysts that can be transmitted through irrigation of contaminated water. Cysts and oocysts are resistant to some environment conditions and this resistance enables the survival of some species for extended periods of time on crops, pastures and other surfaces (WHO 2006; Alum et al. 2014). Irrigated vegetables and fruit have been implicated in the transmission of several protozoan infections to humans (Froese and Kindzierski 1998).

A number of species of parasitic helminths are endemic in some parts of northern Australia and eggs of a variety of helminths can be transmitted via irrigation water to crops or pastures (NRMMC et al. 2006). However, southern parts of Australia have been shown to be not endemic for helminths of concern in sewage (Stevens et al. 2017).

2.3.2 Effects on human health

Human pathogens that could potentially be found in irrigation waters and their associated health risks are listed in Table 6. Issues concerning livestock health are discussed separately (see Section on Livestock drinking water). 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 (NRMMC et al. 2006; NHMRC and NRMMC 2011; Alegbeleye et al. 2018). For example, contamination of irrigation water by human sewage could increase exposure to a range of pathogens (Table 6).

Table 6 Infectious agents potentially present in domestic wastewater[a]

Organism	Disease	Remarks
Bacteria		
Campylobacter jejuni	Gastroenteritis	Diarrhoea
Escherichia coli	Gastroenteritis	Diarrhoea
Legionella pneumophila	Legionnaire's disease	Malaise, myalgia, fever, headache, respiratory illness
Leptospira	Leptospirosis	Jaundice, fever (Weil's disease)
Salmonella sp.	Salmonellosis	Food poisoning
Salmonella typhi	Typhoid fever	High fever, diarrhoea, ulceration of small intestine
Shigella	Shigellosis	Bacillary dysentery
Vibrio cholerae	Cholera	extremely heavy diarrhoea, dehydration
Yersinia enterolitica	Yersinosis	Diarrhoea
Protozoa		
Balantidium coli	Balantidiasis	Diarrhoea, dysentery
Cryptosporidium	Cryptosporidiosis	Diarrhoea
Cyclospora cayetanensis	Cyclosporidiosis	Severe diarrhoea, stomach cramps, nausea and vomiting lasting for extended periods
Entamoeba hystolica	Amebiasis (Amoebic dysentery)	Prolonged diarrhoea with bleeding, abscesses of the liver and small intestine
Giardia lamblia	Giardiasis	Diarrhoea, nausea
Viruses		
Adenovirus	Respiratory disease	
Enteroviruses	Gastroenteritis, heart anomalies, meningitis	
Hepatitis A virus	Infectious hepatitis	Jaundice, fever
Norwalk agent	Gastroenteritis	Vomiting
Norovirus	Gastroenteritis	
Parvovirus	Gastroenteritis	
Rotavirus	Gastroenteritis	
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

[a](Metcalf and Eddy 2003)

2.3.3 Derivation of guidelines

Expanding interest worldwide in the use of recycled water from wastewaters (e.g. sewage) 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 (NRMMC et al. 2006). Although the present guidelines concern naturally occurring waters rather than recycled waters, the underlying issues regarding risks to human and animal health are similar.

The relevant state guideline or the Australian Guideline for Water Recycling (NRMMC et al. 2006) should be considered for irrigation with recycled water (e.g. https://www.publications.qld.gov.au/dataset/disposal-of-effluent-using-irrigation-technical-guideline, https://www.epa.vic.gov.au/about-epa/publications/1910-2).

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, domestic water supplies are more commonly tested for the presence of *Escherichia coli (E. coli)* to give a general indication of faecal contamination and thus the possible presence of microbial pathogens. *Escherichia coli* is a type of bacteria found in the gut and used as an indicator of faecal contamination of water (NRMMC et al. 2006; NHMRC and NRMMC 2011). The *Escherichia coli* has also been used in this guideline to indicate the level of contamination from humans and warm blooded animals.

In Australia and New Zealand, the management and use of recycled 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 Australian Guidelines for Water Recycling (NRMMC et al. 2006).

It is recommended that a 95th percentile value of *E.coli* be used, based on a number of readings generated over time from a regular monitoring program. If any individual value exceeds the DGV, a repeat sample should be collected to confirm the exceedance and action taken to determine the source of the contamination, and improve the water quality to acceptable levels.

For helminths, a default guideline value of < 1 helminth egg per litre is proposed for the protection of crop consumers, slightly more protective than the ≤ 1 set by the World Health Organisation (WHO 2006), and where *T. saginata* management is required (i.e. there is the risk of domestic sewage contamination of the irrigation water and cattle are exposed to the irrigation site) state or national guideline may require a mean hydraulic retention time of 25 days in an appropriately designed and managed lagoon system, which equates to approximately 4-log removal of helminth ova (NRMMC et al. 2006), or an approved risk-based management strategy for the irrigation scheme (Stevens et al. 2021). Insufficient information is available to set guidelines for protozoa and viruses in irrigation water, However, irrigators should consider relevant food quality assurance programs that might be required for specific crops grown.

2.4 Plant pathogens

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

2.4.1 Description

Agricultural crops and pastures can be affected by various plant pathogens transmitted through a number of different pathways including irrigation water (Alegbeleye et al. 2018). 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. However, plant pathogens in irrigation water used for intensive agricultural and horticultural industries (particularly where irrigation water runoff is reused), can potentially lead to crop damage and economic loss (Hong and Moorman 2005).

Most plant diseases are favoured by conditions of rain, high air humidity and high soil moisture. In particular, the virulence of pathogens that infect aerial tissues is greatly promoted by rain and high humidity (Velásquez et al. 2018).

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.

2.4.2 Effect on irrigation water quality

Plant pathogens of potential concern to irrigation water quality commonly include viruses, fungi, bacteria, nematode and some fungi like species (Hong and Moorman 2005) (Table 7). Plants can exhibit a number of symptoms in response to pathogenic infection, including; growth suppression, discoloration, deformations, necrosis, impaired reproduction, overgrowth, rot and scab (Nazarov et al. 2020).

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 oomycetes) that live freely in water and terrestrial systems, such as Phytophthora, Pythium and Aphanomyces (Bodah 2017). Depending on weather conditions and the phytosanitary condition of crops, the prevalence of diseases can be significant and lead to large decreases in crop yields (Nazarov et al. 2020). Plant diseases cost Australia millions of dollars each year as they reduce productivity, increase the cost of production, impact on our ability to trade both locally and internationally and adversely affect our environment and biodiversity (Department of Agriculture and Water Resources 2016).

Table 7 **Examples of plant pathogens found in irrigation water**[a]

Fungi-like (oomocytes)	*Phytophthora cactorum*	*Phytophthora tropicalis*	*Pythium mamillatum*
	Phytophthora cambivora	*Phytophthora undulata*	*Pythium middletonii*
	Phytophthora capsici	*Pythium acanthicum*	*Pythium monospermum*
	Phytophthora cinnamomi	*Pythium aphanidermatum*	*Pythium myriotylum*
	Phytophthora citricola	*Pythium catenulatum*	*Pythium paroecandrum*
	Phytophthora citrophthora	*Pythium coloratum*	*Pythium rostratum*
	Phytophthora cryptogea	*Pythium debaryanum*	*Pythium salpingophorum*
	Phytophthora drechsleri	*Pythium deliense*	*Pythium spinosum*
	Phytophthora gonapodyides	*Pythium diclinum*	*Pythium splendens*
	Phytophthora megasperma	*Pythium dissotocum*	*Pythium sylvaticum*
	Phytophthora nicotianae	*Pythium flevoense*	*Pythium toruloides*
	Phytophthora palmivora	*Pythium graminicola*	*Pythium tracheiphilum*
	Phytophthora ramorum	*Pythium inflatum*	*Pythium vexans*
	Phytophthora syringae	*Pythium irregulare*	
Fungi	*Alternaria*	*Curvularia*	*Plasmodiophora*
	Ascochyta	*Diplodia*	*Rhizoctonia*
	Aspergillus	*Fusarium*	*Rhizopus*
	Botrytis	*Geotrichum*	*Sclerotium*
	Cephalosporium	*Gliocladium*	*Scopulariopsis*
	Chaetomium	*Microsporium*	*Stemphyllium*
	Cladosporium	*Mucor*	*Trichoderma*
	Colletotrichum	*Penicillium*	*Verticillium*
	Coniothyrium	*Phoma*	*Whetzelinia*
Bacteria	*Corynebacterium flaccumfaciens*	*E. chrysanthemi*	*Ralstonia solanacearum*
	Erwinia carotovora	*Erwinia spp.*	*Xanthomonas campestris pv. begonia*
	E. carotovora pv. Carotovora	*Pseudomonas syringae pv. Aptata*	*X. phaseoli*
	E. carotovora pv. zea		
Virus	Pelargonium flower break	Tomato mosaic	Lettuce big vein
	Tomato bushy stunt	Cucumber green mosaic	Tobamovirus Lake
	Carnation mottle	Tobacco necrosis	Potexvirus

Nematodes	*Aphelenchoides*	*Heterodera*	*Pratylenchus*
	Criconemoides	*Hoploliamus*	*Trichodorus*
	Ditylenchus	*Meloidogyne*	*Tylenchorhynchus*
	Helicotylenchus	*Paratylenchus*	*Tylenchulus*
	Hemicycliophora		

a Adapted from Hong and Moorman (2005)

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.

3.1 Description

Salinity is the presence of soluble salts in waters. High levels of soluble salts in the soil solution may result in reduced plant productivity or at worst the death of crops and native vegetation. Salts are derived from the weathering of the earth's crust and are transported and cycled in the broader environment through rainfall and the movement of water. Other sources of salt inputs into the rootzone are irrigation water, fertilisers, soil conditioners, rising groundwater for instance. Dryland salinity is a result of salts stored into the ground brought closer to the surface due to a rising watertable. When the hydrologic balance of a landscape is altered through human induced disturbances such as tree clearing or irrigation, new hydrologic and solute equilibriums are established with a subsequent translocation of salt to/from the soil and the receiving water environment.

There are two different situations leading to soil salinization due to irrigation:

Under general irrigation practice, the addition of water can result in the elevation of the local watertable, creating potential capillary rise of water and salt into the crop root zone and/or soil surface. Even a low salinity ground water can cause root zone and surface salting.

Soil salinity increase can also occur from insufficient leaching of the salt added in the irrigation water.

In both cases, off-site degradation of surface or groundwaters may occur, as well as 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 watertables within two metres of the soil surface and this could increase to 869,000 hectares by 2015 (MDBA 2015). 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.

Sodicity is a condition that degrades soil properties by promoting enhanced swelling of clay particles leading to clay dispersion from soil aggregates. This dispersion reduces soil structure (i.e. tilth), hydraulic conductivity (the ability of the soil to conduct water), impedes internal drainage and restricts water entry and seedling emergence due to hydraulically retarding soil crusts.

A higher proportion of sodium on the soil exchange complex (ie CEC) promotes clay dispersion and, once dry, soils usually become dense, cloddy with a much reduced tilth because of destruction of the natural soil aggregation processes. Because the relative proportions of exchangeable cations in a soil are determined by the relative concentration of cations Ca, Mg and Na in the soil solution (as measured by its Sodium Adsorption Ratio (SAR)), the composition of irrigation water influences soil sodicity (Rengasamy and Olsson 1995).

Sodicity of a soil is measured by the Exchangeable Sodium Percentage (ESP) whilst that of the irrigation water is measured by its SAR.

3.2 Factors affecting irrigation salinity

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

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

These are discussed separately within this Section to allow a better understanding of the processes involved. The methodology described here aims to assess a realistic irrigation environment, through increased emphasis on the role of soil properties in sustainable irrigation. Figure 1 illustrates the interactions of various processes in relation to salinity and sodicity.

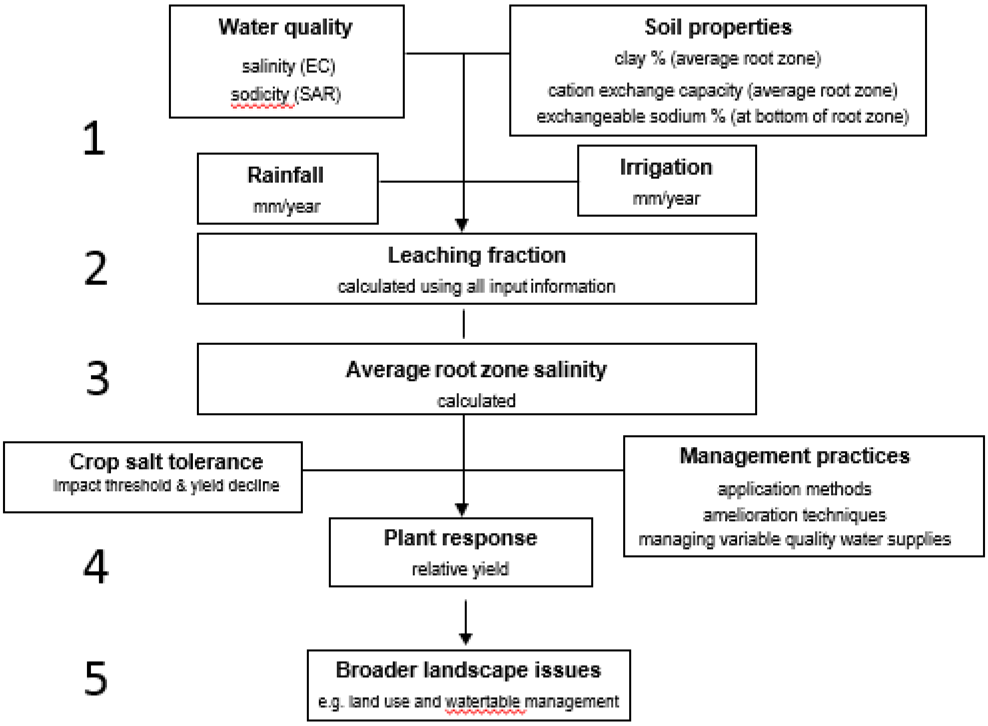


Figure 1 Flow diagram for evaluating salinity and sodicity impacts of irrigation water quality

There are five key steps to determining the suitability of irrigation water with respect to salinity and sodicity (Figure 1).

Prior to the following steps, the crop selection comes first. Generally, irrigators start with the crop selection (for instance a grower is growing lettuce, or pomegranates or grapes, etc, and their respective rootzone threshold salinity is known) the leaching fraction in order to maintain suitable root zone salinity conditions, and finally a consideration of where the salt moves. The Leaching Fraction requirement is calculated using the following steps:

- ***Step 1*** Identify the soil hydraulic properties, water quality, climate (rainfall and evaporation), crop and management (irrigation application methods and rates) practices for the site in question.
- ***Step 2*** Estimate the leaching fraction under the new/changed irrigation regime utilising approaches outlined later in this Section.
- ***Step 3*** Estimate the new average root zone salinity as outlined later in this Section. Average root zone salinity (at saturation extract moisture content) is considered the key metric to assess plant growth in response to salinity levels in the irrigation water.
- ***Step 4*** Estimate relative plant yield (but noting that the impact of salinity and sodicity can be modified by management practices as discussed later in this Section).
- ***Step 5*** Consider salinity problems within the framework of broader catchment issues such as rising regional watertables, groundwater pollution and surface water quality. Watertable salinity (capillary rise) develops because shallow watertables interact with soil hydraulic properties and evaporative demand cause the upward movement of water and salt. It is the greatest threat to the ongoing sustainability of most irrigation schemes in the world.

Before an irrigation scheme is developed, the planning process must include investigation of the regional hydrogeology to minimise the development of salinity, or plan the installation of subsurface drains to limit the groundwater rise to <2m below the soil surface. Of course, the environmental impact of disposing of this saline drainage water needs to be carefully assessed. The guidelines discussed in this chapter concentrate on localised impacts of irrigation, but broader issues of landscape hydro-salinity issues should not be ignored.

The details of this methodology follow, with some worked examples provided later in Section 3.3. The Software *SALF2 (*replacing SALF PREDICT*)* estimates the parameters necessary for a detailed assessment of irrigation water quality in relation to soil properties, rainfall, water quality and plant salt tolerance. The software, which includes numerous empirical statistical functions, is based on summer rainfall areas and should be used with some caution in winter rainfall areas. It incorporates many of the detailed algorithms described in this Section. Copies of the software may also be obtained from its author Dr Roger Shaw or the Queensland Government website: Salinity management handbook | Publications | Queensland Government.

Other useful software for irrigation management and crop selection include MEDLI (www.des.qld.gov.au/science/government/science-division/medli). SWAGMAN Destiny, SIRAG, RUSTIC, WATERSCHED (www.waterschedpro.net.au).

3.2.1 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 ($EC_{1:5}$) measures the ability of water to conduct an electric current, which is carried by ions in solution such as chloride, sodium, sulfate, nitrate, carbonate, bicarbonate, calcium and magnesium. In irrigated agriculture fertigation through the irrigation system is widely practiced and the salinity impacts of fertilisers in water and soil need to be considered in the assessment of potential impact. Indeed, fertilizers are salts and these salts can impact seedlings health through an osmotic effect.

Electrical conductivity is commonly used as an estimate of the concentration (mg/L) of total dissolved salts (TDS) and is expressed in deci-Siemens per metre (dS/m) or micro-Siemens 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 conversion factors other than 640 can sometimes be used to convert EC to TDS (for ECs greater than about 5dS/m, a conversion factor of 800 is suggested – Tanji et al. 2008), it is recommended that only directly measured EC data be used.
The following are some useful conversions are:

- 1 dS/m = 1000 µS/cm
- EC (dS/m) x 640 = TDS (mg/L)
- EC (µS/cm) x 0.64 = TDS (mg/L)
- In summary: 1 dS/m = 1000 µS/cm ≈ 640 mg/L = 640 ppm.

The three measures of electrical conductivity are (DERM 2011):

- EC1:5: the electrical conductivity of a 1:5 soil water suspension, used routinely in analyses
- ECse: the electrical conductivity of the soil saturation extract, used for predicting plant response (commonly predicted from 1:5 and soil properties, or can be measured directly)
- ECs: the electrical conductivity of soil at measured or predicted maximum field water content (approximating field capacity), used to assess salt movement through the soil (usually predicted from 1:5 and soil properties)

A generic water salinity rating can be assigned to irrigation waters based on EC (Table 8). These ratings provide only a general guide and should not be used by themselves to assess the salinity hazard of irrigation waters. As emphasised throughout this section, other factors such as plant species, leaching fraction, soil characteristics, climate, and irrigation management must be considered.

Table 8 **Irrigation water salinity ratings based on electrical conductivity**^{a,b}

EC (dS/m)	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 Criteria for salinity are based on 90% yield of the plant groupings of Maas and Hoffman (1977) and assuming 15% leaching fraction

The EC_{se} metric is one of the critical measurements used in salinity assessment and its biological implications is assessed using published plant salt tolerance tables (see later discussion on soil properties).

Sodicity assessment

Sodicity is the concentration of sodium (Na^+) ions relative to other cations in the irrigation water or on the soil exchange complex. Sodium on the soil cation exchangeable complex acts as a dispersant. However, soil salinity concentration (EC_{se}) also acts as a flocculant of soil particles and Na is a contributor to this salinity.

Elevated levels of Na^+ in irrigation water can lead to sodicity problems in the soil profile. The sodicity levels in irrigation water is quantified by the Sodium Adsorption Ratio (SAR). It is calculated using the following equation, noting that ionic concentrations are expressed in meq/L:

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

Equation 1

Where Na^+, Ca^{2+} and Mg^{2+} are expressed in $mmole_c$/L (where subscript c indicates charge).

Note: 1 meq/L = 1mmol/L

If mg/L units are used, the equation becomes

$$\mathrm{SAR} = \frac{\frac{\mathrm{Na}}{23}}{\sqrt{\frac{\frac{\mathrm{Ca}}{20} + \frac{\mathrm{Mg}}{12.2}}{2}}}$$

Equation 2

Where units for Na, Ca and Mg are expressed in mg/L. SAR is dimensionless

The SAR value is used to predict soil sodicity (measured as ESP) which in turn can be assessed for its implications in maintaining soil structural stability. Clay mineralogy (usually expressed as the ratio of CEC to Clay percentage (CCR) with units of $mmole_C$ /kg soil) and clay content are combined to assess the SAR limits of irrigation water to ensure soil structural stability as shown in Table 9. However, as the salt concentration in irrigation water acts as a flocculant, both EC **and** SAR need to be considered in the final assessment of water quality suitability. Note that if the soil is a sand, water SAR and soil ESP do not have any impact on soil structure (sand is structureless and is highly drainable by nature) in which case Na toxicity on selected crop need only be considered.

Table 9 **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 (mmol_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	8	5	5	4	4
65–75	Heavy clay	--	4	4	4	4
75–85	Heavy clay	--	--	4	5	5

a From DERM (2011); 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

Some irrigation waters, due to their source (recycled water for instance) can contain high cation concentrations other than sodium (eg potassium, K) in which case it would be more relevant to consider the cation ratio of soil structure stability (CROSS), rather than SAR (Rengasamy 2010) as it provides a more complete assessment of the impact of monovalent cations on soil structural stability

$$\text{CROSS} = \frac{(\text{Na} + 0.56\text{K})}{\left[\left(\frac{\text{Ca} + 0.6\text{Mg}}{2}\right)\right]^{\frac{1}{2}}} \qquad \textbf{Equation 3}$$

CROSS is dimensionless

3.2.2 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 fraction (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 1. The different categories of salt affected soils depending on EC, SAR and pH is presented in Figure 2.

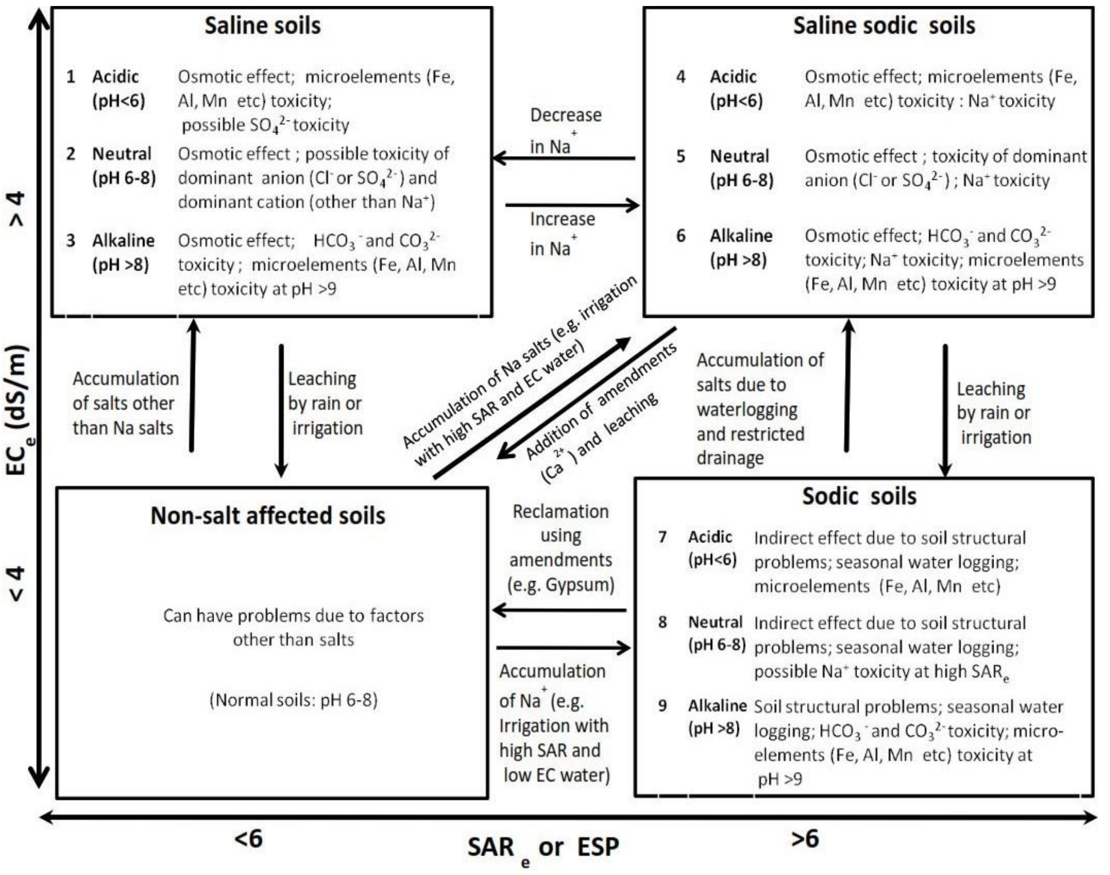


Figure 2 Categories of salt affected soils based on SAR, EC_{se} and pH and their potential impacts on plant growth

After Rengasamy (2010)

Soil salinity assessment

Leaching fraction (LF) and EC are two critical parameters in assessing soil salinity hazards. The leaching fraction is the term given to the portion of applied water (effective rainfall + irrigation) that drains through the root zone to help maintain soil salinity at acceptable concentrations (DERM 2011). These parameters form the basis of predicting soil root zone salinity (as measured by 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).

- Osmotic Effect

High concentration of salts in soil water lowers the water potential gradient between the soil and plant root thereby affecting the readily available water (Figure 3).

Plants draw water from the bulk soil by creating an osmotic potential gradient (difference) across the soil/root boundary where a higher concentration of salt must exist within the root for water uptake to occur. In a saline soil, given that the osmotic potential across the soil/root boundary is decreased, the plant becomes stressed and for water uptake to continue there must be some osmotic adjustment, where the plant absorbs and compartmentalises salt or organic solutes, thus restoring the osmotic potential required for water uptake (Bernstein 1975, cited in GRDC 2009).

In saline soils, soils may appear wet yet crop cannot extract adequate water. The impact of this osmotic "drought" can be greater than the toxicity effects of Na and Cl especially in dryland agriculture. In the short term keeping the soil profile wet near field capacity mitigates the impact of the osmotic effect. The impact of osmotic effect can be exacerbated if irrigation water includes fertilizer (fertigation) or amendments such as soluble Ca salts to correct sodicity issues.

