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Toxicant default guideline values for aquatic ecosystem protection

Nickel in freshwater

Technical brief

September 2026

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Summary

The default guideline values (DGVs) and associated information in this technical brief should be used in accordance with the detailed guidance provided in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* website (ANZG 2018).

Nickel is a commonly occurring natural element that is essential to some organisms. Nickel is mined and processed globally and used for many purposes, including the production of alloys, food preparation equipment, mobile telephones, batteries, medical equipment, automotive and engine components, buildings and power generation. Anthropogenic sources of nickel include motor vehicle emissions, landfills, sewage, stormwater runoff and industries such as mining.

The ANZECC and ARMCANZ (2000) nickel default guideline value (DGV) for 95% species protection in freshwater with a hardness of 30 mg/L $CaCO_3$ was 11 µg/L, based on chronic toxicity data for 7 species from 4 taxonomic groups. However, water quality parameters other than hardness, such as pH, alkalinity and dissolved organic carbon (DOC), also affect nickel bioavailability and toxicity in aquatic systems. Bioavailability models have been developed for nickel, including biotic ligand models (BLMs) and trophic-level-specific multiple linear regressions (MLRs). These models can be used to derive bioavailability-based guideline values that account for a wider range of water quality parameters compared to the ANZECC and ARMCANZ (2000) DGVs. Since 2000, more chronic toxicity data have become available, including data for tropical and temperate organisms and for Australian and/or New Zealand species, from which updated DGVs have been derived.

The DGVs reported in this technical brief used trophic-level-specific MLR bioavailability models developed by Peters et al. (2021) to account for the influence of pH, calcium, magnesium and DOC on nickel toxicity. From a database of chronic toxicity data for 44 species (20 temperate and 24 tropical species), there were data for 26 species for which test water pH, hardness (i.e. calcium and magnesium concentrations) and DOC were reported. These were used in a tiered approach to derive DGVs and bioavailability-adjusted guideline values (BAGVs) for both Australia and New Zealand.

Very high reliability DGVs for dissolved nickel in freshwater were derived from chronic toxicity data for 26 species from 9 taxonomic groups. The DGVs (shown in the table below) are based on high bioavailability reference conditions for both Australia (pH 7.7, Ca 4 mg/L, Mg 4 mg/L, DOC 2.2 mg/L) and New Zealand (pH 7.6, Ca 14 mg/L, Mg 5 mg/L, DOC 0.5 mg/L). The DGVs are slightly lower in New Zealand than in Australia due to the two country's different overall water chemistries, especially lower DOC in New Zealand. The DGVs can be adjusted to derive BAGVs based on pH and Ca, Mg and DOC concentrations in local waterbodies.

Default guideline values (DGVs) for dissolved nickel in freshwater, for Australia and New Zealand

Level of species protection (%)	DGVs for nickel in freshwater for Australia (µg/L)	DGVs for nickel in freshwater for New Zealand (µg/L)
99	0.66	0.39
95	3.4	2.3
90	6.9	5.0
80	14	11

The 95% species protection DGVs or BAGVs for dissolved nickel in freshwater are recommended for application to slightly-to-moderately disturbed ecosystems. Where ambient water data for pH, Ca, Mg or DOC are not available and a BAGV cannot be calculated, the DGV should be used in the interim. The DGVs can be used for both temperate and tropical ecosystems, and supersede the ANZECC and ARMCANZ (2000) DGVs for nickel in freshwater.

The ANZG (2018) Guidelines provide a tiered assessment framework for assessing nickel bioavailability and toxicity so that the effects of the key toxicity modifying factors on nickel bioavailability can be taken into account. The tiered assessment framework represents a decision tree to guide the user through the process of accounting for nickel bioavailability when comparing monitoring data with guideline values, and incudes a metals bioavailability tool to adjust DGVs to BAGVs based on local water chemistry.

1 Introduction

Nickel is the fifth most common element on earth and occurs extensively in the earth's crust (Nickel Institute n.d.). It primarily occurs as oxides, sulfides and silicates (Pyle and Couture 2012). Nickel ores are mined in over 23 countries, and are smelted or refined in 25 countries, including Australia. Approximately 2.5 million tonnes of nickel are produced annually, and the world demand for nickel was growing at a compound annual average rate of 5% per annum over 2010–2020 (INSG 2024).

More than two-thirds of nickel produced is used in the production of alloys (e.g. stainless steel) with other metals such as iron, copper and chromium (INSG 2024). Nickel-containing materials include food preparation equipment, mobile telephones, medical equipment, automotive and engine components, buildings, batteries and power generation (Nickel Institute n.d.). Anthropogenic sources of nickel include motor vehicle emissions, landfills, sewage, stormwater runoff and industries such as mining. Magmatic sulfide and laterite ores are naturally enriched in nickel. Nickel laterites have a fine dispersive nature and are formed by the extensive chemical and physical weathering of ultramafic rocks under tropical humid conditions (Mudd 2010). Recent estimates show that 60% of the world's nickel reserves are contained in laterite deposits (USGS n.d.). In 2018, 48% of the world's nickel production came from the tropical Asia–Pacific region, including Indonesia, New Caledonia and the Philippines (USGS n.d.).