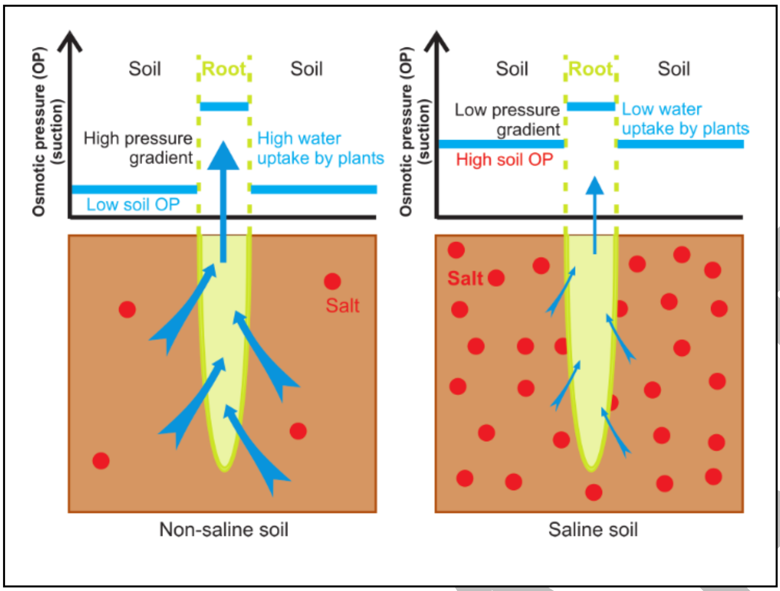


Figure 3 The relative water uptake by plant roots in saline and non-saline soil

Source: Kelly and Rengasamy (2006)

Note: OP = osmotic potential with units of KPa

As mentioned above, the osmotic adjustment from the plant requires solutes accumulations (ions or organic solutes) and ions compartmentalisation. These processes require energy from the plant (ATP production) which is costly. It has been suggested by Grattan et al. (unpublished) and Munns et al. (2020) that ultimately, the more energy the plant spends osmoregulating, the less it has to spend on growth processes and that the osmotic drought might in fact be due to the energy expense to regulate it.

Indeed, when looking at the relationship between the osmotic potential and EC (
Figure 4 - an ECse of 5 dS/m is only -2 bar "increasing" to -4 bar at 10 dS/m). These are not values that should cause much of an effect on soil water availability given plants can dry soil down to -15 bar and beyond. It is not a big challenge for the plant to be able to osmoregulate at these osmotic soil water

potentials. This emphasises the hypothesis by Grattan that the energy spent in osmotic adjustment as the main cause of reduced plant growth.

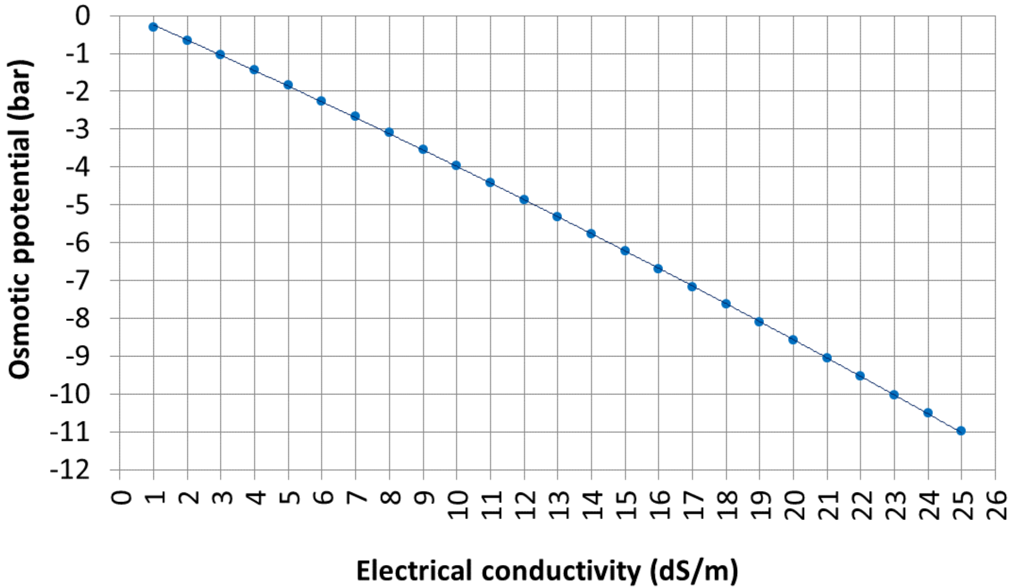


Figure 4 Conversion of soil electrical conductivity (EC) to soil osmotic potential (Ψ)

- Leaching fraction

Leaching fraction (LF) is defined as the proportion of applied water (irrigation + rainfall) that drains below the root zone in the soil profile (see Figure 5). It is often expressed as a percentage (see Equation 4).

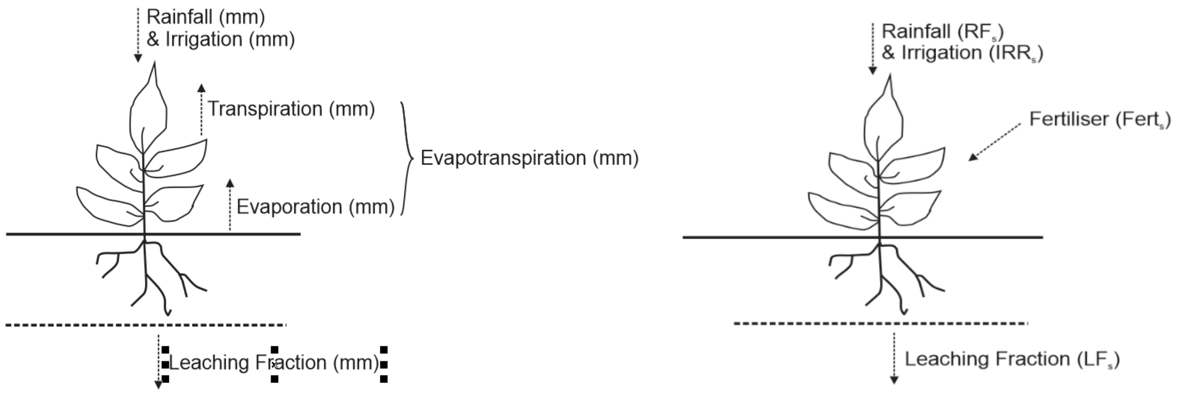


Figure 5 Schematic representation of the water balance (left) and the salt balance (right) within the soil profile

Note: leaching fraction is part of the deep drainage (D_d)

Prediction of LF is fundamental to irrigation salinity risk assessment, as it allows calculation of root zone salinity and its consequential effects on plant production. A simple empirical approach estimates LF by considering the mass balance of salts under steady state conditions. That is, equilibrium occurs between the inputs and the outputs of salt after a given period (years), at which point the accumulation of further salt storage in the soil becomes zero.

The leaching fraction does not need to be applied with every irrigation event: strategic leaching irrigations can be applied as required, such as when the EC_{se} approaches the plant salinity threshold (see Table 14 and Table 15). Calculation of the leaching fraction after the (summer or winter) wet season may give a better result as the average deep drainage from rainfall and irrigation water will be higher than irrigation water alone, thereby increasing the calculated leaching fraction and hence salt leaching.

Irrigation managers can apply an adaptive management approach with respect to LF by measuring root zone salinity and applying an excess irrigation to increase the deep drainage and hence increase the leaching fraction. This potentially has the advantage of managing to the crop's specific salinity requirements, and mitigating the groundwater risks associated with excessive deep percolation. This is effectively the approach pioneered by the US Salinity Lab (1954) that based their recommendations on irrigation schemes operating in an effectively desert Californian landscape.

Leaching fraction can be estimated most easily by calculating the ratio of salt inputs (in rainfall and irrigation) to outputs (in deep drainage). On the assumption that the water draining at the bottom of the root zone does no change further with soil depth, consideration of mass balance at equilibrium allows us to write:

$$\mathrm{ECi \ x \ Di = ECd \ x \ Dd}$$

which on rearrangement can be expressed as:

$$\mathrm{LF} = \frac{\mathrm{ECi}}{\mathrm{ECd}} = \frac{\mathrm{Dd}}{\mathrm{Di}}$$ **Equation 4**

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 (measured)

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

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

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

$$\mathrm{EC_i} = \frac{(\mathrm{EC_r} \times \mathrm{D_r}) + (\mathrm{EC_{iw}} \times \mathrm{D_{iw}})}{(\mathrm{D_r} + \mathrm{D_{iw}})}$$ **Equation 5**

where:

EC_i = electrical conductivity of water infiltrating soil (rainfall + irrigation) with units of dS/m (measured)

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 (measured)

D_r = rainfall depth in mm/year (measured)
D_{iw} = depth of irrigation water applied to the soil profile in mm/year (measured).

Note that it is assumed that all the applied irrigation water and rainfall infiltrates the soil. Clearly this is an approximation as rainfall runoff occurs from most soils whilst furrow irrigation is rarely 100% efficient in terms of infiltration.

- LF prediction models

Irrigation water salinity can be predicted using several approaches as outlined in Table 10.

Table 10 Summary of methods and data required for estimating leaching fraction for different conditions[a]

Condition model	**Field data**	**Method**
Prior to irrigation Shaw and Thorburn (1985); Shaw (1996)	(a) EC and amount of irrigation water, rainfall, $EC_{1:5}$ at bottom of root zone, maximum field water content (b) Clay and CEC from 0 to 0.9 m, ESP at 0.9 m, annual rainfall and irrigation and EC of irrigation.	Leaching fraction predicted from soil properties and water application. 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
Changed irrigation water salinity Shaw and 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 deep drainage which in turn is adjusted for changes. resulting from irrigation water salinity and sodicity.

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 predicting the steady state LF at the bottom of the root zone that would occur under irrigation.

The soil properties of dominant influence on deep drainage are clay content, clay mineralogy (CCR) (expressed as CEC/clay ratio, mmole_c/kg of clay), and the exchangeable sodium percentage (ESP). 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 the increase in soil solution electrolyte concentration will result in a change in leaching for a given soil ESP, an adjustment of the predicted leaching fraction is made when 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 rainfall conditions and is calculated from the general equation for each soil group (as given in Table 11) using the general form:

$$\mathrm{LF_r} = \frac{\mathrm{EC_r}}{\mathrm{EC_d}}$$

Equation 6

Note that EC_d in Equation 6 is also the ECs of Table 8.

where EC_d (the *in situ* salinity of the soil solution at moisture contents allowing gravity drainage) can be approximated by 2.2 x EC_{se} (the "measured" EC of the soil saturation extract). Note that 2.2 represents the generalised relationship between water content of the soil saturation extract and maximum soil water content. EC_{se} is better estimated from the easily measured $EC_{1:5}$ using empirical relationships based on other easily measured soil properties (Shaw 1996).
LFr can be predicted from the following equation:

$$\mathrm{LF_r} = \frac{\mathrm{EC_r}}{2.2 \times 10^{\left[\mathrm{a+blog}\left(\frac{0.03\times \mathrm{rainfall}}{\mathrm{ESP}}\right)\right]}}$$

Equation 7

where the coefficients 'a' and 'b' capture the effect of ESP on soil permeability and their values vary with the clay percentage and CCR values (see Table 11).

Table 11 **Parameters used in Equation 8 to estimate LF under rainfall**[a]

Clay content range (%)	Parameter	CCR <0.35 (mmole$_C$ /kg)	CCR 0.35-0.55 (mmole$_C$ /kg)	CCR 0.55-0.75 (mmole$_C$ /kg)	CCR 0.75-0.95 (mmole$_C$ /kg)	CCR >0.95 (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
3-45	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)

Under irrigated conditions the salinity of the infiltrating waters is always larger than that of rainfall and the soil solution electrolyte concentration increases causing a flocculation effect thereby increasing deep drainage. Hence the LF under irrigated conditions, LF_i, can be calculated as:

$$\mathrm{LF_i} = \mathrm{LF_r}\left(\frac{\mathrm{EC_i}}{\mathrm{EC_r}}\right)$$ **Equation 8**

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) noting EC_r = 0.03 dS/m

The ratio of the EC values is always greater than 1, hence LFi will always be greater than LFr.

On the basis of experience with heavy textured soils in the Lockyer Valley using variable salinity irrigation waters a non-linear adjustment was developed by Shaw (1995). Applying the non-linear adjustment to Equation 8, Shaw showed that the adjusted LF under irrigation, LF_f, could be expressed as:

$$\mathrm{LF_f} = \mathrm{Lf_r}\left[2.65\left(\frac{\mathrm{EC_i}}{\mathrm{Ec_r}}\right)^{0.5} - 1.35\right]$$ **Equation 9**

where:

LF_f = predicted LF under irrigation after allowing for irrigation water quality and amount

EC_i = electrical conductivity of water infiltrating the 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 (for instance more than 10 years), steady-state conditions should exist and the following mass-balance equation is valid:

$$\mathrm{LF} = \frac{\mathrm{EC_i}}{\mathrm{Ec_s}}$$ **Equation 10**

where:

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

EC_s is the equivalent of EC_d of

Equation 4 and is determined from soil $EC_{1:5}$ measurements (taken at the bottom of the root zone). However, the $EC_{1:5}$ value needs to be converted to EC_s using the dilution calculations described in the following section. EC_i can be calculated (on an annual basis) using Equation 5.

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, Equation 10 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 described 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 depth of irrigation and salinity (chloride) of the irrigation water, and the maximum field water content of the soil after drainage ceases (the upper storage limit or field capacity).

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 and Gardner 1986). The equation of Rose et al. (1979) is:

$$\mathrm{S_2 = S_1 + \left[\left(\frac{DiS_i}{D_d I} - S_i\right)\lambda - S_i\right]\left[\left|1 - \exp(-D_d \frac{1}{Z\theta})\right|^{t}\right]} \qquad \textbf{Equation 11}$$

where:

S_1 and S_2 are the **mean** root zone salinities determined at two different times,
t is the time between measurements,
z is the depth of root zone,
θ is the volumetric water content at which drainage has essentially ceased (<0.1mm/day),
Dd is the deep drainage, below the root zone,
Di is the depth of infiltration,
Si is the solute concentration of the irrigation water,
λ is a factor to account for the shape of the soil salinity profile.

Since D_d is the only unknown in the equation, it can be calculated by manipulating Equation 11. After D_d is converted to LF, it can be used to calculate the average root zone salinity at steady state conditions. The model can also indicate the time period when steady state conditions will be reached, and how much the soil salinity will increase (or possibly decrease) until that time. If the EC root zone value at steady state is too high for the selected crop to be grown, irrigation management practice will need to be modified.

- Electrical Conductivity of soil

Measurement of soil salinity has traditionally been based on EC and chloride concentration determined in the laboratory using 1:5 soil:water suspension procedure (Rayment and Lyons 2010). 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 growth response and EC_s for salt mass balance calculations.

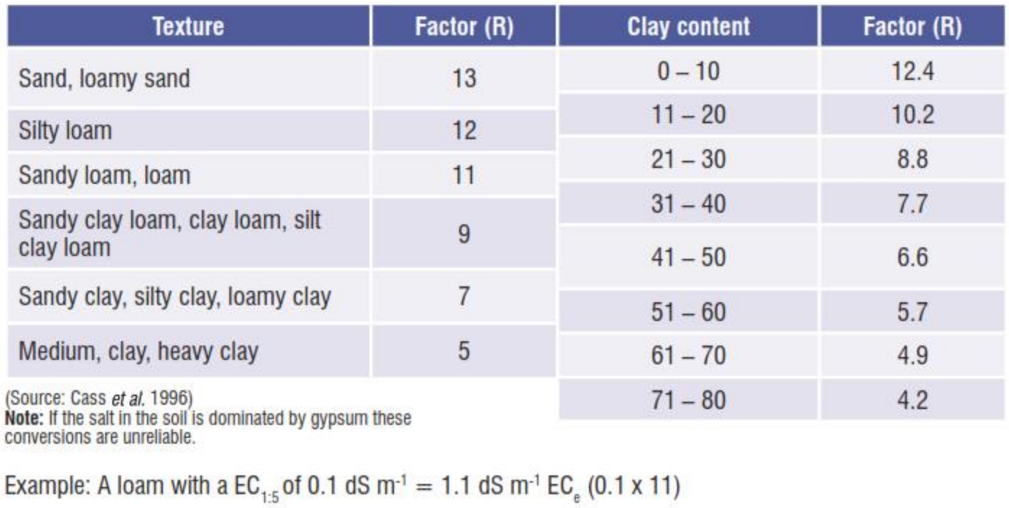
EC_{se} (dS/m) is defined as the electrical conductivity of the soil saturation extract, while EC_s (dS/m) is the electrical conductivity of the soil solution at maximum field water content. Maximum field water content is the maximum measured water content of the soil in the field, two to three days following wetting. It is expressed on a mass basis (g/100 g) and can be considerably lower than the conventional estimate of laboratory 'field capacity' using ground samples equilibrated at -0.1 or -0.33 bar water potential (Gardner and Coughlan 1982).

The soil:water ratio of 1:5 was established in response to practical 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 linked to published soil behaviour and plant responses, as the ratio is far more dilute than is found under field conditions. It is also a fixed ratio 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 and was chosen by the USSL (1954) as it is the lowest reproducible soil water content after which sufficient water can be readily removed from the soil paste for subsequent EC measurement. It also consistently relates to field soil water contents and soil textures (Rhoades 1983).

To assess the effect of soil salinity on the growth of specific crops, the EC$_{se}$ value is required. EC$_{se}$ can be estimated using conversion in Table 12 (Cass et al. 1996).

Table 12 Multiplier factors for different soil textures to convert EC$_{1:5}$ to EC$_{se}$



Texture	Factor (R)
Sand, loamy sand	13
Silty loam	12
Sandy loam, loam	11
Sandy clay loam, clay loam, silt clay loam	9
Sandy clay, silty clay, loamy clay	7
Medium, clay, heavy clay	5

Clay content	Factor (R)
0 – 10	12.4
11 – 20	10.2
21 – 30	8.8
31 – 40	7.7
41 – 50	6.6
51 – 60	5.7
61 – 70	4.9
71 – 80	4.2

(Source: Cass *et al.* 1996)
Note: If the salt in the soil is dominated by gypsum these conversions are unreliable.

Example: A loam with a EC$_{1:5}$ of 0.1 dS m^{-1} = 1.1 dS m^{-1} EC$_e$ (0.1 x 11)

Convective movement of solutes in soils becomes limited once the soil water content is drier than the Upper Storage Limit field water content. The salinity at this water content, EC$_s$, coincides with the water content at which soil profile drainage has effectively ceased. It is used in leaching fraction estimations and in solute movement studies and modelling. Table 13 shows the relative dilutions with respect to field water contents for the three measures of EC.

Table 13 Relative dilution above maximum field water content for three measures of soil salinity, EC$_{1:5}$, EC$_{se}$ and ECs^a

Measure	Dilution above drained upper limit water content
EC$_{1:5}$	5 to >40 times
EC$_{se}$	2 to 3 times
EC$_s$	1 time

a From DERM (2011)

- 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} when combined with the predicted leaching fraction (LF) of the soil under irrigation.

Converting EC_{iw} to EC_{se}

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

$$\mathrm{EC_{se}} = \frac{\mathrm{EC_{iw}}}{2.2 \times \mathrm{LF_{av}}}$$ **Equation 12**

EC_{se} = average root zone salinity (in dS/m)

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

2.2 is a empirical conversion factor which is soil texture independent and

LF_{av} is the **average** LF of the whole root zone and is calculated (following Shaw et al. 1987; Rhoades 1982) as:

$$\mathrm{LF_{av}}(\%) = (0.976\mathrm{LF} \times 0.282)^{0.625}$$ **Equation 13**

where:

LF is the LF at the bottom of the root zone and is calculated as previously discussed.

The EC_{se} value can then be used to match plant species to a particular irrigation situation as described in the plant salt tolerance Table 14 and Table 15.

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

Equation 12 has the limitation that the conversion factor 2.2 is soil texture invariant. As an alternative Shaw (1994) developed an empirical model based where a given mass of dissolved salt in a system at two water contents is represented by:

$$\theta_{se}\mathrm{EC_{se}} = \theta_{1:5}\mathrm{EC_{1:5}}$$ **Equation 14**

This equation can be then rearranged as:

$$\mathrm{EC_{se}} = \mathrm{EC_{1:5}}\left(\frac{\theta_{1:5}}{\theta_{se}}\right)$$ **Equation 15**

where:

θ_{se} = water content (g/g) at the soil saturation extract

EC_{se} = electrical conductivity (dS/m) of the soil solution at the water content Q_{se}

$\theta_{1:5}$ = water content (g/g) of the 1:5 soil water suspension

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

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.
- 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 more accurately estimate EC_{se} from $EC_{1:5}$, the above factors need to be taken into consideration. This can be done by adding an exponent term *c* to the water content ratio term to take into account the solubility effects of different salt concentrations and compositions and the effect of suspended clays.

Equation 15 then becomes:

$$\mathrm{EC_{se}} = \mathrm{EC_{1:5}}\left(\frac{\theta_{1:5}}{\theta_{\mathrm{se}}}\right)^{c}$$

Equation 16

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

The *c* 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 predict EC_{se} from the more easily measured $EC_{1:5}$.

Equation 17

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

where:

ADMC = air dry moisture content at 40°C and expressed as a percentage of the oven dry soil weight (g/100 g). ADMC is a surrogate of clay content and clay mineralogy which is strongly correlated with the ratio $\left(\frac{\theta_{1:5}}{\theta_{se}}\right)$ from Equation 16.

CI% is the measured soil chloride level (in the 1:5 solution) with units of mg/kg soil

The predicted EC_{se} value can then be used to assess the effect of soil salinity on the relative growth of the crop species of interest (via Table 14 andTable 15).

Calculation of EC_S

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

$$\mathrm{EC_s} = 2.2 \times \mathrm{EC_{se}} \qquad \textbf{Equation 18}$$

Soil sodicity assessment

Two common methods of measuring soil sodicity are:

- exchangeable sodium percentage (ESP), which is the amount 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), which is the relative concentration of sodium to calcium plus magnesium in the soil solution (see Equation 1).

- Exchangeable sodium percentage (ESP)

ESP is determined by routine CEC and exchangeable cation methods as outlined by Bruce and Rayment (1982) and Rayment and Lyons (2010). It is calculated using the following equation:

$$\mathrm{ESP} = \frac{\mathrm{Na} \times 100}{\mathrm{CEC}} \qquad \textbf{Equation 19}$$

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 and Higginson 1992) an overestimation of ESP will occur from summation of cations; It can be corrected using exchange acidity measurements (Rayment and Lyons 2010)
- in alkaline soils where Tucker's solution at pH 8.4 (Rayment and Lyons 2010) has not been used to extract cations. In which case, sparingly soluble Ca salts will give inflated Ca and hence an underestimate of ESP.
- In saline soils, unless the soil is prewashed with an alcohol solution, the CEC will not accurately be measured (So et al. 2004; SSA 2013) as the soluble salts are measured as exchangeable cations.

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

The SAR of irrigation water (or soil solution) 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 complex 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).

$$\mathrm{ESP} = \frac{100(-0.0126 + 0.01475\mathrm{SAR})}{1 + (-0.0126 + 0.01475\mathrm{SAR})} \qquad \textbf{Equation 20}$$

The relationship can be seen in Figure 6 and is not very different to a 1:1 relationship.

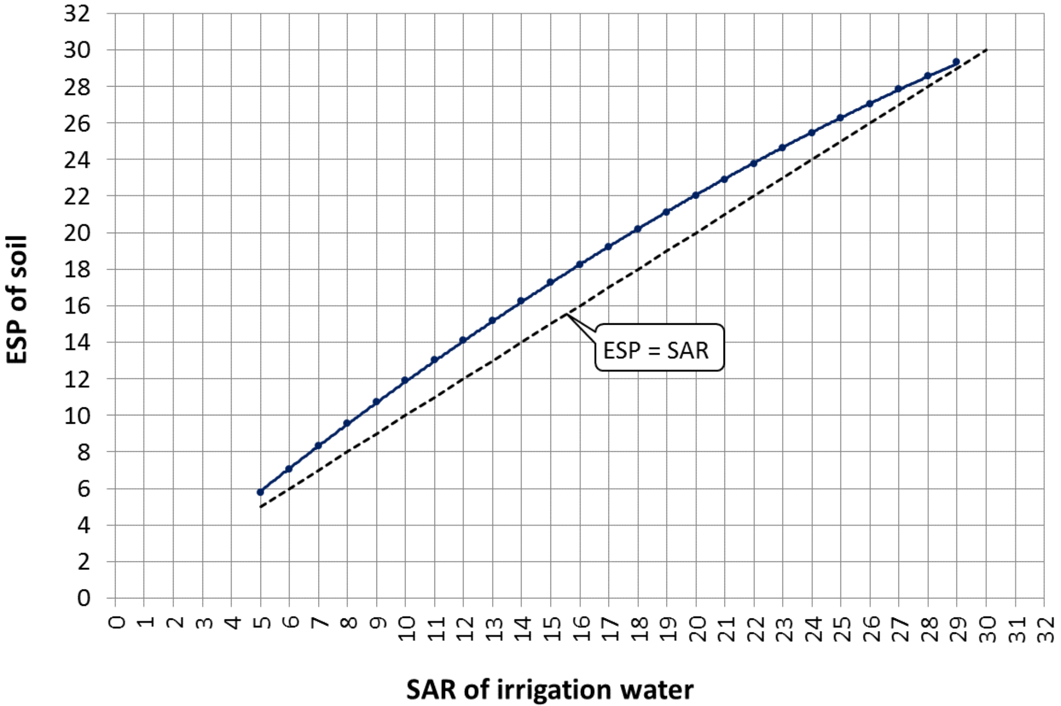


Figure 6 Conversion of irrigation water SAR to soil equilibrium ESP

After T. Kodel, pers. comm.

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 the equivalent of an additional 6 percent of cations to the exchange complex in the top 0.6 m of soil, each year.

However, the correction of soil ESP though the addition of calcium amendments can be achieved in the year of application. The addition of Ca will replace Na on the soil exchange complex. This in turn will improve soil structure and have a positive impact on soil percolation. It is important to note that the addition of calcium will increase soil salinity through the addition of a salt (Ca-x) and increase Na displaced from the exchange complex into the soil solution which can become a problem with Na sensitive crops in the year of application. The increase in salinity and Na availability may need to be considered in the season of application (timing of application of Ca, rate of application of Ca and crop selection).

- Predicting changes in SAR

The SAR of an irrigation water provides an indication of the effect the water is likely to have on the ESP of 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, within 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 respiration, with the result that carbonate salts can remain in solution.

Additionally, the amount of deep drainage (i.e., 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 nominal reflects the highest SAR reached in the soil profile.

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

Equation 21

where:

$\mathrm{SAR_d}$ = SAR of drainage water at the bottom of the root zone

LF = leaching fraction at the bottom of the root zone

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

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

$\mathrm{Ca_d}$ = Ca concentration in the drainage water (in $\mathrm{mmole_c}$ /L)

$\mathrm{Ca_d}$ is predicted from the ionic strength, HCO_3:Ca ratio, and partial pressure of CO_2. $\mathrm{Ca_d}$ 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):

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

Equation 22

where:

$\mathrm{SAR_d}$ = SAR of the deep drainage water at the bottom of the root zone $\mathrm{SAR_{iw}}$ = 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 10. Once a LF is determined, Equation

22 can be used in conjunction with Equation 20 to estimate a new soil ESP and this new ESP can then be used in the methods outlined in Table 10.

Assessing soil structural stability using SAR and EC of irrigation water

In most cases in Australia, rainfall events in excess of the soil water deficit can leach some of the accumulated salts below the root zone. Hence, 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.

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 Figure 7).

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 3.2 dS/m and an SAR of 13 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.

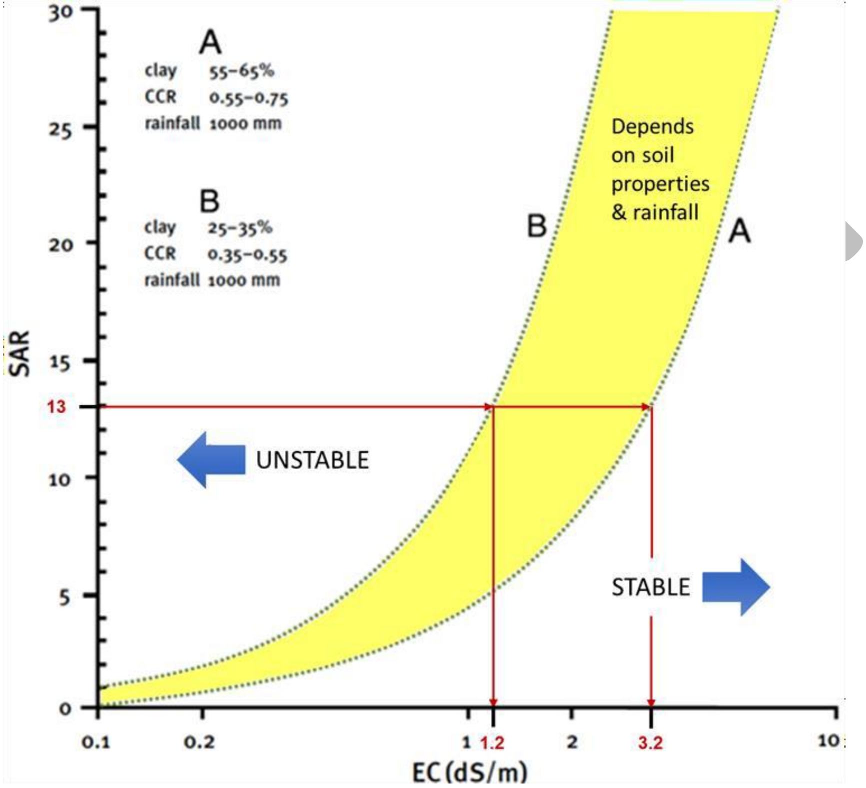


Figure 7 Relationship between SAR and EC of irrigation water for prediction of soil structural stability

Adapted from DNR 1997

However, the relationship presented in Figure 7 cannot be representative of the range of soils encountered throughout Australia (Bennett and Raine 2012). It is useful as a general guide to understand areas of threshold electrolyte concentrations beyond which dispersion will occur but should not be considered as an absolute guide for irrigation. Bennett and Raine (2012) showed that threshold electrolyte concentration (TEC) beyond which aggregate-dispersion will occur leading to a reduction in K_{sat}, was clearly soil specific, even within a soil order of the Australian Soil Classification. In their study defining TEC as that which causes a 20% reduction of K_{sat} in 58 Australian soils, the authors showed that the soil specific TEC was best predicted by clay content and clay net negative charge. The soil specificity of the TEC curve and comparison with the generic comparison between SAR and EC is illustrated in Figure 8.

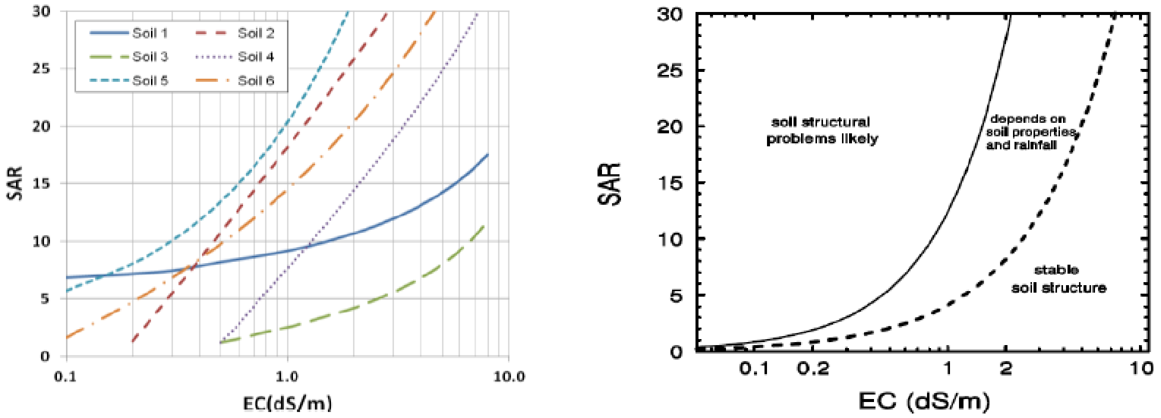


Figure 8 Comparison of the TEC curves (leading to a 20% reduction in K_{sat}) for six soils (left) and the relationship between SAR and EC for soil structural stability as presented in ANZECC and ARMCANZ (2000).

Source: Bennett and Raine (2012)

3.2.3 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 of specific ions, particularly sodium, chloride, boron and other metal ions);
- nutritional disorders (due to excessive concentrations of some ions).
- Plants respond to salinity in the whole root zone. Two measures of average root zone salinity are commonly used: arithmetic average and water uptake weighted average (Figure 9).

  Indeed, soil salinity values usually increase from the soil surface peaking at the bottom of the root zone. Because the majority of root water extraction is taken from the upper part of the root zone, this lower soil salinity has more effect on plant growth than the EC at the bottom of the root zone. Hence there is a need for some depth-weighted averaging applied to the above calculations (Equation 5). This could either be a simple arithmetic mean or a root water uptake weighted mean. The following graphs show the effect of these averaging assumptions (Figure 9)
- 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 (Bernstein and Francois 1973; Devitt et al. 1984; Rhoades 1982) have shown average root zone salinity to provide an appropriate measure of root zone salinity for determining plant response to salinity.

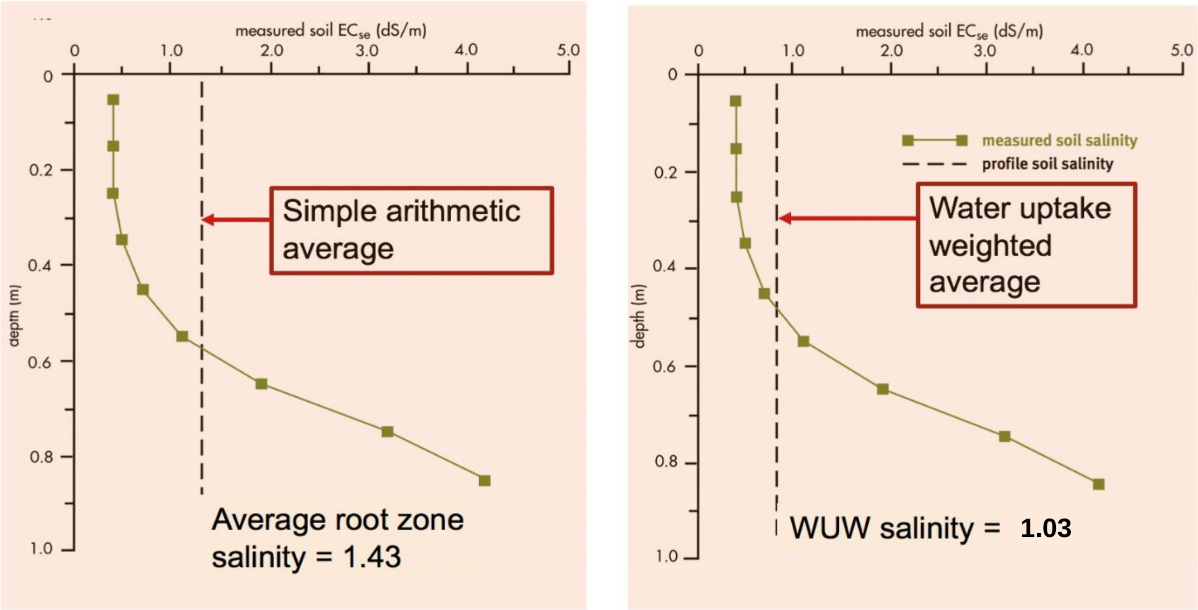


Figure 9 Two measures of root zone salinity (RZS): average RZS (left) and water uptake weighted RZS (right)

Adapted from DERM (2011)

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 should 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.

Salinity usually increases with depth because the amount of water passing below any given soil depth (i.e., the leaching fraction) increases with depth until the bottom of the root zone is reached. At this point the LF remains constant with increasing depth. As the upper part of the root zone has the lower salinity and contributes more of the root water uptake this layer is more important than more saline deeper soil layers. Hence the idea of a water uptake weighted average.

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.

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 response are sodium, chloride and boron (see Sections 4.2, 4.3, 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 usual 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 Figure 10). most of the data was obtained using constant flow or daily watering saline solutions to keep the soil solution EC more or less constant, which is not likely to happen in field conditions.

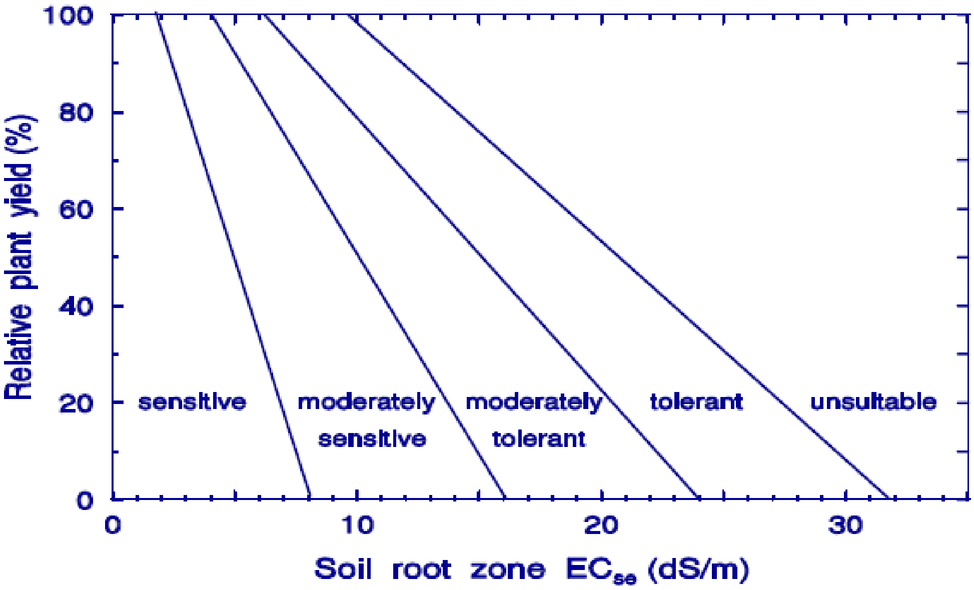


Figure 10 Relative crop yield in relation to soil salinity (EC_{se}) for plant salt tolerance groupings of Maas and Hoffman (1977)

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 for a particular irrigation situation can be assessed. This interrelationship is shown in Figure 11. This figure (modified from Rhoades 1982) illustrates that the leaching fraction has a profound influence on the root zone salinity, the type of plants that can be grown. In this figure, the EC_{se} -EC combinations to left of any LF line specify acceptable root zone salinities for the different plant types. The EC_{se} range alone (y-axis) specifies the acceptable soil salinity for each type of plant category, eg 5 to 8 dS/m for tolerant plants. Then within any given EC_{se} range, the higher the LF the higher the acceptable irrigation water salinity. This means that very saline water (eg EC>6dS/m) can be used on tolerant plants only if the LF is 0.5 or more.

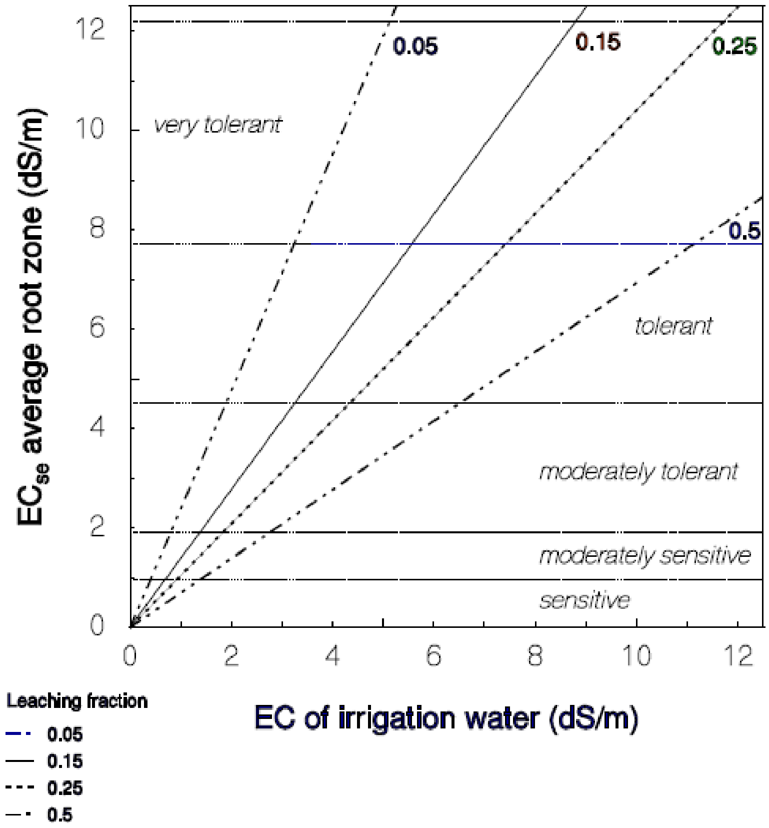


Figure 11 Interrelationships between irrigation water salinity, root zone salinity, leaching fraction and plant salt tolerance

Modified from Rhoades 1982

Table 14 is a compilation of plant salt tolerance data, including threshold salinity values and rate of yield decline with increasing salinity. Table 15 includes salinity threshold values as well as EC$_i$ at different leaching fractions. Leaching Fraction and EC$_{se}$ define the acceptable irrigation salinity for any irrigation depth (in mm/yr) and climate. The data are correct for uniformly salinised soils in which the dominant anion is chloride. These tables provide a guideline of plant suitability based on **average** root zone salinity (EC$_{se}$), which was discussed previously. 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:

$$\mathrm{Y_r} = 100 - \mathrm{B}(\mathrm{EC_{se}} - \mathrm{A})$$ **Equation 23**

where:

Y_r = relative yield (in %)

A = the salinity threshold (in dS/m) and

B = the percentage productivity decrease per dS/m increase above the threshold value (from Table 14 and Table 15).

EC$_{se}$ values are also listed in Table 14.

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

$$\mathrm{EC}_{\mathrm{se}90\%} = \mathrm{A} + \frac{10}{\mathrm{B}}$$ **Equation 24**

It can also be applied to 75 percent or 50 percent yield values as shown in Table 14.

Equation 23 is derived from Maas and Hoffman (1977). However, as pointed out by Grattan (unpublished), there is uncertainty about the salinity threshold (A) in that it lacks physiological justification and because the standard errors associated with the threshold values is large (Grieve et al. 2012). Also, the preliminary steps for the derivation of Equation 23 rely on obtaining a yield unaffected by salinity, which in itself is difficult to obtain (Van Straten et al. 2019). The authors suggest using the 90% yield EC_{se} parameter to overcome the lack of robustness of the yield threshold parameter.

A non-linear response-curve have since been developed (van Genuchten and Gupta 1993). It is a preferable model mathematically (Van Straten et al. 2019) even though the Maas and Hoffman model remains used because of the ease of interpretation. It is recommended for use in these Guidelines.

Finally, Tedeschi (2020) points out that the Maas and Hoffman relationship was developed under saturated soil conditions and a constant EC, which is rarely met in the field or, if so, then only for a limited amount of time. Therefore, it should be corrected, taking into account the actual water content of the soil (Tedeschi et al. 2017).

3.2.4 Factors affecting the expression of salinity

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

Table 14 Salinity threshold for plants, expressed in units of saturation extract (EC_{se}), arranged in alphabetical order by common name, within broad plant groups[a]

Note: ECse values are reported for plant responses when yield is first affected (the salinity threshold) as well as those values corresponding to 90%, 75% and 50 % of the maximum, unconstrained yield. Also listed is the percentage plant yield reduction per unit increase in ECse, for salinity values > threshold ECse.

Common name	Scientific name	Salinity threshold (EC_{se} dS/m)	Productivity decrease per dS/m increase (%)	Soil Salinity EC_{se} (dS/m) at			Reference[b]
				90% yield	75% yield	50% yield	
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*	1.9	9.1	3.8	5.1	7.1	10
Safflower	*Carthamus tinctorius*	6.5					6
Sorghum	*Sorghum bicolor*	6.8	15.9	7.4	8.4	9.9	4
						2.8-3.06	11
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
Plant crop			7.5				18
Ratoon cane			11.8-16.9				18
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	18.0-19.0	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
- Own roots: Sultana vines, Shiraz, Chardonnay		1.8	9-15 (sultana on own roots				12, 13, 14, 15, 16, 17
- Rootstocks		1.8-3.3	6 (Sultana on rootstock)				
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
Pistachio		9.4	8.4				19
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
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
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
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
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 DERM (2011), b References: 1 Maas and Hoffman (1977); 2 Ayers and Westcot (1976); 3 Russell (1976); 4 Maas (1986); 5 West and Francois (1982); 6 Bresler et al. (1982); 7 Ayers (1977); Heuer et al. (1986); 9 Shaw et al. (1987); 10 Grattan et al. (2002); 11 Daniells et al. (2001); 12 Biswas et al. (2009); 13 Walker et al. (2002); 14 Zhang et al. (2002); 15 Walker et al. (2004); 16 Walker (2010); 17 Tregeagle et al. (2006); 18 Kingston and Anink (2005); 19 Sanden et al (2018)

Table 15 Maximum irrigation water salinity (EC_i) values that constrain the subsequent equilibrium soil salinity to the threshold value, for different Leaching Fractions

Note: The smaller the LF, the lower the acceptable EC_i.

Common name	Scientific name	EC_i (dS/m) at					Reference
		Salinity threshold (EC_{se} dS/m)	25% LF (e.g., sand)	20% LF (e.g., sandy loam)	17% LF (e.g., loam)	12% LF (e.g., light clay)	
Grains							
Canola (oilseed rape)	*Brassica napus*	2-4	3.0	2.6	2.2	1.7	b
Faba bean	*Vicia faba*	2-4	3.0	2.6	2.2	1.7	b
Narrow-leaf lupin	*Lupinus angustifolium*	2-4	3.0	2.6	2.2	1.7	b
Pasture							
Balansa clover	*Trifolium michelianum*	4-8	6.0	5.2	4.4	3.4	b
Bermuda grass	*Cynodon dactylon*	6.9	6.9	5.9	5.1	3.9	a
Brome. Meadow		4.4	4.4	3.8	3.2	2.5	c
Brome,smooth	*Bromus inermis*	1.5-3	2.3	1.9	1.7	1.3	a
Burr Medic	*Medicago polymorpha* ssp *brevispina*	2-4	3.0	2.6	2.2	1.7	b
Cocksfoot	*Dactylis glomerata*	<2	1.5	1.3	1.1	0.8	a
Murex medic	*Medicago murex*	<2	1.5	1.3	1.1	0.8	b
Perennial ryegrass	*Lolium perenne*	5.6	5.6	4.8	4.1	3.1	a
Persian clover	*Trifolium resupinatum*	4-8	6.0	5.2	4.4	3.4	b
Rye (forage)	*Secale cereale*	2.5	2.5	2.2	1.8	1.4	c
Serradella	*Ornithopus* spp	<2	1.5	1.3	1.1	0.8	b
Sorghum	*Sorghum bicolor*	4-8	6.0	5.2	4.4	3.4	b
Subterraneum clover	*Trifolium subteranneum*	<2	1.5	1.3	1.1	0.8	b
Sweet clover	*Melilotus officinalis*	4	4.0	3.4	2.9	2.2	c
Tall fescue	*Festuca arundinacea*	3.9-8	6.0	5.1	4.4	3.3	a, b
Tall wheat grass	*Thinopyrum elongatum*	4-8	6.0	5.2	4.4	3.4	b
Timothy	*Phleum pratense*	2	2.0	1.7	1.5	1.1	c
Triticale (forage)		6.1	6.1	5.3	4.5	3.4	c
Vetch (common or spring)	*Vicia angustifolia*	3	3.0	2.6	2.2	1.7	a,c
Wheatgrass, standard crested	*Agropyron sibiricum*	3.5	3.5	3.0	2.6	2.0	a
Yellow serradella	*Ornithopus compressus*	<2	1.5	1.3	1.1	0.8	b
Fruits							
Pumpkin	*Cucurbita pepo pepo*	1.5-3	2.3	1.9	1.7	1.3	a
Vegetables							

Asparagus	*Asparagus officinalis*	4.1	4.1	3.5	3.0	2.3	a
Radish	*Raphanus sativus*	1.2	1.2	1.0	0.9	0.7	a
Rosemary	*Rosmarinus officinalis*	6-8	7.0	6.0	5.1	3.9	a
Turnip	*Brassica rapa*	0.9	0.9	0.8	0.7	0.5	a

Sources:
a = Maas (1986)
b = Agriculture WA (2003)
c = Kotuby-Amacher et al. (1987)

Adapted from AGWR (NRMMC, EPHC and AHMC, 2006)

Climate
The climate of a region contributes to salinity in the soil under irrigation. The two key processes of influence are evapotranspiration and rainfall within the water balance as described in Equation 25

$$\text{Rainfall} + \text{Irrigation} = \text{Transpiration} + \text{Evaporation} + \text{Deep drainage} + \text{Runoff} + \text{Soil water storage} \qquad \textbf{Equation 25}$$

Deep drainage (D_d) via the leaching fraction is the driver for salinity control in irrigation paddocks as well as influencing the outbreak of secondary salinization in dryland landscapes after land use change. Under dryland conditions the quantity of D_d depends on the balance between the soil water deficit preceding rainfall (affected by crop type eg trees vs grasses), the amount of net rainfall (rainfall-runoff) and the seasonality of the rainfall. For example, in Mediterranean climates of southern Australia, hot dry summers promote large soil water deficits which are then followed by cold, wet winters. Infiltration in excess of the soil water deficit promotes D_d provided the soil permeability allows much of the rainfall to be routed into infiltration. In northern Australia rainfall is summer dominant and often comes in big rainfall events. Hence even though evapotranspiration is higher during summer some of the storm events are sufficient to exceed water holding capacity of the soil (modified by the permeability of the soil profile) that translates into D_d.

Such variability in is illustrated in Figure 12 and Figure 13 where the strong seasonal effects on potential irrigation demand can be seen for Cairns (with a low summer demand) and Bendigo (low winter demand).

Seasonal patterns of rainfall have been shown to make a difference to natural soil salt levels (Shaw 1996). 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. Moreover 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.

Milly (1994) observed 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 high rainfall summer environment where rainfall is not limiting would be expected to give a similar deep drainage response as a winter rainfall environment where evapotranspiration is limiting. Different situations will occur at intermediate annual rainfalls and are best explored using daily water balance models such as HowLeaky, APSIM or MEDLI.

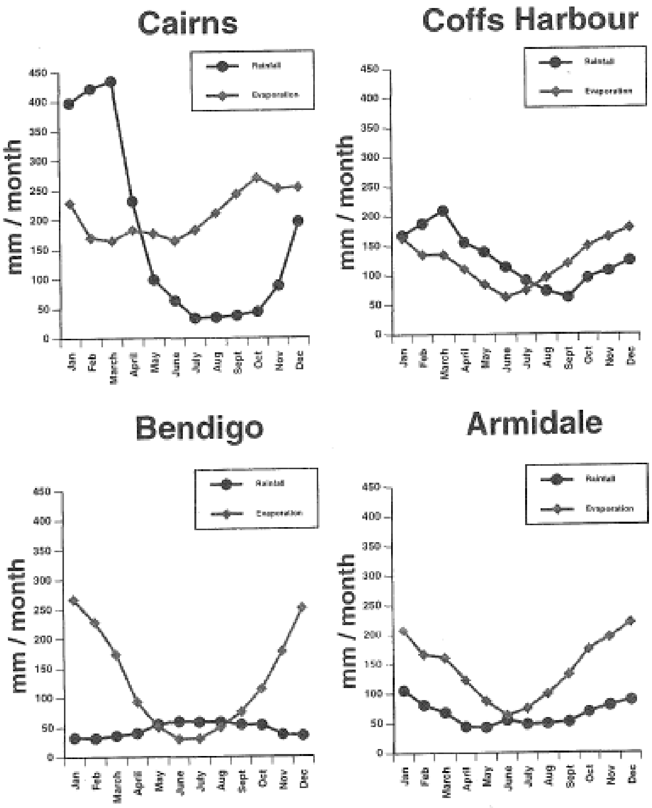


Figure 12 Measured long term average monthly rainfall and Class A Pan evaporation for a range of climatic zones in Australia

Source: Gardner et al. (1995)

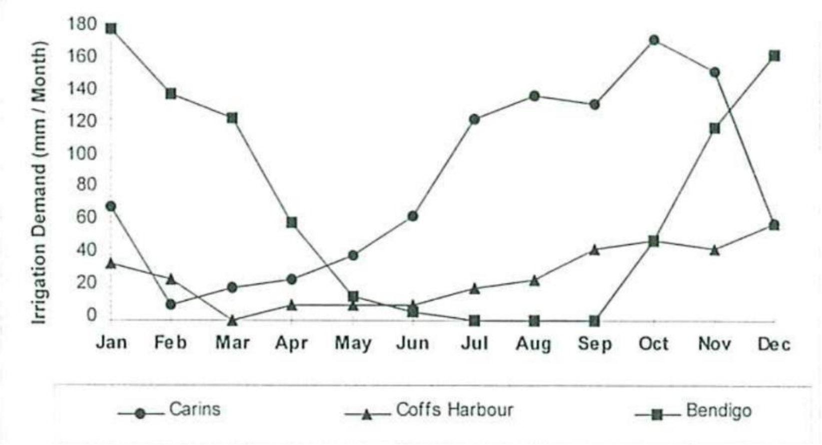


Figure 13 Monthly distribution of the potential irrigation demand for a pasture (mm/month) at three locations in Australia

Source: Gardner et al. (1995)

It must be noted that the empirical deep drainage predictions presented in this section are based on Queensland climate and soil data and may not be predictive of the different soils and climates of southern Australia. More research is needed to validate and/or adapt these equations to a wider of soil/climate conditions across Australia.

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 allows the watertable to rise to near the soil surface, thereby creating a discharge area with evaporative concentration of salts. Hydrologically sensitive landscapes often show evidence of past seepages or shallow watertables. If land use change occurs in these types of landscapes, thereby changing the hydrological balance, dryland salinity 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 flow is restricted downslope of these barriers.
- Heavy soils or soil compaction at the base of slopes or clays deposited at the confluence of streams slow the movement of water through the soil, so that the groundwater flow is restricted 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 internal 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 land use change a new and wetter hydrologic equilibrium is established, and rising watertables can move salt from these historic salt stores closer to the soil surface. Human activities can disturb the hydrologic balance by increasing water inputs to the catchment, or by introducing barriers to groundwater 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 lag time (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), the buffering storage of the regolith, the outflow capacities of the catchment and the length of the catchment. Finely balanced catchment systems with low storage and low subsurface outflow capacities will experience salting in perhaps a few years compared with a decade in systems with greater capacities. When deep rooted perennial native vegetation is replaced by shallow rooted, seasonal growing agriculture crops, the frequency and value of the maximum soil water deficit preceding rainfall is reduced, thereby promoting the opportunity for deep drainage from similar amounts of seasonal infiltration. This happens with irrigation and irrigation induced dep drainage contributes to watertable rise. During these periods, extra water moves below the root zone to the groundwater, increasing the likelihood of watertable rise.

3.2.5 Irrigation management for salinity control

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 about 2 m or preferably below the root zone. Methods to prevent/control shallow watertables include:

- identify restrictions to groundwater flow 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;
- ensure flood irrigation layout and water application rates are optimised to achieve high irrigation distribution efficiency and reduce deep drainage;
- lining of irrigation channels and/or use of pipes for on-farm distribution to minimise seepage from channels;
- incorporating sub surface drainage infrastructure where it is both economically and environmentally sustainable.
- Appropriate irrigation scheduling to prevent over irrigation

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 soil husbandry practices, including soil amendments. These are described in detail by Ayers and Westcot (1976) and Salinity management handbook - Datasets | Publications | Queensland Government.

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

3.3.1 Scenario 1

A farmer has been irrigating for 10 years from a local bore and is concerned soil salinity build up may be affecting the yield of his crop. He is also interested in changing to more salt tolerant crops that are more suited for his water quality and soil type.

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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

irrigation applied = D_i = 600 mm /yr

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

Annual rainfall = D_r = 650 mm /yr

EC of rainfall = EC_r = 0.03 dS/m

Convert $EC_{1:5}$ to EC_s

$$\mathrm{EC_s} = 2.2 \times \mathrm{EC_{1:5}} \left[\frac{(500 + 6\mathrm{ADMC})}{\mathrm{SP}}\right]$$

Derived from Equation 15 and Equation 18

$$\mathrm{EC_s} = 2.2 \times 0.9 \times 8.83 = 17.7\ \mathrm{dS/m}$$

Calculate weighted EC of input water EC_i

$$\mathrm{EC_i} = \frac{[(\mathrm{D_{iw}} \times \mathrm{EC_{iw}}) + (\mathrm{D_r} \times \mathrm{EC_r})]}{\mathrm{D_i}}$$

From Equation 5

$$\mathrm{EC_i} = \frac{[(600 \times 1.7) + (650 \times 0.03)]}{1250} = 0.83$$

Calculate leaching fraction (bottom of root zone)

$$\mathrm{LF} = \frac{\mathrm{EC_i}}{\mathrm{EC_s}} = 0.047 \sim 5\%$$

From Equation 10

Calculate average root zone leaching fraction

$$\mathrm{LF_{av}} = (0.976 \times \mathrm{LF} + 0.022)^{0.625} = 0.19 = 19\%$$

From Equation 13

Predicted average root zone salinity as Saturation Extract

$$\mathrm{EC_{se}} = \frac{\mathrm{EC_i}}{(2.2 \times \mathrm{LF_{av}})} = 2.0\ \mathrm{dS/m}$$

From Equation 12

The predicted EC_{se} can then be compared with plant salt tolerance data provided in Table 14 or Table 15 to estimate the severity of potential yield reduction of the crop.

3.3.2 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

Average CCR= 0.81

ESP at 0.9 m = 4.5%

Water

SAR = 5

EC_i = 1.7 dS/m

Salinity of Rainfall= ECr =0.03 dS/m

Depth irrigation = D_i = 700 mm

Depth rainfall = D_r = 550 mm

Total depth = $D_t = D_i + D_r$

Calculate the Leaching Fraction under rainfall using:

$$\mathrm{LFr} = \frac{\mathrm{ECr}}{2.2 \times 10^{\left[\mathrm{a+blog}\left(\frac{0.03\times \mathrm{rainfall}}{\mathrm{ESP}}\right)\right]}}$$

From Equation 7

where:

a = 0.794

b = –1.105

noting the coefficients a and b are obtained from Table 11.

Therefore:

$$\mathrm{LFr} = \frac{0.03}{2.2 \times 10^{\left[0.794-1.105\log\left(\frac{0.03\times 550}{4.5}\right)\right]}} = 0.009 = 1\%$$

Calculate Leaching Fraction under new irrigation water quality:

$$\mathrm{LF_f} = \mathrm{Lf_i}\left[2.65\left(\frac{\mathrm{EC_i}}{\mathrm{Ec_r}}\right)^{0.5} - 1.35\right]$$

From Equation 9

This relationship accounts for an increase in LF due to increased solute concentration of the soil solution from the added salt in the irrigation water

$$\mathrm{LF_f} = 0.0948 = 9.5\%$$

Calculate average root zone Leaching Fraction

$$\mathrm{LF_{av}}(\%) = (0.976 \times \mathrm{LF_f} \times 0.282)^{0.625} = 0.26$$
$$= 26\%$$

From Equation 13

Predicted average root zone salinity:

$$\mathrm{EC_{se}} = \frac{\mathrm{EC_i}}{(2.2 \times \mathrm{LF_{av}})} = 3\ \mathrm{dS/m} \qquad \text{From Equation 12}$$

The predicted $\mathrm{EC_{se}}$ can then be compared with plant salt tolerance data provided in Table 14 and Table 15 to determine likely crop response to this irrigation regime.

3.3.3 International guidelines

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. However, guidelines developed on water composition alone are still being used, especially in the field of recycled water use. Usually, three categories for EC and Total Dissolved Salts (TDS) thresholds are being presented in Guidelines and regulations based on wastewater use restriction as none, slight to moderate and severe with respect to salinity. The summary of EC and TDS thresholds are presented in Table 16.

Table 16 Salinity (electric conductivity (EC) thresholds in agricultural water reuse regulations and guidelines

Regulation/Guideline	EC (dS/m)	Reference
FAO • Saskatchewan • US-EPA	None: <0.7 Slight to moderate :0.7-3.0 Severe: >3	Ayers and Westcot (1985) Water Security Agency treated municipal wastewater irrigation guideline (2014) USEPA (2012)
Alberta and Atlantic Canada	Unrestricted: <1.0 Restricted: 1.0-2.5 Unacceptable: >2.5	Alberta Environment (2000) Atlantic Canada Wastewater Guidelines Manual (Environment Canada 2006)
Italy	3	Barbagallo et al. (2001)
Israel	1.4	Inbar (2007)
AGWR	0.2-2.9	NRMMC, EPHC and AHMC (2006)
Portugal	<1	Marecos do Monte (2007)

4 Major ions of concern for irrigation water quality

4.1 Bicarbonate

No guideline value is recommended for bicarbonate in irrigation waters.

The FAO (Ayers and Westcot 1985) recommends a maximum concentration of 91.5 mg/L of bicarbonates without restrictions in use of irrigated water. Iran recommends 90mg/L (Ministry of energy Iran 2010) and Jordan up to 400mg/L (Government of Jordan 2006). Bicarbonates guideline values in irrigated waters are not mentioned in other regulations internationally.

4.1.1 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:

$CO_2 + CaCO_3 + H_2O \leftrightarrow Ca^{2+} + 2\ HCO_3^-$

4.1.2 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 and permeability (discussed in Section 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 deposited 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. The process might also be due to the chlorophyll formation in leaves being impacted by bicarbonate. In an experiment using various levels of ammonium bicarbonate in irrigation water, Shahabi et al. (2005) showed that high levels of bicarbonate (30 meq/L and above) caused a significant reduction in chlorophyll development in apple leaves after 40 days of irrigation. In the longer-term, after 120d of irrigation, chlorophyll development was affected in all treatments (except at 5 meq/L of bicarbonate). Similarly, certain cultivars of pomegranates showed a reduced chlorophyll index, performance index and translocation of Fe^+ and Na^+ (from roots to shoots) as the concentrations of $NaHCO_3$ in the irrigation water increased (up to 15mM). The hypothesis is that decrease of the chlorophyll index is due to a Fe deficiency caused by an excess of $NaHCO_3$, leading to foliar chlorosis (Karimi and Eini-Tari 2017). In other studies, photosynthetic parameters decreased in chlorotic leaves induced by high bicarbonate concentrations in citrus (Pestana et al. 2005), peach rootstocks (Molassiotis et al. 2006)

and peas (Jelali et al. 2011). 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 (Phosphorus (Jones 2001); Iron (Karimi and Eini-Tari 2017; Deng et al. 2010).

4.2 Chloride

There are three distinct issues concerning chloride concentrations in irrigation waters: relating to the risk of (1) foliar injury to crops; (2) salty taste in the crop/fruit and (3) increased uptake by plants of cadmium from soil.

4.2.1 Foliar injury

> ***Guideline values for prevention of foliar injury due to chloride in irrigation water from sprinkler application are provided in Table 17.***

The chlorides of sodium, potassium, calcium and magnesium are highly soluble in water. Chloride is an essential nutrient for plants and plays many regulatory roles in their physiology, being involved in photosynthesis, transpiration and plant growth. However, it can accumulate to toxic levels in leaves, leading to leaf tip burn (Niu et al. 2008) or leaf abscission and reduction of yield or aesthetic value in the case or ornamental plants. Chloride behaves similarly to sodium with similar foliar symptoms. High levels of chloride in the soil solution will lead to yield decline due to an osmotic and turgor effect, hence threshold values for salinity (expressed as EC) should be used as a guide to water quality (Section 3).
Plant sensitivity to chloride is presented in Table 17.

Table 17 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	Sugarbeet
Plum		Lucerne	Sunflower
Grape		Safflower	
		Sorghum	

a After Maas (1990)

4.2.2 Salty taste

Chloride accumulation in the plant can also result in a salty taste, which affects crop markets negatively. Chloride accumulation in berries in the case of grape juice or wine, is detrimental, conferring the product a salty taste and in some cases, leading to its impossibility to sell (Coli et al. 2015; Leske et al. 1997). Studies investigating the sensory impacts from chloride in wine indicated a threshold for white and red wine juices and wine product of 133 mg/L of chloride (de Loryn et al. 2014).

Australia is recognised for a high concentration of chloride in wines, which is influenced by the variety, root stock, scion, soils, irrigation water and methodology (Walker et al. 2010; Stevens and Pitt 2011; Coli et al. 2015). A recent study has benchmarked soil sodium and chloride concentrations and indicated that some wine districts in Australia need to manage these hazards (Edwards 2014).

4.2.3 Interaction between chloride in irrigation water and cadmium in soil

Guideline values for assessing chloride levels in irrigation water with respect to increased cadmium uptake by crops are provided in Table 18.

Table 18 Risks of increasing cadmium concentrations in crops due to chloride in irrigation waters[a]

Irrigation water chloride (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 and Kroontje 1973):

$$\mathrm{Cd}^{2+} + \mathrm{Cl}^- \rightarrow \mathrm{CdCl}^+ \qquad \mathrm{Log\ K} = 2.0$$

$$\mathrm{CdCl}^+ + \mathrm{Cl}^- \rightarrow \mathrm{CdCl}_2^0 \qquad \mathrm{Log\ K} = 2.6$$

$$\mathrm{CdCl}_2^0 + \mathrm{Cl}^- \rightarrow \mathrm{CdCl}_3^- \qquad \mathrm{Log\ K} = 2.4$$

$$\mathrm{CdCl}_3^- + \mathrm{Cl}^- \rightarrow \mathrm{CdCl}_4^{2-} \qquad \mathrm{Log\ K} = 2.5$$

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 and McLaughlin 1996), cadmium concentrations in crops are increased.

In brown mustard (Brassica juncea), high Cl^- in the soil solution (up to 200mM NaCl) caused an enhanced accumulation of cadmium (Lopez-Chuken et al. 2010). Similar results were also observed for wheat (Dahlin et al. 2016), maize or tobacco (Lopez-chucken 2012).

It has been clearly demonstrated that for commercial crops, the presence of high concentrations of Cl^- in the soil solution significantly increases crop Cd concentrations (McLaughlin et al. 1994; Weggler-Beaton et al. 2000; Weggler et al. 2004; Ozkutlu et al. 2007; Ishikawa et al. 2015). 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).

4.2.4 Guideline values in other regulations

In some regulations internationally, chloride concentrations are provided as a single value and do not take into account crop sensitivity: Amongst the most restrictive are Saudi Arabia and Iran (100mg/L). Italy and Iran both recommend irrigating with a maximum Cl concentration in water of 250mg/L, China (Ministry of Agriculture 2007) recommends 350mg/L, the AGWR (NRMMC, EPHC and AHMC 2006) recommend 340mg/L. Italy specifies a Cl threshold of 250 mg/L in irrigation water (Barbagallo et al. 2001) Other regulations are based on the restriction on use (FAO (Ayers and Westcot 1985); see Table 19; Oman (Oman's Ministerial Decision No 145 of 1993)).

Table 19 Chloride concentration thresholds in agricultural water reuse as recommended by the FAO

Chloride (Cl^-)		Degree of restriction on irrigation	
		none	Restricted use
Surface irrigation	mg/L	<142	142-355
Sprinkler irrigation	mg/L	<142	>142

Note: A "restriction on use" indicates that there may be a limitation in choice of crop, or special management may be needed to maintain full production capability. A "restriction on use" does not indicate that the water is unsuitable for use

4.3 Sodium

Guideline values for prevention of foliar injury due to sodium in irrigation water following sprinkler application are provided in Table 20. Values for specific toxicity effects are provided in Table 21.

Sodium, like chloride and boron, is of great importance for plant health, either directly by being part of the metabolic processes, or indirectly by interfering with other nutrients uptake. In regulations internationally, recommended Sodium concentration in irrigation water varies from as low as 69 mg/L (unrestricted use (Ayers and Westcot 1985), to as high as 312 mg/L in the AGWR (NRMMC, EPHC and AHMC, 2006). In South Africa (DWAF 1996), the regulation mentions a target range (up to 70), a maximum range (mostly applicable to fine textured neutral to alkaline soils) of 70 to 460 depending on crop sensitivity, and a short-term use on site-specific basis with Na^+ values higher than 460mg/L.

In Oman (Oman's Ministerial decision No.145 of 1993), sodium threshold value for water use for irrigation is set at 200mg/L for food crops and 300mg/L for non-food crops.

Table 20 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	Sugarbeet
Plum		Lucerne	Sunflower
Grape		Safflower	
		Sesame	
		Sorghum	

a After Maas (1990)

Table 21 **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 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 3); and (3) calcium and magnesium deficiency through reduced availability and imbalance with respect to sodium.

5 Metals and metalloids

5.1 Scope

The irrigation water quality guidelines assessed the following criteria for each 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 (crop or 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 metals and metalloids' toxicity to the soil biota (micro and macro flora and fauna) is an issue receiving international attention, ecotoxicity is generally observed at lower soil concentrations than phytotoxicity (Efroymson et al. 1997a, 1997b). While the guidelines have considered the potential environmental impacts of inorganic contaminants in irrigation water on soil biota, insufficient information was available in 2000 to set water quality guideline values based specifically on ecotoxicity to soil biota and any additional data has not been reviewed for the 2020 update.

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 Chapters of the Water Quality Guidelines, guideline values are different. For instance, the guideline value for cadmium in aquatic systems ranges from 0.06 to 0.2 µg/L (99th and 95th percentile of protection) (ANZG 2021), whereas for irrigation water it ranges from 10 to 50 µg/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.

5.2 Methodology for development of guideline values

5.2.1 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 sewage (recycled water) from domestic and industrial sewer catchments (if recycled water is used, the Australia Guideline for Water Recycling should also be referred to) (NRMMC et al. 2006). 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, from natural geochemical enrichment for instance.

5.2.2 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 plant growing similar methods).

5.2.3 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 may lead to the build-up of inorganic contaminants in the soil surface layer to unacceptable levels for the land use or future land use. There is the potential for contaminants to reach concentrations in soil that are toxic to crops or cause a reduction in crop quality via plant root uptake.

5.2.4 Calculation of irrigation loading rates and time periods

In order to develop the default guidelines values (DGV) the following set of assumptions were used to calculate the contaminant loading rates resulting from irrigation:

- annual application of irrigation water is 10 ML/ha/yr (i.e. 1000 mm/yr);
- inorganic contaminants are retained in the top 0.15 m 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 typical basis for developing irrigation water quality guidelines and similar logic has been used in the development of similar guidelines published by Canada , Food and Agriculture Organisation of the United Nations, United States and the World Health Organisation (Ayers and Westcot 1985; Pescod 1992; CCME 2006; WHO 2006; US EPA 2012).

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, and the 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 manifest than alkaline soils. Thus, a single guideline value must be treated with caution, as effects of contaminants on plants and organisms are therefore soil (condition) specific.

The potential for contamination of groundwater is also an issue that is highly soil 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 default guideline values for metal contaminants in irrigation water may then be revised if groundwater contamination is considered a potential risk.

The guideline 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 guideline 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; WHO 2006; US EPA 2012; CCME 2018). Two DGVs 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:

- **DGV or Default Guideline Value**. The DGV is the maximum concentration (mg/L) of contaminant in the irrigation water which can be tolerated to protect the receptors identified in the scope of the guidelines (Section 1.2) assuming 100 years of irrigation, based on the irrigation loading assumptions previously mentioned and should be used in conjunction with the ACL (see underneath for the definition of the term) as a trigger for further investigation.
- **SGV or short-term guideline value**. The SGV 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 DGV or considering site specific related

receptors and exposure. As short-term use is typically site-specific, where available the SGV should also be used in conjunction with the ACL as a trigger for further investigation.

The DGV and SGV 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 crops or impacts on crop quality. Where SGV and DGV 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:

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

The ACL is calculated based on background concentrations of contaminants in Australian agricultural soils, mixing of the contaminant in the top 0.15 m of soil, soil bulk density of 1300 kg/m^3, and Australian guideline values for contaminant concentrations in agricultural soils treated with sewage biosolids (EPA NSW 1997; DEHP Qld 2016). Once the ACL 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 ACLs may be overly protective in some situations and less protective in others. The ACL 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 ACL adjusted accordingly.

The ACL for contaminants has been calculated as follows:

$$\mathrm{ACL} = \frac{(\mathrm{MAC} - \mathrm{BC}) \times \mathrm{Depth} \times \mathrm{BD}}{100} \;\; (\mathrm{kg/ha})) \qquad \textbf{Equation 26}$$

where:

MAC = maximum allowable soil concentration of a contaminant (mg/kg)
BC = assumed background concentration (mg/kg) (Table 22)
Depth = soil depth (0.15 m)
BD = soil bulk density (kg/m^3)

Example calculation of ACL 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 Efroymson et al. 1997a; Efroymson et al. 1997b; and Scott-Fordsmand and Pedersen 1995)). Recommended maximum zinc concentrations in soil, above which adverse effects on either plants or microorganisms are likely, vary from 100–200 mg/kg (Efroymson et al. 1997a; Efroymson et al. 1997b; Scott-Fordsmand and Pedersen 1995). In Australia, maximum allowable concentrations (MACs) for zinc in agricultural soils receiving sewage biosolids is typically 200 mg in Australia and 300 mg/kg in New Zealand (Table 21). 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 22. The median background concentration of zinc in New Zealand soil (Median of median's supplied) was estimated to be 33.1 mg/kg, excluding the rock-group scoria, which was naturally high in Zinc (Table 22).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 22), to derive the ACL. Median values for contaminants were used to derive the background concentrations used for calculating ACL in these guidelines due to the log normal distribution of many of the datasets.

Zinc ACL is calculated as follows:

$$\mathrm{ACL} = \frac{(200 - 39) \times 0.15 \times 1300}{100} = 300\ \mathrm{kg/ha}$$

Equation 27

Using only an analysis of the irrigation water and the irrigation water application frequency and amounts, the user can calculate the ACL (assuming soils have no history of metal contamination). Once the ACL has been reached for a particular site, it does not necessarily follow that irrigation must cease. When the ACL 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 23, the DGV and SGV for zinc are 2 mg/L and 5 mg/L, respectively. The DGV and SGV would allow zinc concentrations in soil to reach over 1000 and 500 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 ACL for zinc is 300 mg/kg, equating to approximately 154 mg/kg applied to the top 0.15 m of the soil. The ACL has been introduced to avoid this undesirable situation, but still allow for the 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 DGV, SGV and soil ACL guidelines for a range of metals and metalloids are summarised in Table 23. The DGV should be used as the first screen as to determine a potential long-term impact if the DGV is exceeded, and superseded by good scientific assessment for use of the SGV recommend or an SGV determined for a specific site, where there is sufficient data to ensure a high level confidence in a derived SGV.

Table 22 Datasets used to derive suggested upper background values for uncontaminated Australian and New Zealand soils

	Olszowy et al. (1995) Number of samples = 120			Barry (1997) Number of samples = 120			CSIRO (unpublished) Number of samples = 350			Tiller (1983) Number of samples not stated		Spouncer and Mowat (1991a-d) Number of samples = 209			Cavanagh et al. (2015) Based on rock-groups	
	Depth = 0-150mm			Depth = 0-100mm			Depth = 0-150mm			Depth not stated		Depth = 0-100mm			Depth = 0 - 300mm	
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	Median mg/kg	95th percentile mg/kg
Al																
As	5-53	7	5	1–20	3	3									0.55 - 8.07	0.95 - 33.77
Be																
B												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					0.016 - 0.27	0.08 - 2.05
Cr	5-56	8	5	<9-573	132	65	2.5-673		115	41					5.66 - 54.17	0.95 - 262.1
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		4.56 - 30.19	0.95 - 129.4
Fe																
Pb	5-56	14	14	5-81	27	24	2-60.5		14	12					5.69 - 154.62	0.95 - 562.1
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					1.9 - 27.38	0.95 - 154.3
Se				<0.05-3.2	0.37	0.28										
U																
V	5-121	21	12												13.03 - 71.29	0.95 - 295.8
Zn	5-92	21	10	<12-263	76	73	1-219		48	32		<2-180	34		13.03 -71.29 (409.09 Scoria)	0.95 - 295.8 (1697.5 Scoria)

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

Table 23 Summary of agricultural irrigation water Default Guideline Value (DGV), Site Specific guideline value (SGV) and soil Added Contaminant loading limit (ACL) guidelines for heavy metals and metalloids

Metal	Suggested background [a] mg/kg	Biosolid MPC food production [c] mg/kg	Biosolids soil metal limits [d] mg/kg	Calculated soil ACL kg/ha	Suggested soil ACL kg/ha	DGV mg/L	SGV mg/L
Al	-	-	–	-	-	5	20
As	10	20	20	20	20	0.1	2
Be	-	-	–	-	-	0.1	0.5
B	1.0[b]	-	–	-	-	0.5	0.75 to 15
Cd	0.12	1	1	2	2	0.01	0.05
Cr (VI)	-	-	–	-	-	0.1	1
Co	27	-	–	-	-	0.05	0.1
Cu	28	100	100	140	140	0.2	5
F	-	-	–	-	-	1	2
Fe	-	-	–	-	-	0.2	10
Pb	18	150	300	195	195	0.2	2.0
Li	-	-	–	-	-	0.075	0.075 to 2.5
Mn	201	-	–	-	-	0.2	10
Hg	0.03	1	1	1.89	2	0.002	0.002
Mo	1	-	–	-	-	0.01	0.05
Ni	17	60	60	84	85	0.2	2
Se	0.5	5	–	9	10	0.02	0.05
U	-	-	–	-	-	0.01	0.1
V	-	-	–	-	-	0.1	0.5
Zn	39	200	300	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 22).

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

Where the ACL = - , there was insufficient background data to calculate the ACL.

c MPC = maximum permitted concentration in soil before further investigation (EPA NSW, 1997; DEHP Qld, 2016), note a 100 mg/kg concentration for chromium in DEPH Qld, (2016) was not included.

d (NZWWA 2003).

5.3 Aluminium

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

DGV Default guideline value in irrigation water	5.0 mg/L
SGV Site specific guideline value in irrigation water	20 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	Not determined

Aluminium is found in abundance in the geosphere (8% in the earth's crust) in complexes with oxygen, fluorine, and silicone. Aluminium compounds are very stable and are contained in all soils, mostly in alumino-silicate minerals, although aluminium may be present in an ion-exchangeable form in acidic soils (Scott-Fordsmand and Peterson 1995).

Crop yield and quality considerations

Surveys in the United States and the United Kingdom have reported aluminium concentrations in natural water sources of 0.014–1.2 mg/L. Total concentrations in some Australian water sources can be considerably higher due to the presence of clay minerals (aluminosilicates); for example, up to 18 mg/L in the Murray River (NHMRC and NRMMC 2011).

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 soil pH values less than 4.8 to 5.5 (Slattery et al. 1999). 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 toxicity values cannot be applied directly to irrigation waters because of soils' capacity 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). Toxicity values indicate that aluminium is toxic to plants at relatively low concentrations, and the irrigation water SGV and DGV have been developed to minimise the risk of phytotoxicity. An ACL for aluminium has not been determined, as it is inappropriate to set an ACL for a major soil constituent.

5.4 Arsenic

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

DGV Default guideline value in irrigation water	0.1 mg/L
SGV Site specific guideline value in irrigation water	2.0 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	20 kg/ha

Arsenic is a naturally occurring element that is widely distributed in the Australian environment. It exhibits complex chemical and environmental behaviours as it can exist in a number of oxidative states (Rajakovic and Rajakovic-Ognjanovic 2018). Arsenic can exist in anionic (negatively charged) and cationic (positively charged) form, the charge dictating the behaviour of the element in the 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

Arsenic in irrigation water can be from natural sources and mining/industrial/agricultural wastes (NHMRC and NRMMC 2011). In surface and groundwater not affected by arsenic mineral deposits or pollution, the concentration of arsenic is generally less than 0.005 mg/L, and have been reported in natural waters to range between 0.001 and 0.002 mg/L. However, in areas with a natural source of Arsenic, the concentration in water has been reported up to 12 mg/L (WHO 2011).

. 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). The median arsenic concentration in uncontaminated Australian soils is 5 mg/kg (Smith et al. 2003) (Table 22). The most common source of elevated As concentrations in soil in Australia is from mining activities, primarily in Western Australia and Victoria (Smith et al. 2003). Other sources are agriculture (cattle dips), forestry and industry. In New Zealand, elevated levels of arsenic in soils and waters occur as a result of geothermal activity, exacerbated by geothermal power exploitation, and the past and present use of arsenic-based pesticides (Robinson et al. 2004). 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 and Wolf 1963). Nutrient solutions containing 0.5–10 mg/L (depending on plant species) can result in toxic effects on crops (NAS and NAE 1973). Efroymson et al. (1997b) 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 DGV and SGV 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 (NRMMC 2004) and suitable soil background data have allowed the derivation of an ACL limit for arsenic in soils.

5.5 Beryllium

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

DGV Default guideline value in irrigation water	0.1 mg/L
SGV Site specific guideline value in irrigation water	0.5 mg/L
ACL Added contaminant loading limit 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

In Australian river waters, the concentration of Beryllium is reported to be < 0.00001 to 0.00012 mg/L (10–30 ng/L average) (NHMRC and NRMMC 2011). Beryllium can enter source water through the weathering of rocks, atmospheric deposition, and discharges. The primary source of beryllium in the environment is the burning of fossil fuels. Other less significant sources are slag and ash dumps and industrial wastewater (NHMRC and NRMMC 2011; Shah et al. 2016).

Beryllium concentrations in surface soils worldwide are typically 0.58 to 3.52 mg/kg and can range up to 60 mg/kg (Shah et al. 2016).

Beryllium is toxic to both animals and plants. There is 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 (Efroymson et al. 1997b). 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 and LeRiche 1968). The translocation of beryllium from the roots of the plants to the foliage does not occur readily (Shah et al. 2016). As there is 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, it is not possible at this stage to determine a soil ACL for beryllium.

5.6 Boron

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

DGV Default guideline value in irrigation water	0.5 mg/L
SGV Site specific guideline value in irrigation water	0.75 to 15mg/L. Refer to values in **Error! Reference source not found.**
ACL Added contaminant loading limit 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

The boron concentration in uncontaminated water sources is usually less than 0.5 mg/L (NRMMC et al. 2006; ANZG 2021).

Total concentrations of boron in soils of 10 mg/kg have been shown to cause no adverse effects on plants (Efroymson et al. 1997b) who also noted a toxic effect at 0.5 mg/kg. 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 22), placing these soils just above the limit for very sensitive crops.

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 **Error! Reference source not found.**. 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).

In general, maximum boron concentrations tolerated by plants in irrigation water without a reduction in yield or vegetative growth are approximately equal to soil water boron concentrations listed in Table 24**Error! Reference source not found.** (Ayers and Westcot 1985).

Table 24 Relative tolerance of agricultural crops to boron[a]

Relative Boron Tolerance of Agricultural Crops			
Very Sensitive (<0.5 mg/1)		**Moderately Sensitive (1.0 – 2.0 mg/1)**	
Lemon	*Citrus limon*	Pepper, red	*Capsicum annuum*
Blackberry	*Rubus spp.*	Pea	*Pisum sativa*
Sensitive (0.5 – 0.75 mg/1)		Carrot	*Daucus carota*
Avocado	*Persea americaiva*	Radish	*Raphanus sativus*
Grapefruit	*Citrus X paradisi*	Potato	*Solarium tuberosum*
Orange	*Citrus sinensis*	Cucumber	*Cucumis sativus*
Apricot	*Prunus armeniaca*	**Moderately Tolerant (2.0 – 4.0 mg/1)**	
Peach	*Prunus persica*	Lettuce	*Lactuca sativa*
Cherry	*Prunus avium*	Cabbage	*Brassica oleracea capitata*
Plum	*Prunus domestica*	Celery	*Apium graveolens*
Persimmon	*Diospyros kaki*	Turnip	Brassica rapa
Fig, kadota	*Ficus carica*	Bluegrass, Kentucky	*Poa pratensis*
Grape	*Vitis vinifera*	Oats	*Avena sativa*
Walnut	*Juglans regia*	Maize	*Zea mays*
Pecan	*Carya illinoiensis*	Artichoke	*Cynara scolymus*
Cowpea	*Vigna unguiculata*	Tobacco	*Nicotiana tabacum*
Onion	*Allium cepa*	Mustard	*Brassica juncea*
Sensitive (0.75 – 1.0 mg/l)		Clover, sweet	*Melilotus indica*
Garlic	*Allium sativum*	Squash	*Cucurbita pepo*
Sweet potato	*Ipomoea batatas*	Muskmelon	*Cucumis melo*
Wheat	*Triticum eastivum*	**Tolerant (4.0 – 6.0 mg/1)**	
Barley	*Hordeum vulgare*	Sorghum	*Sorghum bicolor*
Sunflower	*Helianthus annuus*	Tomato	*Lycopersicon lycopersicum*
Bean, mung	*Vigiia radiata*	Alfalfa	*Medicago sativa*
Sesame	*Sesamum indicum*	Vetch, purple	*Vicia benghalensis*
Lupine	*Lupinus hartwegii*	Parsley	*Petroselinum crispurn*
Strawberry	*Fragaria spp.*	Beet, red	*Beta vulgaris*
Artichoke, Jerusalem	*Helianthus tuberosus*	Sugarbeet	*Beta vulgaris*
Bean, kidney	*Phaseolus vulgaris*	**Very Tolerant (6.0 – 15.0 mg/l)**	
Bean, lima	*Phaseolus lunatus*	Cotton	*Gossypium hirsutum*
Groundnut/Peanut	*Arachis hypogaea*	Asparagus	*Asparagus officinalis*

a Maximum concentration of boron tolerated in soil-water without yield or vegetative growth reductions. Boron tolerances vary depending upon climate, soil conditions and crop varieties. Maximum concentrations in the irrigation water are approximately equal to these values or slightly less (Ayers and Westcot 1985)

Therefore, for crop protection from boron toxicity it is recommended that the values listed in **Error! Reference source not found.** are used to determine the SGV in irrigation waters. The DGV 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 and Westcot 1976). It is recommended that additional water quality monitoring of boron should be undertaken if these sources of irrigation water are used. Historically, boron concentrations in recycled water have been identified as a concern. However, recent data suggests that most recycled water would be less than 0.5 mg/L if high industry loads are not identified (NRMMC et al. 2006). Insufficient data is available at present to allow the determination of an ACL for boron.

5.7 Cadmium

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

DGV Default guideline value in irrigation water	0.01 mg/L
SGV Site specific guideline value in irrigation water	0.05 mg/L
ACL Added contaminant loading limit 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 and Pedersen 1995).

Crop yield and quality considerations

Low background concentrations of cadmium (0.001 mg/L) are found in many natural waters in Australia and worldwide (NHMRC and NRMMC 2011; ANZG 2021). However, cadmium can be released into the environment through wastewater, contamination of fertilisers, and metallurgical industries.

Uncontaminated soils in Australia generally contain around 0.05–0.10 mg/kg cadmium (McLaughlin et al. 1996). These concentrations are similar to New Zealand soil uncontaminated soil with a median ranging from 0.016 – 0.27 mg/kg (Table 22). However, fertilised agricultural soils may contain higher concentrations due to the addition of phosphate fertiliser containing cadmium as an impurity, or additions of manures, composts or biosolids (McLaughlin et al. 1996; Cavanagh 2014). In rural areas, cadmium inputs via atmospheric deposition may also contribute to elevated cadmium concentrations in the soil (Merry and Tiller 1991).

Cadmium is toxic to both animals and plants at low concentrations. Reported cadmium poisoning cases in Japan from 1947 to 1965 (Itai-Itai disease) led to increasing concern regarding cadmium in the environment, and much research done indicates that carcinogenity may also be a possibility (Merian, 1984). 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 and 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 4.2, Table 18).

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 (Efroymson et al. 1997b), but human and animal health concerns from ingestion of cadmium- contaminated crops are triggered at sub-phytotoxic concentrations. Therefore, the DGV and SGV have 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 (EPA NSW 1997; EPA Victoria 2004; DEHP Qld 2016), a cadmium ACL has been derived for soils receiving irrigation.

5.8 Chromium (VI)

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

DGV Default guideline value in irrigation water	0.1 mg/L
SGV Site specific guideline value in irrigation water	1.0 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	Not determined

In its pure form chromium is a steel-grey, bright, brittle and very hard metal that 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

Most surface waters contain between 0.001 and 0.010 mg/L of chromium (WHO 2003). In general, the chromium content of surface waters reflects the extent of industrial activity. In natural waters, chromium is present mainly in the trivalent chromium (III) and hexavalent chromium (VI) forms. Studies in lake water showed that the ratio of chromium (III) to chromium (VI) is affected by the amount of organic matter and dissolved oxygen (ANZG 2021).

Chromium is used in many industrial processes (alloy, tanning, textile dyes, pigments, glazes and treated timber) (Subramanian et al. 2003). Most soils and rocks contain small amounts of chromium oxide. Weathering, oxidation, and bacterial action convert this insoluble compound into soluble Cr(III) salts (NHMRC and NRMMC 2011). Background concentrations in soil range from 5 to 673 mg/kg (Table 22).

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.

Depending on prevailing redox conditions in the 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

(Efroymson et al. 1997b). Therefore, it is 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 1996) and supported by WHO (2006) irrigation water quality guideline for chromium of 0.1 mg/L. As there is no available data on chromium (VI) background concentrations in Australian soils or chromium (VI) toxic thresholds for soils, it is not possible to set an ACL limit at this stage. However, it is recommended that future guidelines attempt to set chromium (VI) soil limits.

5.9 Cobalt

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

DGV Default guideline value in irrigation water	0.05 mg/L
SGV Site specific guideline value in irrigation water	0.1 mg/L
ACL Added contaminant loading limit 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. 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

Concentrations in unpolluted surface waters are generally in the order of <0.001 mg/L (DWAF 1996). Soil concentrations range from < 2 to 165 mg/kg (Table 22).

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 and 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. Efroymson et al. (1997b) noted that concentrations in a solution of 0.06 mg/L may reduce the vegetative growth of plants. Cobalt in nutrient solution is toxic to tomatoes at a concentration of 0.1 mg/L, and this concentration approximates a toxicity threshold for other plants (DWAF 1996; WHO 2006). Given this data, a DGV of 0.05 mg/L in irrigation water is considered appropriate and protective for continuous use. The SGV 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 ACL has not been determined at this stage.

5.10 Copper

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

DGV Default guideline value in irrigation water	0.2 mg/L
SGV Site specific guideline value in irrigation water	5.0 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	140 kg/ha

Copper is widely distributed in rocks and soils as carbonate and sulfide minerals. Copper, is a near noble metal, only dissolving in oxidising acids.

Crop yield and quality considerations

Background concentrations of copper in freshwater have been noted in the range 0.00011 to 0.00015 mg/L and is usually present in uncontaminated surface waters at concentrations less than 0.01 mg/L (NHMRC and NRMMC 2011; ANZG 2021).

Copper is an important component of several plant enzymes and is essential in small concentrations for plant growth. For healthy plant growth, soil's copper content 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 and New Zealand 11.4 mg/kg (Table 22). However, copper concentrations in soils can range from 0.4–412 mg/kg (Table 22). Higher concentrations can occur due to the 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 the 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 and Brümmer 1984; Sanders 1982), and toxicity is therefore related to the soil's pH. In South Australia, the permitted added biosolids copper applied recognises the relationship between copper availability and toxicity on two variables; soil pH and organic carbon content. The added copper limit is based on these two variables (EPA SA 2020).

Copper toxicity from nutrient solutions has been noted at concentrations between greater than 0.03 mg/L (Efroymson et al. 1997b). Therefore, elevated levels of copper in irrigation water may have a direct phytotoxic effect on plants. To prevent this the DGV for copper has been set at 0.2 mg/L recognising soil adsorption. Given the existence of datasets for background concentrations of copper in Australian soils, and existing copper limits for agricultural soils receiving biosolids (EPA Victoria 2004; DEC 2012), a copper ACL has been derived for soils receiving irrigation.

5.11 Fluoride

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

DGV Default guideline value in irrigation water	1.0 mg/L
SGV Site specific guideline value in irrigation water	2.0 mg/L
ACL Added contaminant loading limit 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 0.5 mg F/L, although this is dependent on the sources of the water with some groundwater containing up to 10mg/L (NHMRC and NRMMC 2011). Excessive fluoride intake can lead to dental and skeletal fluorosis, characterised by hyper-mineralisation 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 fluoride affinity (Larsen and Widdowson 1971). Due to the soil's ability 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 that fluoride uptake and toxicity 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 Livestock drinking water). The DGV has been set on the assumption that that irrigation water could be phytotoxic to sensitive plants or contaminate livestock drinking water. The SGV has been set on the assumption that irrigation water could potentially contaminate livestock water. An ACL 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.

5.12 Iron

***It is recommended that the upper limit guideline values for iron in irrigation waters and soils should be*:**

DGV Default guideline value in irrigation water	0.2 mg/L
SGV Site specific guideline value in irrigation water	10 mg/L
ACL Added contaminant loading limit 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

Ferrous iron dissolved in irrigation water is relatively unavailable to plants as it oxidises (ferric iron) and precipitates upon aeration when applied to the 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 concentrations in uncontaminated surface waters are usually less than 1 mg/L and can range up to 5 mg/L or higher when supplied through rusting iron pipes (NHMRC and NRMMC 2011)

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 is insufficient data to determine a toxicity threshold of iron for plants growing in soils. Generally, there are no direct negative effects of iron in soil yet it can contribute to soil acidification and loss of availability of phosphorus and molybdenum (Efroymson et al. 1997b; WHO 2006; US EPA 2012). However, there have been a few reports of iron concentrations of approximately 10–50 mg Fe/L present in solution culture reducing plant growth (Efroymson et al. 1997b; WHO 2006; US EPA 2012). Iron deficiency develops mostly in alkaline soils, where iron precipitates as hydroxides and becomes unavailable to plants.

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 (WHO 2006). 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 (WHO 2006).

In view of the potential clogging of irrigation systems (trickle or drippers), the DGV 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 SGV has been set so that continual irrigation of plants will not expose them to phytotoxic concentrations of iron. An ACL for iron has not been determined, as it is inappropriate to set an ACL for a major soil constituent.

5.13 Lead

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

DGV Default guideline value in irrigation water	0.2 mg/L
SGV Site specific guideline value in irrigation water	2.0 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	195 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 and Pedersen 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 and Herms 1983). As for other cationic metals, low soil pH mobilises lead in soil allowing greater plant uptake (von Judel and 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 50–2600 mg/kg, while phytotoxicity of lead in solution is typically greater than 0.5mg/L (Efroymson et al. 1997b). Given the evidence from solution culture of potential direct lead toxicity to plants, the SGV and DGV have been set to minimise these risks recognising that lead is strongly retained by soils. Given the existence of datasets for background concentrations of lead in Australian soils, and existing lead limits for agricultural soils receiving biosolids which are a maximum acceptable concentration of lead in soil of 150 mg/kg (EPA NSW 1997; DEHP Qld 2016), a lead ACL has been derived for soils receiving irrigation. The guideline also considers the Canadian guideline for lead in irrigation water (CCME 2021).

5.14 Lithium

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

DGV Default guideline value in irrigation water	0.075 mg/L (citrus)
SGV Site specific guideline value in irrigation water	0.075 to 2.5 mg/L
ACL Added contaminant loading limit 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 is available on the background concentrations of lithium in Australian soils. Soil concentrations of lithium between 2 and 500 mg /kg in soil are toxic to a range of crops (Efroymson et al. 1997b), 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 1996). 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). Efroymson et al. (1997b) 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 SGV and DGV 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, an ACL limit has not been determined.

5.15 Manganese

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

DGV Default guideline value in irrigation water	0.2 mg/L
SGV Site specific guideline value in irrigation water	10 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	Not determined

Manganese is present in the environment in the divalent (Mn(II)), tetravalent (Mn(IV)), and heptavalent (Mn(VII)) states. Most of the divalent compounds are soluble in water (NHMRC and NRMMC, 2011). The most common tetravalent compound, manganese dioxide, is insoluble; however, the heptavalent permanganate is soluble.

Crop yield and quality considerations

Uncontaminated rivers and streams generally have low manganese concentrations, ranging from 0.001 mg/L to 0.6 mg/L (NHMRC and NRMMC 2011). High concentrations may occur in polluted rivers or under anoxic conditions, such as at the bottom of deep reservoirs or lakes or in groundwater. 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 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 is increased under reducing conditions (waterlogging) and low soil pH values. Toxicity to a number of crops usually only occurs in acid soils (WHO 2006).

Manganese is essential for plant growth, as it is involved in nitrogen metabolism and chlorophyll synthesis. Manganese is low in toxicity to animals and humans unless ingested in large amounts (NHMRC and NRMMC 2011; WHO 2011. 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 (Efroymson et al. 1997b). Manganese may also cause clogging of irrigation equipment due to oxidation of Mn^{2+} to MnO_2 (WHO 2006) DGV and SGV guidelines have been set to protect against direct phytotoxicity and damage to irrigation infrastructure. An ACL for manganese has not been determined as it is inappropriate to set an ACL for a major soil constituent.

5.16 Mercury

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

DGV Default guideline value in irrigation water	0.002 mg/L
SGV Site specific guideline value in irrigation water	0.002 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	2 kg/ha

In its pure form, mercury 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

Concentrations of total mercury in natural water generally low. Studies overseas have reported concentrations of less than 0.0005 mg/L, with some sources less than 0.00003 mg/L (30 ng/L) (NHMRC and NRMMC 2011). The highest value was 0.0055 mg/L from some wells in Japan.

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 22. 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 1974); however, carrots and mushrooms can accumulate mercury from soils. There is limited data describing mercury's toxicity to plants in soil (Efroymson et al. 1997b). Efroymson et al. 1997b derived a solution phytotoxicity benchmark of 0.005 mg/L for inorganic mercury and 0.0002 mg/L for methyl mercury. Given that solution culture studies indicate that there may be direct phytotoxic effects to plants of mercury in irrigation waters, the DGV and SGV 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 (EPA Victoria 2004; DEHP Qld 2016), a mercury ACL has been derived for soils receiving irrigation.

5.17 Molybdenum

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

DGV Default guideline value in irrigation water	0.01 mg/L
SGV Site specific guideline value in irrigation water	0.05 mg/L
ACL Added contaminant loading limit 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.

Crop yield and quality considerations

Molybdenum is present in ground and surface waters at very low concentrations, generally below 0.01 mg/L (NHMRC and NRMMC 2011). Higher concentrations have been reported in the vicinity of molybdenum mining operations. Fly ash deposited onto soils from coal-fired power stations can be a significant source of molybdenum. Application of fertilisers may also increase the concentration of molybdenum in ground and surface water.

Molybdenum commonly exists as an anion in waters and soils. The 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. Median concentrations of molybdenum in Australian soils are 1.0 mg/kg with a range of 0.2–20 mg/kg (Table 22). Plants absorb molybdenum predominantly as the MoO_4^{2-} anion from the soil solution and concentrate it in tissue. Tissue concentrations of >100 mg/kg 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 (WHO 2006)). High levels of molybdenum in livestock diets may also induce copper deficiency. A critical level of molybdenum for ruminants is 3–10 mg/kg dry matter (Axelson et al. 2018). 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 (Efroymson et al. 1997b). Given that toxic concentrations of molybdenum may arise in herbage at soil solution concentrations apparently below those at which phytotoxicity is noted, the DGV and SGV 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. An ACL limit has not been set for molybdenum due to lack of soil data or soil toxicity benchmarks.

5.18 Nickel

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

DGV Default guideline value in irrigation water	0.2 mg/L
SGV Site specific guideline value in irrigation water	2.0 mg/L
ACL Added contaminant loading limit in soil receiving irrigation water	85 kg/ha

Nickel is a silvery-white metal that 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 and Pedersen 1995).

Crop yield and quality considerations

Background concentrations of nickel in Australian and New Zealand waterway are very low (0.0001 to 0.00015 mg/L (ANZECC and ARMCANZ 2000).

Nickel concentrations in soils in Australia range from 5 mg/kg to 520 mg/kg with an average <100 mg/kg (CSIRO, unpublished). In New Zealand the median ranges from 1.9 to 27.4 mg/kg and can range up to 154 mg/kg (Table 22) Soils developed from serpentine rocks contain much higher nickel quantities (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 and Brümmer 1984).

Although nickel is now accepted as an essential micro-nutrient for plant growth (Marschner 1995), nickel has never been 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 greater than 0.13mg/L are toxic to plants (Efroymson et al. 1997b). Therefore, the DGV and SGV for nickel have been set to reduce the risk of direct nickel toxicity to plants. Given the datasets for background concentrations of nickel in Australian soils, and existing nickel limits for agricultural soils receiving biosolids (DEHP Qld 2016; EPA SA 2020), a nickel ACL has been derived for soils receiving irrigation.

5.19 Selenium

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

DGV Default guideline value in irrigation water	0.02 mg/L
SGV Site specific guideline value in irrigation water	0.05 mg/L
ACL Added contaminant loading limit 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 enzyme glutathione peroxidase activity.

Crop yield and quality considerations

Concentrations in unpolluted surface waters are generally in the order of <10 µg/L (DWAF 1996). 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.

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 22). The main issue regarding selenium in irrigation water is elevated selenium concentrations in forage crops and toxicity of selenium to animals consuming high selenium fodder. Cattle require selenium concentrations of 0.03–0.1 mg/kg in forage 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 1996). Plants can absorb relatively large amounts of selenium without displaying any phytotoxicity symptoms. Efroymson et al. (1997b) derived a phytotoxic soil 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 DGV and SGV 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 ACL limit has been set for selenium based on current regulatory guidelines (EPA NSW 1997; DEHP Qld 2016).

5.20 Uranium

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

DGV Default guideline value in irrigation water	0.01 mg/L
SGV Site specific guideline value in irrigation water	0.1 mg/L
ACL Added contaminant loading limit 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. Uranium may be present in the environment due to leaching from soils, rocks and natural deposits, release in mill tailings, combustion of coal and other fuels, and use of phosphate fertilisers (NHMRC and NRMMC 2011). Natural uranium consists almost entirely of the U-238 isotope.

Crop yield and quality considerations

Studies overseas have reported uranium concentrations in drinking water of generally less than 0.001 mg/L; however, concentrations as high as 0.7 mg/L have been reported in some private water supplies in Canada (NHMRC and NRMMC 2011).

Typical concentrations of uranium in surface soils range from 0.7–9 mg/kg, and as with most other cations, uranium binds strongly to negatively charged soil surfaces.

Only a small fraction of the uranium in soil is available to plants due to adsorption on soil particles and organic matter (Harmsen and de Haan 1980). Uranium taken up by plants usually accumulates in the roots (Hamilton 1974). Phytotoxicity resulting from elevated uranium concentrations in soils is thought to involve inhibition of enzyme systems and binding to nucleic acids. Efroymson et al. (1997b) note that phytotoxicity is considered 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 5 to 10 mg/kg (Efroymson et al. 1997b). Therefore, DGV and SGV 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 an ACL limit for uranium.

5.21 Vanadium

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

DGV Default guideline value in irrigation water	0.1 mg/L
SGV Site specific guideline value in irrigation water	0.5 mg/L
ACL Added contaminant loading limit 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.

Crop yield and quality considerations

Concentrations of vanadium in surface soils range from 5–250 mg/kg (Edwards et al. 1995, Olszowy et al. 1995). In Australian vanadium's median concentration from the survey of Olszowy et al. (1995) was 12 mg/kg (Table 22). Concentrations in uncontaminated surface waters are generally <0.0001 mg/L (DWAF 1996).

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 result from 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 (Efroymson et al. 1997b). Depending on soil type and species of plant, vanadium concentrations of 10 mg/kg soil are thought to inhibit crop growth (DWAF 1996). Efroymson et al. (1997b) set a vanadium solution concentration toxic benchmark of 0.2 mg/L. The limited information available indicates phytotoxicity at soil concentrations of >10 mg/kg and solution concentrations of 0.8 mg/L. The DGV and SGV have 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 ACL limit at this time.

5.22 Zinc

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

DGV Default guideline value in irrigation water	2.0 mg/L
SGV Site specific guideline value in irrigation water	5.0 mg/L
ACL Added contaminant loading limit 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 and Pedersen 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 (Weast 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

In surface and ground waters, the concentration of zinc from natural leaching is usually less than 0.01 mg/L (NHMRC and NRMMC 2011). Zinc can enter the environment from both natural processes (e.g. weathering and erosion) and anthropogenic (e.g. zinc production, waste incineration, urban runoff) processes (ANZG 2021).

Zinc concentrations in soil range from <2 to 295.8 mg/kg (Table 22) excluding the scoria rock-grop in New Zealand which has median zinc concentration of 409 mg/kg. Zinc is more readily available to plants in acid (pH_{CaCl_2}<6) light-textured soils (MacLean 1974; MacLean and Dekker 1978; Hornburg and Brümmer 1989). Toxicity in soil is reduced at pH >6.0 and in fine-textured or organic soils (WHO 2006). 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 (Efroymson et al. 1997b). Zinc toxicity in plants is evidenced by chlorosis, in leaf size reduction, necrosis of tips and foliage distortion (Chapman 1966). However, effects on symbiotic nitrogen-fixing bacteria may occur at lower soil zinc concentrations than those causing direct phytotoxicity (Chaudri et al. 1993). Therefore, the DGV and SGV for zinc have 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 ACL limit has been set for zinc based on current regulatory guidelines EPA Victoria 2004; DEHP Qld 2016). Modification of the regulatory guideline based on soil cation exchange capacity and pH has been adopted in South Australia (EPA SA 2020).

6 Nitrogen and phosphorus

***Upper-limit default guideline values (DGV) and site-specific or short-term guideline values (SGV) for nitrogen and phosphorus in irrigation water are defined in* Table 25*. They are based on maintaining crop yield, preventing bioclogging of irrigation equipment and minimising off-site impacts. Concentrations in irrigation water should not exceed the recommended upper-limit guideline value*.**

As N and P are macro nutrients, there may be lower-limit guideline that are appropriate also (LDGV and LSGV) when consider N and P concentrations in irrigation water with fertilizer requirements.

Table 25 Agricultural irrigation water long-term guideline value (DGV) and short-term guideline SGV in irrigation water (short-term — up to 20 yrs)

Element	DGV (mg/L)	SGV (mg/L)
Nitrogen	5	15 to 120[a]
Phosphorus	0.05[b]	0.5 to 12[a]

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

6.1 Methodology for development of guidelines

Developing site specific guidelines for major nutrients (nitrogen and phosphorus) for irrigation water is not trivial. This can be compounded by fertiliser applications. The different objectives are often conflicting and derived from different dimensions i.e., crop yield, economic, environmental, biophysical, and social. Sometimes market forces are blamed for excessive irrigation and fertiliser uses that result in adverse environmental consequences. This is a complex system and often site-specific condition must be considered. For example, irrigation in Great Barrier Reef catchments should focus on minimising/preventing nutrient, sediment, and agrochemical movements into the Great Barrer Reef Marine Park. The concepts of default guideline value (DGV) and short-term guideline value (SGV) 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 assess 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 (NRMMC et al. 2006; ANZG 2021).

6.1.1 Nitrogen

Given the potential for nitrogen to affect plant maturation, the DGV 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, Ayers and Westcot 1985; WHO 2006).

The SGV for nitrogen has also been developed to ensure that groundwater and surface water nitrogen does not exceed drinking water guidelines ((NHMRC and NRMMC 2011)) . 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 (11 mg/L nitrogen or 50 mg/L nitrate).

Volatilisation, denitrification and soil immobilisation provide safety margins against nitrogen overloading (EPA NSW 1997). Considering the range of nitrogen concentrations removed in harvestable portions of crops (Table 26), the SGV range quoted should be used as a guide only. Site-specific assessment for particular crops should be undertaken (see Section 6.2).

6.1.2 Phosphorus

Environmentally significant phosphorus concentrations in water are generally considered to be greater than 0.01 to 0.05 mg/L (ANZECC and ARMCANZ 2000; ANZG 2021) for eutrophication. Eutrophication is a major concern due to P pollution in waterways and it occurs in many places including groundwater dependent ecosystems. However, aquatic plant growth (including algae) is not dependent on phosphorus alone. Suppose all other nutrients and conditions are not optimal (e.g. poor light, high turbidity, high grazing rates, poor attachment substrates). In that case, some systems will cope naturally with relatively high nutrient loads without excessive aquatic plant growth. From the viewpoint of bioclogging of irrigation equipment, the DGV has been set low enough to restrict algal growth (i.e. 0.05 mg P/L), assuming all other algal growth conditions are adequate. The growth of toxic blue green algae and fish kills due to low DO are some of the potential impacts. Area specific guideline should consider protection of the local water quality objectives for relevant water bodies.

Major considerations in developing the interim site-specific SGV were: the fertiliser value of phosphorus in water to the crop; phosphorus removal from irrigation sites through the harvestable portion of crops; other fertiliser inputs; and soil phosphorus adsorption/retention capacities of soils. An inherent difficulty in setting an SGV for phosphorus is the complexity and site specificity of the phosphorus reactions in soil. To minimise off-site environmental impacts of phosphorus while considering agronomic implications, it is recommended that the site-specific SGV 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 and Murdoch 1997; Ritchie and Weaver 1993; Sharpley 1993; Stevens et al. 1999).

6.2 Nitrogen

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

DGV Default guideline value in irrigation water	5 mg/L
SGV Site specific guideline value in irrigation water	15–120 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 in four water soluble forms: 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).

6.2.1 Effects on crop growth and off-site impacts considerations

Nitrogen is an essential plant nutrient. Excess quantities of nitrogen in soil can lead to leaching into ground and surface waters, altered plant morphology and stimulation of algal growth and 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/L of N = 45 mg/L of NO_3^- or 13 mg/L of NH_4^+). The most available forms to plants are nitrate or ammonium. Nitrate is the usual form found in natural waters (Ayers and Westcot 1985), while NH_4^+ is the principal form found in sewage water before the water is recycled or treated (NRMMC et al. 2006). 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 guideline values.

Nitrate also poses a threat to animal and human health in drinking water, and plays an active role in eutrophication (ANZECC and ARMCANZ 2000; NRMMC et al. 2006; NHMRC and NRMMC 2011). Health-based guideline for drinking water for ammonia (0.5 mg NH_3/L or 0.4 mg N/L as NH_3), nitrate (50 mg NO_3^-/L or 11.3 mg N/L as NO_3 has been adopted for potable water for infants under 3 months old, and 100 mg NO_3^-/L or 22.6 mg N/L as NO_3 for those over 3 months old (NHMRC and NRMMC 2011). Health effects due to excessive nitrogen in water supplies include methaemoglobinemia and cancer (Follett 1989). Methemoglobinemia 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 blood's oxygen carrying capacity; this problem does not arise in adults (WHO 1984).

Nitrate may also be converted to suspected carcinogenic nitrosamines in the human digestive tract (Bouwer 1990). Nitrites should be kept below 3 mg NO_2^-/L or 0.9 mg N/L as NO_2^- based on health considerations (NHMRC and NRMMC 2011).

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 and Westcot 1985). Most crop yields are generally unaffected until nitrogen concentrations in irrigation water exceed 30 mg/L (Ayers and Westcot 1985).

Because the prokaryotic blue-green algae (cyanobacteria) can fix atmospheric nitrogen, it has been considered inappropriate to set a guideline value on eutrophication potential related to nitrogen concentrations in irrigation water. Moreover, phosphorus is considered the limiting nutrient for algae growth in many freshwater systems (Schmitz 1996).

Given the potential for nitrogen to influence plant maturation, the DGV 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 1996; Ayers and Westcot 1985; WHO 2006). The SGV range has been based on annual crop nitrogen usage and export (Table 26) and minimises the risk of ground and surface water nitrogen concentrations exceeding 11 to 23 mg N/L (i.e. generally fit for human consumption; see discussion above). The SGV 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, atmospheric depositions, N fixation, background N in soil and mineralisation or dilution on entering water bodies. These 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.

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 (e.g. MEDLI (DES QLD 2015)), as nitrogen uptake and removal from irrigation sites vary considerably with the type of crop grown (see Table 26).
No ACL has been determined because nitrogen is a major plant nutrient.

Calculating site-specific short-term guideline values for nitrogen

$$\mathrm{SGV_N} = \mathrm{N_{env}} + \mathrm{N_{removed}} + \mathrm{N_{gasloss}}$$ **Equation 28**

where:

$\mathrm{SGV_N}$ Site-specific guideline value for nitrogen (N) in irrigation water (mg/L)

$\mathrm{N_{env}}$ environmentally significant N concentration, i.e. >23 mg N/L potentially toxic to people > three-month-old or 11 mg N/L for infants < 3 months old (NHMRC and NRMMC 2011) to be conservative.

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

$\mathrm{N_{gasloss}}$ 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 recommended that a value of 0 be used, providing a safety margin.

For calculation of $\mathrm{N_{removed}}$:

$$\mathrm{N_{removed}} = \frac{\mathrm{N_{harv}} - \mathrm{N_{fert}}}{\mathrm{I_W} \times 10}$$ **Equation 29**

where:

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

$\mathrm{N_{harv}}$ nitrogen removed in harvestable portion of the crop (kg/ha/crop). Calculated by multiplying the mean N concentration in the crop to be grown (kg/t;Table 26) by the expected yield (t/ha; site-specific data).

$\mathrm{N_{fert}}$ nitrogen applied in fertilisers (kg/ha/crop)

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_{env} can supply up to 230 kg N/ha/crop, sufficient N for most crops (Table 26). 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) and 10 is the conversion factor for m^3/ha and nitrogen removed in the crop (kg/ha) to mg/L.

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

$$\mathrm{N_{removed}} = \frac{\left(3.4\ \mathrm{kg}\frac{\mathrm{N}}{\mathrm{t}} \times 50\mathrm{t/ha}\right) - 100\ \mathrm{kg/ha}}{1 \times 10\ L/ha} = 7\ \mathrm{mg/L} \tag*{\textbf{Equation 30}}$$

With 1 m of irrigation depth this equates to 70 kg/ha.

For calculation of $N_{gasloss}$:

$$\mathrm{N_{gasloss}} = (N_{env} + \mathrm{N_{plant}}) \times \mathrm{N_{dv}} \tag*{\textbf{Equation 31}}$$

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_{ens} and $N_{removed}$ are defined above.

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

$\mathrm{N_{gasloss}} = (11 + 7.0) \times 0.05$

$= 0.9\ \mathrm{mg/L}$

From the above examples:

$SGV_N = 11 + 7.0 + 0.9$

$= 18.9\ \mathrm{mg/L}$

Assumptions for calculating the SGV range

Irrigation height = 1.00 m

N_{env} = 11 mg/L

$N_{removed}$ = minimum (20 kg/ha) and maximum (1015 kg/ha) N removal by crops listed in Table 26 (excludes stubble crops) with no N added in fertilisers.

$N_{gasloss}$ = nil

The SGV 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 26), the SGV = 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 26 Indicative nitrogen and phosphorus removal with harvestable portions of crops

Crop	Moisture (%)	Harvestable Portion (t/ha)	N (kg/t)	P(kg/t)
Apple	84.0	--	0.3	0.1
Apricot	83.0	--	2.3	0.3
Artichoke	84.0	--	4.3	0.8
Asparagus	94.0	--	2.2	0.4
Avocado	--	--	1.3	0.2
Babaco	94.0	--	2.1	--
Banana	75.0	--	2.2	0.5
Barley	11.0	1.6 (1.5, 1.7)	17.6 (16.0, 18.2)	3.5 (2.7, 4.1)
Barley straw	--	1.6 (1.5, 1.7)	5.3	1.2 (1.2, 1.3)
Bean Fresh	91.0	9.8 (4.5, 15.0)	14.0 (2.5, 35.6)	0.6 (0.4, 0.9)
Bean Grain	9.8	1.8 (1.2, 2.3)	40.1 (38.0, 42.0)	5.5 (3.6, 7.7)
Beetroot	91.0	38.9 (17.7, 60.0)	2.5 (2.0, 2.8)	0.4 (0.3, 0.5)
Black currant	80.0	--	1.8	0.3
Blackberry	84.0	--	1.9	0.2
Blueberry	85.0	--	1.1	0.1
Broccoli	90.0	20.0	5.0 (4.5, 5.4)	0.7 (0.7, 0.8)
Brussels sprouts	88.0	25.0	6.2 (5.9, 6.5)	0.9 (0.8, 0.9)
Cabbage	92.0	51.3 (17.5, 80.0)	2.4 (1.5, 3.5)	0.4 (0.3, 0.6)
Canola	--	1.5	46.0	7.3
Canola / Rape	8.5	--	35.0	5.1
Cantaloupe/melon	87.0	--	1.9	0.6
Capsicum	92.0	--	2.2	0.3
Carambola	91.0	--	1.2	0.2
Carrot	89.0	53.1 (35.4, 80.0)	1.9 (1.3, 2.3)	0.3 (0.3, 0.4)
Casimiroa	80.0	--	0.1	0.2
Cassava	66.0	--	2.6	0.4
Cauliflower	91.0	42.7 (38.0, 50.0)	2.7 (2.4, 3.1)	0.5 (0.4, 0.6)
Celery	95.0	111.9 (50.0, 190.0)	1.8 (1.3, 2.5)	0.4 (0.3, 0.7)
Cereal rye	11.0	--	14.0	3.4
Cherry	80.0	--	1.5	0.2
Chickpea	10.0	--	33.0	3.8
Chicory	80.0	--	2.2	0.6
Chilli	81.5	--	3.4 (2.2, 4.5)	1.2
Chives	90.0	--	2.4	0.5
Chopped corn	--	--	12.0	2.4
Citrus fruit	--	--	2.9	0.4
Clover/grass	--	--	21.5 (21.0, 22.0)	1.9 (1.7, 2.0)
Coffee	--	--	46.0	3.4
Cotton	--	--	22.0	6.6
Cowpea	10.0	--	39.0	6.9
Cranberry	88.0	--	0.5	0.1
Cucumber	96.0	41.9 (18.0, 70.0)	1.5 (1.4, 1.6)	0.3
Currants	82.0	--	2.2	0.5
Custard apple	--	--	2.6	0.3
Date	21.0	--	3.6	0.5
Eggplant	93.0	--	1.8	0.3
Fennell	94.0	--	1.5	0.3
Fescue	--	11.0	24.0	4.0
Field Pea	10.0	1.0	37.5 (35.0, 40.0)	2.8 (2.0, 3.6)
Fig	83.0	--	2.2	0.3
Florence fennel	--	40.0	2.0	0.3

Crop	Moisture (%)	Harvestable Portion (t/ha)	N (kg/t)	P(kg/t)
Forage millet	--	6.7 (5.0, 9.0)	17.0	2.0
Forage sorghum	--	7.7 (6.0, 10.0)	18.0	3.0
Garlic	61.0	--	8.2	1.7
Gherkin	93.0	--	2.2	0.4
Ginger	89.0	--	1.8	0.4
Gooseberry	87.0	--	1.3	0.4
Grain Sorghum	--	2.9 (2.5, 3.5)	16.6 (12.0, 21.2)	2.5 (2.0, 3.2)
Grape	80.0	--	1.2 (1.0, 1.3)	0.3
Grapefruit	89.0	--	1.1	0.2
Grass	--	--	24.0	2.8
Guava	83.0	--	1.2	0.3
Hops	0.0	--	54.0	7.4
Horseradish	76.0	--	7.2	0.8
Iceberg	--	60.0	1.3	0.3
Kale	--	20.0	6.0	0.8
Kikuyu	--	30.0	26.0	3.0
Kiwifruit	84.0	--	1.5	0.2
Kohlrabi	--	45.0	2.8	0.4
Lablab	11.0	--	36.0	10.0
Lavender	30.0	--	4.5	0.5
Leek	91.0	55.0	2.3 (2.0, 2.5)	0.3 (0.2, 0.3)
Lemon and Limes	87.0	--	1.9	0.2
Lentil	10.0	--	37.0	3.3
Lettuce	96.0	41.8 (25.4, 50.0)	1.9 (1.8, 2.0)	0.3 (0.3, 0.4)
Linola	--	--	31.0	4.4
Linseed/Flax	8.5	--	25.0	3.8
Longan	72.0	--	1.6	0.1
Longanberry	--	--	2.8	0.2
Lucerne	--	29.0	31.5 (28.0, 35.0)	3.0 (2.0, 4.0)
Lucerne seed	--	--	60.0	6.8
Lupin	8.8	--	54.3 (48.0, 61.0)	3.8 (3.3, 4.3)
Lupins	--	1.4 (1.3, 1.4)	50.0	5.2 (5.0, 5.4)
Lychee	--	--	2.0	0.4
Maize	10.0	7.6 (1.1, 13.0)	11.9 (8.2, 16.1)	2.5 (0.9, 3.0)
Mandarin	--	--	1.6	0.2
Mango	90.0	--	6.5	0.8
Mangosteen	85.0	--	0.8	0.2
Medic seed	10.0	--	64.0	6.8
Millet/Canary Seed	11.0	--	20.0	3.3
Mulberry	89.0	--	3.5	0.4
Mushroom	91.0	--	6.0	0.8
Mustard	8.5	--	33.0	8.1
Nectarine	86.0	--	1.4	0.2
Oat straw	--	1.3 (1.1, 1.6)	7.1 (6.9, 7.3)	0.9 (0.7, 1.3)
Oaten	--	--	16.5 (13.0, 20.0)	2.1 (1.6, 2.5)
Oats	11.0	1.3 (1.1, 1.6)	16.6 (15.0, 17.3)	3.5 (2.7, 4.3)
Okra	90.0	--	3.1	0.6
Onion	89.0	52.0 (44.0, 60.0)	1.8 (1.8, 1.9)	0.4 (0.3, 0.4)
Orange	82.0	--	1.3	0.2
Parsley	83.0	--	5.8	0.7
Parsnip	81.0	--	3.8	0.9
Passionfruit	--	--	3.3	0.4
Pasture	--	--	22.0 (18.0, 26.0)	2.3 (1.8, 2.8)
Pawpaw	--	--	1.3	0.3
Pea straw	--	0.5	12.0	2.0
Peach/Peacharine	86.0	--	1.2	0.2
Peanut	10.0	--	36.0	3.2
Pear	85.0	--	0.2	0.0
Peas	81.5	--	8.0 (4.8, 11.2)	1.3

Crop	Moisture (%)	Harvestable Portion (t/ha)	N (kg/t)	P(kg/t)
Pepino	93.0	--	1.0	--
Peppers	74.0	--	5.9	0.8
Perennial ryegrass	--	6.0	35.0	3.0
Persimmon	--	--	1.0	0.2
Phalaris	--	9.0	11.0	3.0
Pigeon pea	10.0	--	31.0	7.6
Pineapple	--	--	0.8	0.1
Plum	86.0	--	1.5	0.2
Poppy	11.5	--	21.0	5.7
Potato	76.0	35.9 (31.7, 40.0)	3.0 (2.4, 3.3)	0.4 (0.4, 0.5)
Potato (tubers)	80.0	--	3.0	0.4
Prune	--	--	5.6	0.9
Pumpkin	90.0	--	2.1	0.6
Pyrethrum	--	--	17.0	2.2
Radicchio	--	25.0	2.5	0.4
Radish	93.0	30.0	2.8 (2.0, 3.5)	0.3
Raspberry	84.0	--	1.8	0.3
Rhubarb	95.0	--	1.1	0.2
Rice	14.0	--	10.3	2.4
Safflower	--	--	29.0	3.1
Serradella	10.0	--	--	4.9
Sesame	5.0	--	34.0	7.2
Silverbeet	93.0	--	2.9	0.4
Sorghum	10.0	--	15.8 (15.0, 17.0)	2.8 (2.3, 3.2)
Soybean	--	1.6 (0.9, 2.2)	50.4 (7.8, 66.1)	4.7 (1.1, 6.1)
Spinach	93.0	30.0	3.4 (3.2, 3.6)	0.4 (0.3, 0.5)
Squash	92.0	--	3.9	0.3
Stonefruit	--	--	1.2	0.1
Strawberry	91.0	--	1.9	0.3
Sugar Cane	--	--	0.9 (0.7, 1.2)	0.1 (0.1, 0.2)
Summer grain legumes	--	3.0	17.0	4.0
Sunflower	--	1.5 (1.2, 1.7)	44.5 (30.0, 51.8)	6.5 (5.8, 7.8)
Sweet Corn	--	--	3.9	0.6
Sweet Potato	--	24.0	2.5	0.6
Tea	--	--	40.0	4.0
Tobacco	--	--	39.0	2.5
Tomato	94.0	85.1 (38.0, 194.0)	1.6 (1.4, 1.9)	0.5 (0.3, 0.6)
Triticale	11.0	2.2 (2.1, 2.3)	13.3 (5.2, 20.0)	2.4 (0.9, 3.9)
Turnip	93.0	--	1.9	0.5
Vetch	10.0	--	42.0	4.2
Watermelon	94.0	--	1.5	0.3
Wheat	11.0	1.7 (1.5, 1.9)	23.5 (21.8, 28.0)	3.6 (2.5, 4.2)
Wheat straw	--	1.7 (1.5, 1.9)	5.3	1.2 (1.1, 1.3)
White clover	--	20.0	37.0	4.0
Winter cereals	--	5.0	15.0	3.0
Winter grain legumes	--	4.0	27.0	3.0
Zucchini	94.0	--	2.9	0.3

Sourced from: Sceswell and Huett 1998; Fink et al. 1999; SACES 1995; NSW Agriculture 1995; DL Reuter pers comm; interim data from project 5.4D of the National Land and Water Resources (NLWRA) Audit, 2000 (NLWRA 2001); and (ANZECC and ARMCANZ 2000). Values are averages if more than one data point and parenthesis are minimum and maximums reported

6.2.2 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 nitrogen losses 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 both nitrogen and carbon limited the soils' denitrifying capacity and maintained reclaimed water in the upper, more microbially active, part of the soil column through split applications, which 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). Suppose forms of nitrogen are transformed during water storage or irrigation to nitrate. In that case 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. 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 that influence ammonia's volatilisation are wind speed, soil/air temperature, and pH (Freney et al. 1983; Smith et al. 1996), as they increased (pH>7) volatilisation increases. The method of irrigation also has major impact on ammonia volatilisation during irrigation application (Smith et al. 1996; DES QLD 2015).

6.3 Phosphorus

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

DGV Default guideline value in irrigation water	0.05 mg/L
SGV Site specific guideline value in irrigation water	as calculated mg/L[a]

a Requires site-specific assessment. See below.

Phosphorus exists in 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.

6.3.1 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 many microorganisms' rapid growth (e.g. algal blooms; Schmitz 1996). If all other conditions are ideal for microbial growth and phosphorus is the limiting nutrient, increased phosphorus concentrations in irrigation water (>0.05 mg/L, Foy and 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 and 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 DGV 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 SGV 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.

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

6.3.2 Calculating site-specific guideline value for phosphorus

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; Haygarth and Jarvis 1999; Brodie 2012; Mekonnen and Hoekstra 2018), a guideline value for phosphorus in irrigation water has been set. Site-specific guidance should restrict environmentally significant concentrations of phosphorus (i.e. concentrations which could cause algal blooms) moving in to water bodies.

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

The model should also consider soil colloidal phosphorus, preferential macropore flow and surface fluxes of phosphorus. However, there are limited data presently available to easily quantify these phosphorus fluxes. Therefore, these phosphorus pools have been excluded from the model described below.

The soil phosphorus buffering capacities (PBC), or sorption capacities, assumptions based on the dominant soil order will be required or should be assessed on-site. However, consideration of plant-available soil phosphorus is also important to obtain suitable yields while minimising phosphorus movement off-site (Burkitt et al. 2002; Gourley et al. 2019). 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 and Withers 1998; Ulen 1998). Many of the limitations above 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 waster reuse guidelines that balance plant nutrient demands with nutrients applied through irrigation. This is attempts to reduce off-site impacts of excessive applications of nutrients by ensuring nutrients supplied do not exceed plant demands NRMMC et al. 2006.

Proposed changes to guideline values for other inorganic contaminants in irrigation water include two guideline values, the DGV and SGV. 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 DGV 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 SGV be low enough to prevent phosphorus in irrigation water overloading soil with phosphorus and allow environmentally significant phosphorus concentrations to move from soils into water bodies.

The model for calculating site-specific SGVs for phosphorus described below balances 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). The SGV for phosphorus can be calculated for site specific uses based on the plant available P required for crop growth, considering the current soil P status (Colwell P or Olsen P) and the environmentally significant P in water for the site considered as:

$$\mathrm{SGV_P} = \frac{(2.413\mathrm{PBI}^{0.119} - 1.43) \times (\mathrm{Ptarget} - \mathrm{Psoil})}{\mathrm{Irrigation\ Depth} \times 10} + \mathrm{Penv}$$

Where:

Ptarget = Colwell P target in soil for crop production (mg/kg) (e.g.Table 27)
PBI = Phosphate buffer index, PBI equation from Figure 14
Psoil = Colwell P measured in soil (mg/kg)
Penv = The environmentally significant phosphorus concentration in water, i.e. algal blooms occur > 0.05 mg P/L
10 is the conversion of kg/ha to the irrigation water equivalent of mg/L. For Olsen P use the alternate equation in Figure 14.
Irrigation Depth = 1 m.

For example, if the soil had a very low PBI of 50, the Ptarget = 22 mg/kg of Colwell P, the current soil Collwell P concentration (0 to 10 cm depth) was 15 mg/kg, the depth of irrigation depth was 1 m, and Penv = 0.05, then $\mathrm{SGV_P}$ = 1.74,

$$\mathrm{SGV_P} = \frac{(2.413 \times 50^{0.119} - 1.43) \times (22 - 15)}{1 \times 10} + 0.05$$

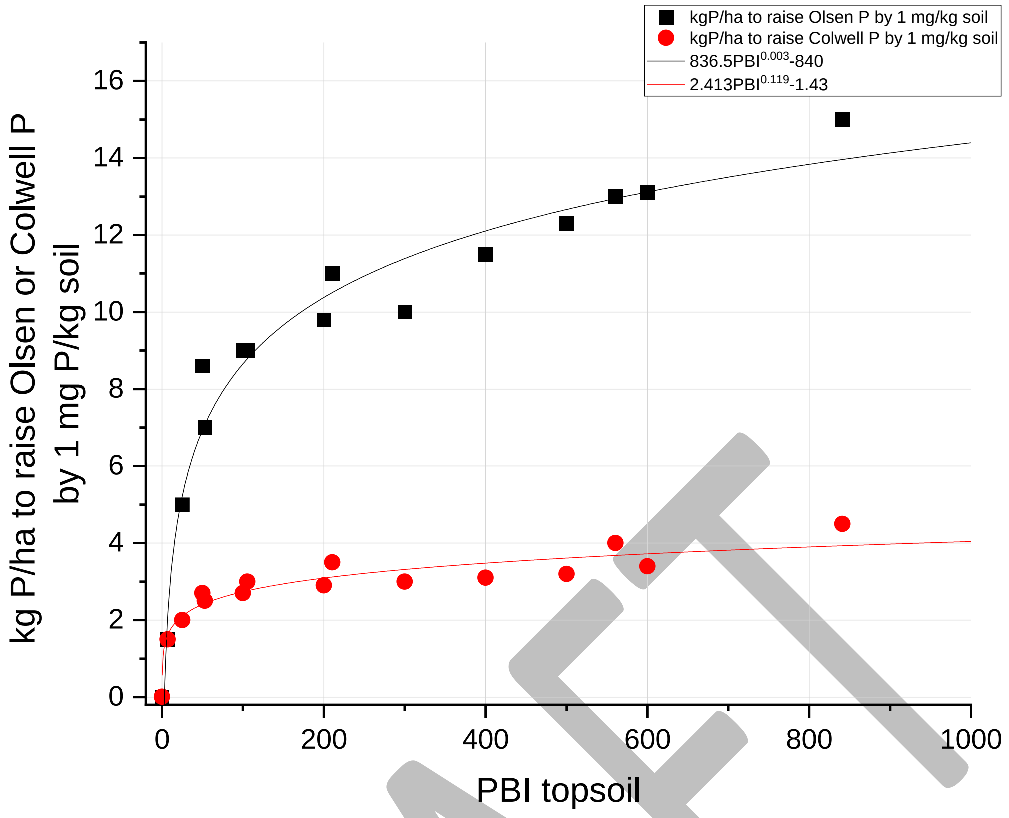

Figure 14 Estimates of phosphorous required to increase Colwell and Olsen P by 1 mg/kg in soil. These values should be confirmed with agronomists or suitably qualified advisors.

Source: Simpson et al. 2009; APAL 2013

Table 27 Interpolated critical Colwell-P values (mg/kg) for several crops at different P buffer capacity categories

P buffer capacity category	Temperate pasture	Wheat	Potato	*Medicago* spp.
Extremely low (PBI <15)	16	10	14	32
Very very low (PBI 15-35)	22	16	29	36
Very low (PBI 36-70)	27	22	44	39
Low (PBI 71-140)	32	29	65	41
Moderate (PBI 141-280)	39	38	96	n/a
High (PBI 281-840)	43 @ PBI 300	43 @ PBI 300	118 @ PBI 300	n/a
Very high (PBI >840)	n/a	n/a	n/a	n/a

After Moody (2007)

Note:

1) Given the range of values obtained for phosphorus sorption capacity, plant requirements and removal of phosphorus, and potential impact from movement of P off-site, it is recommended that site-specific data be assessed when assessing the SGV 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 SGV derived from the above equation would be 0.05 mg P/L. This would not be practical considering plants' nutritional requirements for phosphorus and current phosphorus fertilisation practices.

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

7 Organic matter

In some situations, the irrigation water may contain more organic matter than the soil can assimilate, causing the clogging of pores and anaerobic conditions to develop therefore impeding the soil microbial processes and plant root development. This is particularly the case when irrigating with effluent. The NSW guideline for the use of effluent for irrigation (NSW DEC 2004) specifies that an average loading rate of 1500kg/ha/month can be taken as the maximum organic loading for most soils. The minimum irrigation area required based on organic loading can be estimated as follows:

$$\text{Minimum irrigation area } A = CQ / (1{,}000 \times Lc) \qquad \text{Equation 32}$$

Where:
A = irrigation area (ha)
C = concentration of BOD_5 (mg/L)
Q = average effluent flow rate (kL/month)
Lc = critical loading rate of constituent (kg/ha/month)

Other regulations internationally specify various maximum levels of BODs for effluents depending on the type of crop irrigated. These values vary from a BOD of 5mg/L for crops eaten raw (ISO, 2020) to 60mg/L for irrigation of industrial and seeded crops (ISO, 2020). The US EPA recommends 10mg/L for food crop and 30mg/L for process food crops/non-food crops. In Australia, the organic loads are specified in each jurisdiction and in the AGWR (NRMMC, EPHC and AHMC 2006). They vary depending on the type of land use (recreational, forestry, etc) and crop irrigated (food/non-food). The range for these values is 10-80mg/L.

8 Pesticides

Guideline values for pesticides in irrigation waters are listed in* Table 28*. They consider likely adverse effects of herbicides on crop growth but do not consider potential impacts on aquatic ecosystems, residue accumulation in irrigated crops or human health impacts. 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 28 Interim guideline value concentrations for a range of herbicides registered in Australia for use in or near waters[a]

Herbicide	Residue limits in irrigation water [b]	Hazard to crops from residue in water[c]	Crop injury threshold in irrigation water (mg/L)
Acrolein	0.1 mg/L	+	Flood or furrow: beans 60, corn 60, cotton 80, soybeans 20, sugar-beets 60. Sprinkler: corn 60, soybeans 15, sugar-beets 15
Amitrole	0.002 mg/L	++	Lucerne 1600, beans 1200, carrots 1600, corn 3000, cotton 1600, grains sorghum 800
Asulam		++	
Atrazine	10 µg/L	++	
Bromacil	0.2 µg/L	+++	
Chlorpyrifos		++	
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
Diazinon			
Dicamba	0.006 µg/L	++	Cotton 0.18
Dichlobenil		++	Lucerne 10, corn 10, soybeans 1.0, sugar-beets 1.0–10, corn 125, beans 5
Diquat		+	
Diuron	0.002 mg/L	+++	
2,2-DPA (Dalapon)	0.004 mg/L	++	Beets 7.0, corn 0.35
Endosulfan			
Fosamine[¥]		+++	
Fluometuron		++	Sugar-beets, alfalfa, tomatoes, squash 2.2
Glyphosate		+	
Hexazinone		+++	
Karbutilate[¥]		+++	
Malathion			
Metolachlor			
Molinate		++	

Paraquat		+	Corn 10, field beans 0.1, sugar-beets 1.0
Picloram		+++	
Propanil	0.5 µg/L	++	Alfalfa 0.15, brome grass (eradicated) 0.15
Simazine	0.5 µg/L	++	
TCA (Trichloroacetic Acid) *		+++	0.5
Terbutryne		++	
Thiobencarb			
Triclopyr		++	
Trifluralin			

a From ANZECC (1992). These should be regarded as interim guideline 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.

¥ Pesticide not registered or approved anymore by APVMA as of April 2021

Other regulation guideline values are summarised in Table 29.

Table 29 Residue limits for selected pesticides in Canada (CCME 2009)

Pesticide	Residue limits in irrigation water (µg/L)
Atrazine	10
Bromacil	0.2
Dicamba	0.006
Propanil	0.5
Simazine	0.5

In Canada, irrigation water quality guidelines are based on the most sensitive crops grown, allowing the derivation of acceptable soil concentration or acceptable application rates of the pesticide and a species maximum acceptable toxicant concentration (SMATC), provided the maximum irrigation rate is known (volume per unit area per year). The lowest SMATC in each crop group is selected and represents the GV (Hamilton et al. 2003).

In the European Union, the presence of pesticides in water is regulated through the Ground Water Directive (European Parliament and Council 2006), the Drinking Water Directive (EPC 1998) and the Water Framework Directive (EC 2000), They set a maximum concentration of 0.1 µg/L for individual pesticides and their degradation products, and 0.5 µg/L for total pesticide residues present in a water sample. There is no specific Directive on the pesticide concentration in irrigation waters though. Environmental Quality Standards have been set for priority substances and other pollutants, amongst which there are a number of pesticides. They specify an annual average and maximum allowable concentration not to be exceeded in different inland or other surface waters (European Parliament and Council 2008) (Table 30).

Table 30 Environmental Quality Standards (EQS) for priority pesticides and certain other pollutants in the European Union

Name of substance	AA¥-EQS inland surface waters	AA-EQS other surface waters	MAC*-EQS inland surface waters	MAC-EQS other surface waters
Atrazine	0.6	0.6	2.0	2.0
Chlorfenvinphos	0.1	0.1	0.3	0.3
Chlorpyrifos	0.03	0.03	0.1	0.1
Diuron	0.2	0.2	1.8	1.8
Endosulfan	0.005	0.005	0.01	0.004
Isoproturon	0.3	0.3	1.0	1.0
Simazine	1	1	4	4
Trifluralin	0.03	0.03	NA	NA

¥AA: annual average (µg/L)

*MAC: maximum allowable concentration (µg/L)

8.1 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. There is however a wealth of information on pesticide residues in surface and groundwaters worldwide. In Australia, in a survey of surface water quality across 73 sites, pesticide residues were detected in 28% of the water samples (along with a range of personal care and pharmaceutical products) (Scott et al. 2014). Herbicides have been found to be the most commonly occurring pollutants found in the Great Barrier Reef marine environment as a result of runoff from agricultural areas (Thorburn et al. 2013).

Pesticides are mainly organic compounds, or in some cases organo-metallic compounds (eg arsenic pesticides although their use is now prohibited in most countries), and are categorised according to their intended use; as insecticides (controlling insect pests), herbicides (controlling weeds), fungicides (control of fungal pests) or nematicides (controlling nematodes). Pesticides encompass a broad range of natural and synthetic compounds of widely differing chemical composition. Each category of pesticide is often grouped into classes of chemically similar compounds; for example, the organochlorine and organophosphate insecticides, pyrethroids, triazines, triazoles, neonicotinoids.

All are screened for health and environmental effects prior to registration for use. Agricultural chemical products must be registered by the Australian Pesticides and Veterinary Medicines Authority (APVMA) prior to use in Australia or by the Environmental Protection Authority and Ministry for Primary Industries in New Zealand. Information about registered products and molecules can be obtained on the databases Pubcris or Infopest. 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 and Simpson 1996).

Pesticide residues are ubiquitous in the environment, 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; transport in runoff waters (Hunter 1992; CCREM 1987; Schofield and Simpson 1996) and groundwaters. In runoff waters, pesticides can be mobilised either under soluble form or sorbed on soil and colloidal particles (Oliver et al. 2012). Leaching of pesticide residues to groundwaters can occur, with the extent of leaching dependent on a range of factors such as the chemical and physical properties of both the pesticide (K_{OC}, K_{OW}, pKa, solubility in water), the soil (organic matter nature and content, porosity) the rainfall regime (volume and intensity of rainfall events – McGrath et al. 2010). Residues of several pesticides, notably the herbicide atrazine, have been found in surveys of some Australian groundwaters, (Keating et al. 1996; Schofield and Simpson 1996; Shaw et al. 2012).

Many factors influence the persistence of pesticide residues in the soil/plant/aquatic environment, 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 and in other regions around the world for decades (Schofield and Simpson 1996; Weaver et al. 2012).

8.2 Derivation of guidelines

While there is a comprehensive list of guideline values for pesticide residues in drinking water in Australia (NHMRC and NRMMC 2011), few guidelines exist for residues in waters used for irrigation purposes. In light of the limited information available, the ANZECC and ARMCANZ (2000) 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 or terrestrial ecosystems or accumulation of residues in crops. 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.

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Aquatic ecosystems impacts are addressed in Chapters 3 and 8 of the Australian and New Zealand guidelines for fresh and marine water quality.

Finally, new compounds are constantly being developed by agrochemical companies and submitted to approval by the competent regulatory bodies (APVMA in Australia). As a result, the list presented in Table 28 can never be exhaustive. A robust method of deriving guideline values (GV) and SGV for terrestrial systems is therefore necessary using similar methodology as the technical briefs developed for aquatic ecosystems. Technical briefs for the toxicant aquatic ecosystem default guideline values in water have been developed and provide additional data on the guideline value derivations, including information on toxicant use and sources, toxicity and factors affecting toxicity. Two types of technical briefs are provided, depending on when the toxicant DGV was published (2000 or post 2017) (ANZG 2018). Each technical brief details the chemistry and ecotoxicity of the toxicant, followed by a comprehensive tabulation of all the toxicity data considered in the guideline value derivation and evidence that the data used passed the necessary quality assurance checks (Warne et al. 2018). Burrlioz 2.0 SSD plots are included, as are details of toxicity testing data, including physicochemistry of the test media, such as pH, temperature and hardness. Full references to all data sources are provided in each technical brief. They are accessible on https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/water-quality-toxicants/search (accessed 7 April, 2021).

These topics are further discussed in Section 11**Error! Reference source not found.** regarding further research and information needs.

9 Radiological quality

Guideline values for the radiological quality of irrigation waters are given in Table 31. The same guideline values also apply for livestock drinking water use.

Table 31 Guideline values for radiological contaminants in irrigation water[a]

Radionuclide	Guideline value (DGV)
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 DGVs also apply for livestock drinking water.

Guidelines values internationally are not specifically derived for irrigation water but mostly for drinking purposes. Most drinking guideline value assume an annual dose of 0.1mS/year. They are summarised in Table 32.

Table 32 Summary of international guidelines for radiological quality in drinking water.

Parameter	**Guideline value**			
	Canadian drinking water guidelines (2009)	Australian drinking water guidelines (2011)	US EPA National Primary Drinking water Regulations	WHO (2017)
Radium 226		Must be screened if gross α or β guideline values exceeded	0.19 Bq/L	1 Bq/L
Radium 228	0.5 Bq/L	Same as above	0.19 Bq/L	0.1 Bq/L
Uranium 238	0.02 mg/L (total U)	See a		10 Bq/L or 30µg/L for total U b
Lead 210	0.2 Bq/L			0.1 Bq/L
Gross alpha (α)		0.5 Bq/L	0.56 Bq/L	0.5 Bq/L
Gross beta (β) (excluding K-40)		0.5 Bq/L	1.85 Bq/L	1 Bq/L

a concentration, as well as that of other radionuclides, should be determined to ensure all gross alpha and beta activity is accounted for, after taking into account the counting and other analytical uncertainties involved in their determination (NHMRC, 2011)

b Separate guidance levels are provided for individual uranium radioisotopes in terms of radioactivity (i.e. expressed as Bq/l). The provisional guideline value for total content of uranium in drinking-water is 30 µg/l based on its chemical toxicity, which is predominant compared with its radiological toxicity (WHO, 2017)

9.1 Description

Radioactive elements in the environment originate both from terrestrial and cosmogenic sources. Cosmogenically produced radionuclides are deposited onto the terrestrial environment through rainfall or dry deposition, and subsequently become part of the hydrological cycle.

Groundwater is a major source of water for agriculture in Australia and the most significant radionuclides naturally occurring in groundwaters are those associated with the uranium and thorium series as the parent nuclides are considered to have been present during the formation of the earth (Kleinschmidt 2011). More specifically, Radium-226, radium-228 and uranium- 238 are the natural radionuclides which are often detectable in groundwater supplies. Therefore, it is important to determine the radiation dose associated to their presence in water bodies. 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 radionuclides arising from activities such as processing of minerals containing uranium and thorium (e.g., from the processing of phosphate ores for fertiliser production, (Pearson et al. 2019)) also needs to be considered in assessing the radiological quality of stock or irrigation waters. Similarly, radionuclides used for medical purposes (for diagnosis or therapeutic activities) and research end up in wastewater treatment plants and the reuse of water in irrigation can present a risk to the environment even after treatment.

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 in soils from nuclear fallout have decreased substantially in the Australian environment since the tests of nuclear weapons in the 1950s and 1960s and are no longer significant. They have still been detected in some Australian soils, however their concentrations are well below the levels of natural radionuclides (M Cooper unpublished). UNSCEAR (2000) estimated that the Southern hemisphere received about a third of global fallout from nuclear atmospheric tests. Nearly sixty years later at the time of the update of these Guidelines, almost two half-lives of ^{137}Cs have passed (30.07 year half-life) and it is therefore difficult to detect this radionuclide in the soil hence the need to develop new radioisotopes for soil erosion and sediment tracing and in order to baseline in case of future radionuclide fallout. Plutonium can be one of them (Lal et al. 2017; Meusburger et al. 2020). In the Sydney region, ratios of ^{240}Pu/^{239}Pu in creek sediments were found to be lower than the global average (estimated to be around 0.18) and similar to those reported for other locations away from nuclear testing sites (Smith et al. 2016. Plutonium ratios in soils from locations close to nuclear weapons test sites in Australia (Maralinga in SA, Montebello Islands off the coast of WA) were different from the global mean, making it possible for the nuclear testing to have influenced regional fallout concentrations (Child and Hochkis 2013; Lal et al. 2017).

9.2 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 and 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). Estimations of the average annual radiation dose from natural sources in Australia is estimated to be about 1.5 mSv (ARPANSA, accessed 15 april 2021 and Drinking water guidelines 2011). Webb et al. (1999) estimated the annual dose in Australia to be at approximately 2mSv/year. National guidelines for drinking water quality in Australia (NHMRC 2011) 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 x 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.

9.3 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 2011) and WHO (2017), but using an annual committed effective dose of 1 mSv instead of 0.1 mSv to calculate guideline values for specific radionuclides in irrigation and stock waters. The following background information and assumptions are as follows (Table 33):

Table 33 Dose coefficients for ingestion of radionuclides by adult members of the public (ICRP 1996).

Category	Radionuclide	Dose coefficient (mS/Bq)	Type of emitter
Natural Uranium series	Uranium -238	4.5 x 10^{-5}	alpha
	Radium 226	2.8 x 10^{-4}	alpha
Natural Thorium series	Radium 228	6.9 x 10^{-4}	beta

The guideline value (Bq/L) for radionuclides in drinking-water are calculated using the following equation:
Annual dose (mSv/year) = Dose per unit intake (mSv/Bq) x Annual water consumption (L) x Radionuclide concentration (Bq/L).
The annual water consumption is estimated at 730L/year (WHO 2017), which is a daily intake of 2L per adult person.
Only key natural radionuclides have been considered. It should be noted that the DGV for uranium-238 is based on chemical toxicity considerations rather than on radiological grounds.

No guideline values are presented for other natural radionuclides, such as thorium isotopes, lead-210 or polonium-210, because they are rarely found in surface or groundwater insignificant 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 guideline 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.

10 General water uses

10.1 pH

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

Parameter	**LSGV**	**LDGV**	**DGV**	**SGV**
pH	**5.0**	**6.0**	**8.5**	**9.0**

10.1.1 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:

$pH = - \log [H^+]$

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 and carbonate and indirectly sodium ratios with calcium and magnesium; these issues are addressed separately (see Sections 4.1 and 4.3).

10.1.2 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).

10.1.3 Derivation of DGVs

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). The World Health Organisation indicates that a normal pH range in irrigation water ranges from 6.5 to 8.0 (WHO 2006).

10.2 Corrosion

Guideline values for assessing the corrosiveness of water are given in Table 34.

Table 34 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	<60mg/L $CaCO_3$	Increased corrosion potential
Langelier Index	<-0.5	Increased corrosion potential
	-0.5 to 05	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. (NHMRC and NRMMC 2011). The rate of corrosion is dependent on a range of interacting factors including pH, electrical conductivity, dissolved oxygen, and temperature (McFarland et al. 2008).

10.2.1 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 occur) 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.

Importantly, all metals will react and corrode differently in response to different physiochemical parameter concentrations (WHO 2017).

- Biological processes

Microorganisms can also increase the rate of corrosion through the formation of biofilms on the metal surface. Microorganisms can cause corrosion through depletion of dissolved oxygen, producing sulfuric acid, preventing corrosive metabolites and participating in the cathodic process (NHMRC and NRMMC 2011). 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. Iron and sulfur bacteria particularly of iron and steel components (NHMRC and NRMMC 2011).

- 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 (total dissolved solids) above which corrosion will establish varies, depending on the individual flow situation. As total dissolved solid concentration increases, corrosion (and fouling) typically increases as well (CCME 2008).

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 and 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 and Westcot 1985). Aggressive ions such as chloride and sulphate can also contribute to dissolution of lime in concrete (WHO 2017). 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 and potential failure (WHO 2017).

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.

10.2.2 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. Several sources conclude that a pH of 6.5-8.5 is the ideal range to limit both corrosion and fouling (NHMRC and NRMMC 2011; CCME 2008; WHO 2017). Along with several other parameters (eg. Dissolved oxygen, temperature, ions in water), pH is used in the calculation of the Langelier Index, which provides an indication of the corrosion or scaling potential of a water (NHMRC and NRMMC 2011).

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). Although other factors such as pH, alkalinity and dissolved oxygen can also influence water corrosiveness (NHMRC and NRMMC 2011), it is recommended that waters be maintained at a hardness level of >60 mg/L ($CaCO_3$) to minimise corrosion (EEC 1997; NHMRC and ARMCANZ 1996).

- Dissolved oxygen

Dissolved oxygen (DO) is the main oxidising agent which causes corrosion, with the tendency to corrode increasing with increasing DO concentration (WHO 2017). 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 and 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 (WHO 2017). Low DO can enable nuisance anaerobic microorganisms to grow producing by-products that can increase corrosiveness, and increase scaling (NHMRC and NRMMC 2011).

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. The Australian Drinking Water Guidelines suggest a DO concentration of >85% saturation can limit corrosion (NHMRC and NRMMC 2011).

- 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:

$CO_2 + H_2O \leftrightarrow H_2CO_3 \leftrightarrow H^+ + HCO_3^-$

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

10.2.3 Corrosion indices

- Langelier Saturation Index (LI)

It is important to be aware that no corrosion index can apply to all types of materials and they are best used as one aspect of understanding water corrosion (WHO 2017).

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:

$$\mathrm{CaCO_3 + H^+ \rightarrow Ca^{2+} + HCO_3{-}}$$

The saturation index is calculated as:

$LI = pH - pH_s$

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 and 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. In South Africa, the target water quality range for LI is -0.2 to +0.2 where no major problem with either corrosion or scaling of irrigation equipment is to be expected (DWAF 1996).

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. DWAF (1996) specifies that the corrosive tendencies appear when RI>6.5

$RI = 2pH_s - pH$

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

The ratio of chloride to carbonate is especially relevant in relation to cast iron and steel water infrastructure (WHO 2017).

10.2.4 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 and 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.

All materials react, and corrode, differently in response to varying water parameters (WHO 2017). It is therefore important to test water quality before constructing (or replacing) water infrastructure to ensure selection of the most suitable materials.

- 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) and the CCME (2008) 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.

10.2.5 Derivation of guideline 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.

At the time of the 2000 update, the National Centre for Groundwater Management (University of Technology, Sydney) had released a number of publications based on extensive research, bringing together current issues on corrosion and groundwater wells which had been used for the derivation of guideline values. The research had 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 guideline values given in this review. The 2021 update and literature review did not identify any significant changes in these guideline values.

10.3 Fouling

Guideline values for assessing the fouling potential of waters are given in Table 35.

Table 35 **Fouling potential of waters as indicated by pH, hardness, Langelier index, Ryznar index and the log of chloride to carbonate ratio**

Parameter[a]	**Value**	**Comments**
pH	<6.5	Limited fouling potential
	7 to 8.5	Moderate fouling potential (groundwater)[a]
	>8.5	Increased fouling potential (groundwater)[b]
Hardness	>350mg/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.

10.3.1 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 36), which is also influenced to some extent by the materials used in construction.

Table 36 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

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 37).

Table 37 Water quality parameters relative to the potential for drip emitter clogging

Water quality parameter	Symbol	Unit	Default impact rating for risk of clogging emitters		
			Insignificant	Minor	Moderate to high
Total suspended solids (includes algae)	TSS	mg/L	<50	50 to 100	>100
pH	pH	$-\log H^+$	<7	7 to 8	>8
Total Dissolved Solids	TDS	mg/L	<500	500 to 2000	>2000
Manganese	Mn	mg/L	<0.1	0.1 to 0.5	>1.5
Total iron	Fe	mg/L	<0.2	0.2 to 1.5	>1.5
Hydrogen sulfide	H_2S	mg/L	<0.2	0.2 to 1.5	>1.5
Bacterial number	n/a	Count/100ml	<10,000	10,000 to 50,000	>50,000

Source: (Asano et al. 2007). Default impact assumes no maintenance and emitters are not self-cleaning

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. These particles are reflected in the concentration of total suspended solids (TSS) where a high TSS can often be associated with excessive scaling and build-up (NHMRC and NRMMC 2011). Ayers and Wescot (1985) give recommended guidelines for concentrations of suspended solids (TSS) 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 and 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 and 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, guideline 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 and Knight 1989), which are dependent on a number of factors including the temperature and ionic strength of water.

$CO_2 (g) \leftrightarrow H_2CO_3$
$H_2CO_3 \leftrightarrow H^+ + HCO_3-$
$HCO_3- \leftrightarrow H^+ + CO_3^{2-}$
$CO_3^{2-} + Ca^{2+} \leftrightarrow CaCO_3$

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 inTable 38.

Table 38 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 (NHMRC, NRMMC 2011). 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 and Knight 1989). Iron bacteria growth can lead to blockages and restrictions in water infrastructure (NHMRC and NRMMC 2011).

Manganese is commonly associated with iron in groundwaters and can contribute to biofouling through similar depositional mechanisms (NHMRC and NRMMC 2011). 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). Ammonia concentration levels >0.5 mg/L can also result in nuisance growth of microorganisms (NHMRC and NRMMC 2011). Nuisance growth of microorganisms can be exacerbated and enabled by low DO levels (NHMRC and NRMMC 2011).

10.3.2 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. Parameters that can influence fouling include iron, ammonia, silica, TDS, DO and temperature (NHMRC, NRMMC 2011).

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 guideline value of 200 mg/L $CaCO_3$ is recommended to limit excess encrustation. This value takes into consideration the influence of hardness on fouling rates. Tis value is also recommended by WHO (2017) and the Australian Drinking Water Guidelines (NHMRC and NRMMC 2011) who suggest that hardness of 200 mg $CaCO_3$/L can start to increase 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 10.1). A pH range between 6.5 and 8.5 is typically accepted as limiting corrosiveness (NHMRC and NRMMC 2011).

10.3.3 Fouling indices

Langelier's index

Langelier's Index (see Section 10.2 on corrosion) uses pH values to estimate the potential for calcium carbonate precipitation. A positive value indicates that precipitation is likely to occur, which can cause encrustation and fouling, while a negative value indicates the potential for corrosion.

Ryznar's index

Ryznar's Index (see Section 10.2 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 $({}^{[Cl^-]}/{}_{[CO\ ^{2-}]})$ is small (see Section 10.2 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 and Kemp 1975).

10.3.4 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. Care should be taken when reducing the pH or hardness of water as lowering too much can increase the corrosive potential of water (NHMRC and NRMMC 2011).

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 (NHMRC and NRMMC 2011).

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.

10.3.5 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, At the time of the 2000 update, the National Centre for Groundwater Management (University of Technology, Sydney) had recently released a number of publications based on extensive research, bringing together current issues on corrosion and fouling in groundwater wells, which can be applicable to many agricultural situations and had been used for the derivation of guideline values.

Publications from these studies provided the basis of many issues discussed in this review, in conjunction with other relevant information.

10.4 Agricultural chemical preparation

Water used to mix and spray agricultural chemicals such as pesticides or fertilisers is of great importance due to potentially unwanted chemical reactions related to water quality and affecting the performance of the agricultural preparation. Commercial products labels usually specify the requirements regarding water quality.

10.4.1 Salinity

Elevated salinity levels in agricultural waters can result in the formation of precipitates after mixing with particular chemical compounds or cause their inactivation or breakdown. 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 and 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. Salinity levels above 1000mg/L NaCl or ECs above 500-1000 µS/cm can result in some chemicals precipitating out of the solution and others being inactivated (GRDC 2014).

10.4.2 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 et al. 1999). Sulfonyl urea herbicides will break down more quickly in acidic conditions and therefore will perform better in water that is slightly alkaline.

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.

10.4.3 Hardness

Hardness is generally defined by the presence of calcium and magnesium salts in water. It is expressed as calcium carbonate equivalent (mg/L $CaCO_3$). More information on hardness is provided in 10.2 (General Water Uses – Corrosion). Consequences of the use of hard waters on agricultural chemical preparations are diverse:

Formation of precipitates: High levels of calcium and/or magnesium ions can result in the formation of unwanted precipitates in agricultural preparations by affecting the properties of surfactants, emulsifiers and wetting agents (Agriculture NSW 2012), Precipitates can block nozzles and pre-filters and cause additional wear of spray rigs (Agriculture NSW 2012).

Reduced solubility: Chemicals with amine formulations, which include the herbicides: glyphosate, 2,4-D amine, MCPA amine and dicamba are adversely affected by hard water. The solubility of the herbicide is reduced which leads to it being less absorbed by the weeds. Schortgen and Patton (2020) showed that 2,4-D amine, efficacy was reduced when the $CaCO_3$ level in water was at least 390mg/L (for horseweed control). This was overcome with the addition of ammonium sulfate. Typically, water containing more than 250-350 meq/L $CaCO_3$ should be treated before using some herbicides, especially if pH is above 7 (GRDC 2014).

10.4.4 Specific ions

The presence of certain ions in agricultural waters (like bicarbonates) 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.

10.4.5 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 is also the case for glyphosate, but does not affect other pesticides such as diuron or simazine for instance. 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.

10.4.6 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 (as described in McLaughlin (2007) under the “bucket test”). Specific details on water quality requirements should be noted from the product label or by contacting the manufacturer. For herbicides, this will be under “Restraints”.

11 Conclusions and future information needs

The information provided in this Chapter has considered the parameters that needed to be reviewed in line with current international recommendations, to adopt a more conservative guideline value when necessary. However, it is important to note that the values are not necessarily appropriate for all situations. Producers should not experience economic losses from treating water to an unnecessary standard for their production system. Such an approach is in line with the revised water quality management framework which considers a cyclical process that adapts with new information and specific site water quality objectives.

Some additional work is recommended for a number of water quality parameters for irrigation and general uses and the future information needs are developed underneath.

11.1 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 guideline value can be set.

Present information on plant pathogens is limited. The nursery industry has conducted preliminary research, but further work must be undertaken before guideline 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.

11.2 Salinity and sodicity

Guidelines developed in this document for salinity and sodicity of irrigation waters incorporate a considerable body of research information. 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 guideline 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.

11.3 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 (ACL) for chromium (VI) be determined as a matter of priority.

An ACL 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 (ACL) to be determined for these two elements. However, it should be noted that an ACL 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.

11.4 Phosphorus

In its present form the P guideline derivation 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 and Weaver 1993; Sharpley 1993; Stevens et al. 1999; Kirkby et al. 1997; Nash and Murdoch 1997).

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.

11.5 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.

A standard protocol for deriving GVs and GVAs for terrestrial systems should be documented. The protocol should be based on the one currently documented for surface water (Warne et al. 2018) and is recommended for development in this section of the NWQMS guidelines (irrigation of water in terrestrial systems). Such a protocol for the terrestrial systems could be relatively easily developed by modifying the logic developed and used for deriving fresh water and marine water based on species sensitivity curves (SSD) (ANZG 2018; Warne et al. 2018). The aquatic logic focuses specifically on the freshwater environment and the protection of aquatic species. A terrestrial GVD could be derived by adapting the freshwater logic with the consideration of previous work completed for terrestrial systems for the NEPC and Melbourne Water (Heemsbergen et al. 2009). The development of a protocol for terrestrial systems would provide a method for developing GVDs (e.g., for pesticides) and determining the GVA specific for particular uses or species in a terrestrial system.

11.6 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. As well as information on plant pathogens (where water is recycled), other research/information needs to include compatibility with mixing nutrient formulations and the related salinity issues and major ion concentrations.

Another emerging issue is potential contamination of agricultural waters sources by persistent organic pollutants such as per and poly fluoroalkyl substances (PFAS). PFAS are environmentally persistent and can accumulate in plant and animal produce, either directly or indirectly via fodder consumption. PFAS are very mobile in the environment and can migrate large distances from source sites in groundwater and surface waters. At present PFAS guidelines are not included and the NEMP guideline should be referred to (HEPA 2020). Guidance in these cases must consider likely transfer rates from the irrigation water or stock water into plants and animals, with the objectives of preventing adverse impacts on human health, produce quality or animal health. Due to the range of PFAS compounds potentially present, these assessments will likely be site specific.

11.7 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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13 Glossary of terms

Term	Definition
Absorption	In chemistry: Penetration of one substance into the body of another. In biology: The act of absorbing (i.e. to take in as fluids or gases through a cell membrane). To take a substance (e.g. water, nutrients) into the body through the skin or mucous membranes or, in plants, through root hairs.
Acid-soluble metal	The concentration of the metal that passes through a 0.45 µm membrane filter after the sample is acidified to pH 1.5-2.0 with nitric acid
Acidic	Having a high hydrogen ion concentration (low pH)
Added contaminant loading limit (ACL)	The maximum contaminant loading in soil defined in gravimetric units (kg/ha). It indicates the amount of contaminant added, above which site specific risk assessment is recommended if irrigation and contaminant addition is continued
Adsorption	The taking up of one substance at the surface of another
Aeration	Any process where a substance becomes permeated with air or another gas. The term is usually applied to aqueous liquids being brought into intimate contact with air by spraying, bubbling or agitating the liquid.
Aerobic	Of organisms requiring oxygen for respiration or conditions where oxygen is available
Aesthetic	Aspects of, say, a water body, that can be considered beautiful or pleasant to the senses
'Aggressive' carbon dioxide	The amount of dissolved carbon dioxide in excess of that required to stabilise the bicarbonate ion present in water
Algae	Comparatively simple chlorophyll-bearing plants, most of which are aquatic and microscopic in size
Alkalinity	The quantitative capacity of aqueous media to react with hydroxyl ions. The equivalent sum of the bases that are titratable with strong acid. Alkalinity is a capacity factor that represents the acid-neutralising capacity of an aqueous system.

Term	Definition
Anaerobic	Conditions where oxygen is lacking; organisms not requiring oxygen for respiration
Anion	Negatively charged ion
Anionic	Characteristic behaviour or property of an ion that has a negative charge. Anions move to the anode in electrolysis.
Anode	The electrode where oxidation occurs
Anthropogenic	Produced or caused by humans
Aquatic ecosystem	Any watery environment from small to large, from pond to ocean, in which plants and animals interact with the chemical and physical features of the environment
Aquifer	An underground layer of permeable rock, sand or gravel that absorbs water and allows it free passage through pore spaces
Benthic	Referring to organisms living in or on the sediments of aquatic habitats (lakes, rivers, ponds, etc.)
Benthos	The sum total of organisms living in, or on, the sediments of aquatic habitats
Binding sites	Sites on a substrate where chemical interaction with an indicator (qv) may occur. The interaction may be electrostatic, polar, hydrogen bonding or covalent bonding.
Bioaccumulation	General term describing a process by which chemical substances are accumulated by aquatic organisms from water, either directly or through consumption of food containing the chemicals
Bioavailable	The fraction of the total of a chemical in the surrounding environment that can be taken up by organisms. The environment may include water, sediment, soil, suspended particles, and food items.
Biochemical (or biological) oxygen demand	The decrease in oxygen content in mg/L of a sample of water in the dark at a certain temperature over a certain of period of time which is brought about by the bacterial breakdown of organic matter. Usually, the decomposition has proceeded so far after 20 days that no further change occurs. The oxygen demand is

Term	Definition
	measured after 5 days (BOD5), at which time 70% of the final value has usually been reached.
Bioclogging	Clogging of irrigation infrastructure due to excessive algae or microbial growth
Biocorrosion	Corrosion caused by microorganisms through formation of biofilms on the metal surface
Biodiversity (biological diversity)	The variety of life forms, including the plants, animals and micro-organisms, the genes they contain and the ecosystems and ecological processes of which they are a part
Biofilm	Layer of materials created by microorganisms on an underwater surface
Biomass	The living weight of a plant or animal population, usually expressed on a unit area basis
Biosolids	Sewage sludge, organic residuals remaining after domestic sewage treatment
Biota	The sum total of the living organisms of any designated area
Bloom	An unusually large number of organisms per unit of water, usually algae, made up of one or a few species
Buffer	A solution containing a weak acid and its conjugate weak base, the pH of which changes only slightly on the addition of acid or alkali
Buffering capacity	A measure of the relative sensitivity of a solution to pH changes on addition of acids or base
°C	Degrees Celsius
Carcinogen	A substance that induces cancer in a living organism
Catchment	The total area draining into a river, reservoir, or other body of water
Cathode	The electrode where reduction occurs
Cation	Positively charged ion

Term	Definition
Cation exchange capacity (CEC)	A measure of a soil's ability to retain cations
Cationic	The characteristic behaviour or property of an ion with a positive charge. Cations move to the cathode in electrolysis.
Chronic	Lingering or continuing for a long time; often for periods from several weeks to years. Can be used to define either the exposure of an aquatic species or its response to an exposure (effect). Chronic exposure typically includes a biological response of relatively slow progress and long continuance, often affecting a life stage.
Colloid	Material in solution typically 1 nm-100 nm in diameter. Colloidal particles do not settle out of solution through the force of gravity. Organic colloidal matter is considered especially important in the transport of inorganic substances such as P through the soil profile.
Contaminant	Biological (e.g. bacterial and viral pathogens) and chemical (see Toxicants) introductions capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death
Corrosion	Deterioration of surfaces through erosion processes such as the conversion of metals to oxides and carbonates
Crop quality	With regard to inorganic contaminants, increased concentration of contaminant in plant tissue that while not phytotoxic, reduces the economic value of the crop due to increased residues
Cumulative	Resulting from successive additions at different times or in different ways
Cyanobacteria	A division of photosynthetic bacteria, formerly known as blue-green algae, that can produce strong toxins
DGV or Default Guideline Value	The DGV is the maximum concentration (mg/L) of contaminant in the irrigation water which can be tolerated to protect the receptors identified in the scope of the guidelines, assuming 100 years of irrigation, based on the irrigation loading assumptions previously in this document and should be used in conjunction with the ACL as a trigger for further investigation

Term	Definition
Divalent	Having a valence (combining power at atomic level) of two (e.g. calcium, Ca2+)
Dose	The quantifiable amount of a material introduced into an animal
Early detection	Measurable biological, physical or chemical response in relation to a particular stress, prior to significant adverse effects occurring on the system of interest.
ECse	The electrical conductivity of the soil saturation extract
ECs	The electrical conductivity of the soil solution at maximum field water content
EC1:5	The electrical conductivity of a 1:5 soil:water extract
Effluent	A complex waste material (e.g. liquid industrial discharge or sewage) that may be discharged into the environment
Electrical conductivity	The ability of water or soil solution to conduct an electric current
Endemic, endemism	Confined in occurrence to a local region
ESP	The exchangeable sodium content of a soil expressed as a percentage of the cation exchange capacity
Eutrophic	Abundant in nutrients and having high rates of productivity frequently resulting in oxygen depletion below the surface layer of a waterbody
Eutrophication	Enrichment of waters with nutrients, primarily phosphorus, causing abundant aquatic plant growth and often leading to seasonal deficiencies in dissolved oxygen
Evapotranspiration	The combined loss of water from a given area during a specified period of time by evaporation from the soil or water surface and by transpiration from plants
Exchangeable sodium percentage (ESP)	The sodium adsorbed onto clay mineral surfaces as a proportion of the total cation exchange capacity of those surfaces
Exposure	The amount of physical or chemical agent that reaches a target or receptor

Term	Definition
Fate	Disposition of a material in various environmental compartments (e.g. soil or sediment, water, air, biota) as a result of transport, transformation and degradation
Field capacity	The greatest amount of water that it is possible for a soil to hold in its pore spaces after excess water has drained away
Flocculation	(1) The process by which suspended colloidal or very fine particles coalesce and agglomerate into well-defined hydrated floccules of sufficient size to settle rapidly (2) The stirring of water after coagulant chemicals have been added to promote the formation of particles that will settle
Fluorosis	Chronic poisoning by fluorine
Fouling	Accumulation of material through chemical, physical or biological processes
Free carbon dioxide	The amount of dissolved carbon dioxide in excess of that required to stabilise the bicarbonate ion present in water
Gross alpha (activity)	A measure of the concentration of alpha-particle emitting radionuclides in water. This is determined by standard techniques involving the evaporation of a water sample and measurement of the alpha activity of the residue.
Gross beta (activity)	A measure of the concentration of beta-particle emitting radionuclides in water. This is determined by standard techniques involving the evaporation of a water sample and measurement of the beta activity of the residue.
Groundwater	Water stored underground in rock crevices and in the pores of geologic materials that make up the earth's crust; water that supplies springs and wells
Guideline values	These are the concentrations (or loads) of the key performance indicators measured for the ecosystem, below which there exists a low risk that adverse biological (ecological) effects will occur. They indicate a risk of impact if exceeded and should 'trigger' some action, either further ecosystem specific investigations or implementation of management/remedial actions.

Term	Definition
Guideline (water quality)	Numerical concentration limit or narrative statement recommended to support and maintain a designated water use
Habitat	The place where a population (e.g. human, animal, plant, microorganism) lives and its surroundings, both living and non-living
Half-life	Time required to reduce by one-half the concentration of a material in a medium (e.g. soil or water) or organism (e.g. fish tissue) by transport, degradation, transformation or depuration
Hardness	The concentration of all metallic cations, except those of the alkali metals, present in water. In general, hardness is a measure of the concentration of calcium and magnesium ions in water and is frequently expressed as mg/L calcium carbonate equivalent.
Hazard	The potential or capacity of a known or potential environmental contaminant to cause adverse ecological effects
Helminth	Helminths are worms; the helminths discussed in this document are human and animal parasites
Hepatotoxin	Toxic substances which adversely affect the liver
Hydrogeology	Study of subsurface waters and with related geologic aspects of surface water
Hydrolysis	(1) The formation of an acid and a base from a salt by the ionic dissociation of water (2) The decomposition of organic compounds by interaction with water
Indicator	A parameter that can be used to provide a measure of the quality of water or the condition of an ecosystem
Ingestion	The swallowing or taking in of food material
In vitro	Outside the intact organism; generally applied to experiments involving biochemical events occurring in tissue fragments or fractions in a laboratory
Ion	An electrically charged atom

Term	Definition
Langelier Saturation Index (SI)	An index based on the tendency of water to deposit or dissolve calcium carbonate. It relates the actual pH of water with the pH at which water is saturated with calcium carbonate (SI = pH - pHs).
Leachate	Water that has passed through a soil and that contains soluble material removed from that soil
Leaching	Where referred to in the salinity and sodicity section of Chapter 4, the downward movement of water and solutes below the root zone
Leaching fraction (LF)	The proportion of water applied to the surface of a soil (e.g. as irrigation or rainfall) that drains below the root zone in the soil profile
Life-cycle study	A chronic (or full chronic) study in which all the significant life stages of an organism are exposed to a test material. Generally, a life-cycle test involves an entire reproductive cycle of the organism. A partial life-cycle toxicity test includes the part of the life cycle observed to be especially sensitive to chemical exposure.
Ligand	A molecule, ion or atom that is attached to the central atom of a co-ordination compound, a chelate or other complex. May also be called complexing agent.
MATC (Maximum acceptable toxicant concentration)	The maximum concentration of a toxic substance that a receiving water may contain without causing significant harm to its productivity or uses as determined by chronic toxicity tests unsafe residues in produce for human consumption
Median	Middle value in a sequence of numbers
Metabolite	Any product of metabolism
Methylation	The introduction of methyl (CH3) groups into organic and inorganic compounds
Methyl mercury	The most common form is the cation CH3Hg+ although (CH3)2Hg also occurs. Both are extremely potent toxicants and can lead to secondary poisoning through biomagnification. They are usually formed in anoxic sediments.
Neurotoxin	Toxic substances which adversely affect the nervous system

Term	Definition
Nutrient solution	Plant growth medium providing all essential elements for plant growth in the absence of soil or other support media. Also referred to as solution culture.
Organic carbon	Generally carbon which is chemically bonded to other carbon atoms, although methane (one carbon atom only) and its derivatives are considered organic
Organism	Any living animal or plant; anything capable of carrying on life processes
Osmosis	Diffusion of a solvent through a semi-permeable membrane into a more concentrated solution, tending to equalise the concentrations on both sides of the membrane
Oxidation	The combination of oxygen with a substance, or the removal of hydrogen from it or, more generally, any reaction in which an atom loses electrons
Partition coefficient	A ratio of the equilibrium concentration of the chemical between a non-polar and polar solvent
Pathogen	An organism capable of eliciting disease symptoms in another organism
Pelagic	Term applied to organisms of the plankton and nekton which inhabit the open water of a sea or lake
Percentile	Division of a frequency distribution into one hundredths
Pesticide	A substance or mixture of substances used to kill unwanted species of plants or animals
pH	Value that represents the acidity or alkalinity of an aqueous solution. It is defined as the negative logarithm of the hydrogen ion concentration of the solution.
pH (CaCl2)	Measurement of soil pH in a 1:2.5 solution of soil:0.01M CaCl2.The CaCl2 solution is used because it has an ionic strength similar to that of soil water.
Photosynthesis	The conversion of carbon dioxide to carbohydrates in the presence of chlorophyll using light energy

Term	Definition
Physico-chemical	Refers to the physical (e.g. temperature, electrical conductivity) and chemical (e.g. concentrations of nitrate, mercury) characteristics of water
Phytoplankton	Small (often microscopic) aquatic plants suspended in water
Phytotoxicity	Toxicity of contaminants to plants
Plankton	Plants (phytoplankton) and animals (zooplankton), usually microscopic, floating in aquatic systems
Pollution	The introduction of unwanted components into waters, air or soil, usually as result of human activity; e.g. hot water in rivers, sewage in the sea, oil on land
Potable water	Water suitable, on the basis of both health and aesthetic considerations, for drinking or culinary purposes
Precipitation	(1) The formation of solid particles in a solution; generally, the settling out of small particles (2) The settling-out of water from cloud, in the form of rain, hail, snow, etc.
Radiological	Pertaining to nuclear radiation
Redox potential	An expression of the oxidising or reducing power of a solution relative to a reference potential. This potential is dependent on the nature of the substances dissolved in the water, as well as on the proportion of their oxidised and reduced components.
Risk	A statistical concept defined as the expected likelihood or probability of undesirable effects resulting from a specified exposure to known or potential environmental concentrations of a material. A material is considered safe if the risks associated with its exposure are judged to be acceptable. Estimates of risk may be expressed in absolute or relative terms. Absolute risk is the excess risk due to exposure. Relative risk is the ratio of the risk in the exposed population to the risk in the unexposed population.
Ryznar (Stability) index	Index relating the pH of water (pH) to the pH of water just saturated with calcium carbonate (pHs)
Salinity	The presence of soluble salts in or on soils or in water

Term	Definition
Sediment	Unconsolidated mineral and organic particulate material that settles to the bottom of aquatic environment
SGV (short-term guideline value)	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 DGV or considering site specific related receptors and exposure. As short-term use is typically site-specific, where available the SGV should also be used in conjunction with the ACL as a trigger for further investigation.
Sodicity	The presence of a high proportion of sodium ions relative to other cations in a soil
Sodium adsorption ratio (SAR)	The concentration of sodium relative to calcium and magnesium in the soil solution
Solution concentration	Concentration of solutes in the soil water phase. The solutes, which may be contaminants, in the soil water are generally regarded as being highly available to organisms.
Sorption	Process whereby contaminants in soils adhere to the inorganic and organic soil particles
Species	A group of organisms that resemble each other to a greater degree than members of other groups and that form a reproductively isolated group that will not produce viable offspring if bred with members of another group. This is insoluble material which resides in the water column, or is dispersed in a sample upon agitation
Standing crop	The weight of organic material that can be sampled or harvested by normal methods at any one time from a given area
Static system	An exposure system of aquatic toxicity tests in which the test chambers contain solutions of the test material or control water that are not usually changed during the test. Depending upon conditions, a static system may or may not be in equilibrium.
Steady state or dynamic equilibrium	The state at which the competing rates of uptake and elimination of a chemical within an organism or tissue are equal. An apparent steady state is reached when the concentration of a chemical in tissue remains essentially constant during a continuous exposure.

Term	Definition
Supersaturation	Refers to a solution containing more solute than equilibrium conditions will allow
Suspension	A system in which very small particles (solid, semi-solid, or liquid) are more or less uniformly dispersed in a liquid or gaseous medium. If the particles are small enough to pass through filter membranes, the system is termed a colloidal suspension. If the particles are of larger than colloidal dimensions they will tend to precipitate, if heavier than the suspending medium, or to agglomerate and rise to the surface, if lighter.
Thermotolerant coliform	Also known as faecal coliforms. In tropical and sub-tropical areas, thermotolerant coliforms may on some occasions include microorganisms of environmental rather than faecal origin.
Threshold concentration	A concentration above which some effect (or response) will be produced and below which it will not
Tolerance	The ability of an organism to withstand adverse or other environmental conditions for an indefinitely long exposure without dying
Total dissolved solids (TDS)	A measure of the inorganic salts (and organic compounds) dissolved in water
Total metal	The concentration of a metal in an unfiltered sample that is digested in strong nitric acid
Toxicant	A chemical capable of producing an adverse response (effect) in a biological system at concentrations that might be encountered in the environment, seriously injuring structure or function or producing death. Examples include pesticides, heavy metals and biotoxins (i.e. domoic acid, ciguatoxin and saxitoxins).
Toxicity	The inherent potential or capacity of a material to cause adverse effects in a living organism
Turbulence	Unorganised movement in liquids and gases resulting from eddy formation
Uptake	A process by which materials are absorbed and incorporated into a living organism

Term	Definition
Volatile	Having a low boiling or subliming pressure (a high vapour pressure)
Water quality objective	A numerical concentration limit or narrative statement that has been established to support and protect the designated uses of water at a specified site. It is based on scientific criteria or water quality guidelines but may be modified by other inputs such as social or political constraints
Watertable	The level of groundwater; the upper surface of the zone of saturation for underground water.

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