Nickel predominantly occurs in the +2 oxidation state (i.e. Ni^{2+}) and forms stable complexes with inorganic and organic ligands (Eisler 1998; Pyle and Couture 2012). In natural freshwater, the dominant form of nickel is soluble Ni^{2+}, but other forms also exist, predominantly as complexes with sulfate and chloride (Morel and Hering 1993). Based on studies of nickel in temperate freshwater, nickel speciation depends on a variety of factors including pH, ionic strength, dissolved organic carbon (DOC), ligand type and concentration, temperature, hardness, alkalinity, other cations and the availability of solid surfaces for adsorption. In anaerobic freshwater sediments, nickel is present as nickel sulfide, which reduces its bioavailability to benthic biota. Other solid forms in sediments, such as iron and manganese oxides and organic carbon can also bind nickel (Brumbaugh et al. 2013; Schlekat et al. 2015). Once nickel has entered an aquatic system, it can be accumulated by biota (including phytoplankton and aquatic plants) or deposited in the sediment by precipitation, complexation and adsorption on clay particles, with subsequent uptake by benthic biota (Cempel and Nikel 2006).

Concentrations of dissolved nickel in unimpacted freshwater typically range from 0.1 µg/L to 0.6 µg/L (Brix et al. 2017), with a world average of 0.5 µg/L (Martin and Windom 1991). In some regions, such as New Caledonia, nickel concentrations in soils and aquatic systems are naturally enriched, but mining of lateritic nickel ores can result in the additional input of metals into freshwaters. Nickel concentrations have been measured in groundwater of up to 980 µg/L and up to 87 µg/L in urban stormwater runoff (Begum et al. 2022).

Nickel is an essential nutrient for micro-organisms and terrestrial plants, and at least 8 nickel-containing enzymes have been identified (Moreton et al. 2009). In aquatic plants and cyanobacteria, the necessity of nickel has been documented in urease and hydrogenase metabolism; however, the necessity of nickel in aquatic animals has not been confirmed (Muyssen et al. 2004).

The previous default guideline values (DGVs) for nickel in freshwater were derived from chronic toxicity data for 7 species from 4 taxonomic groups, normalised to a hardness of 30 mg/L $CaCO_3$ (ANZECC and ARMCANZ 2000). The 99% and 95% species protection DGVs at this hardness were 7 µg/L and 11 µg/L, respectively. A hardness algorithm was provided to adjust the DGVs based on the hardness of the receiving water. However, water quality parameters other than hardness, particularly pH and DOC, also affect nickel bioavailability and toxicity in aquatic systems.

Bioavailability models that capture key toxicity modifying parameters have been developed for nickel, including biotic ligand models (BLMs) (de Schamphelaere et al. 2006; Deleebeeck et al. 2008, 2009a; Peters et al. 2018) and trophic-level-specific multiple linear regressions (MLRs) (Peters et al. 2021). These models can be used to derive bioavailability-based guideline values that take into account a wider range of water quality parameters. Recently, such models have been developed or validated for Australian species and water quality conditions (Peters et al. 2018, 2021). Moreover, over the past 20 years, more chronic toxicity data for nickel in freshwaters have become available, including data for tropical and temperate organisms and for Australian and New Zealand species. The DGVs reported here are based on the ecotoxicity datasets from Stauber et al. (2021), as well as the MLR bioavailability models developed by Peters et al. (2021), and they supersede the ANZECC and ARMCANZ (2000) DGVs for nickel in freshwater. The trophic-level-specific MLRs were developed from chronic EC10 (or equivalent) data and included development and/or validation with Australian and New Zealand species.

2 Aquatic toxicology

2.1 Mechanism of toxicity

Brix et al. (2017) provided a comprehensive review of the current understanding of the mechanisms of nickel toxicity to aquatic biota, with a focus on freshwater organisms. At high concentrations, nickel and nickel compounds can cause ionoregulatory and respiratory impairment, and promote oxidative stress in freshwater invertebrates and fish. Brix et al. (2017) hypothesised that nickel may cause a range of molecular initiating events, including: disruption of calcium, magnesium and iron homeostasis; reactive oxygen species-induced oxidative damage; and an allergic-type response of respiratory epithelia. However, much of the evidence is from acute studies with prokaryotes and mammals. These events may manifest as reduced calcium availability to support formation of exoskeleton, shell and bone for growth, impaired respiration, and cytotoxicity and tumour formation, which may ultimately reduce growth and reproduction and alter energy metabolism. However, evidence for these effects on aquatic biota from chronic exposures to nickel concentrations found in the environment is limited.

For aquatic plants, in addition to oxidative damage, nickel at high concentrations may displace magnesium from the chlorophyll molecule, leading to inhibition of photosynthesis (Brix et al. 2017).

2.2 Toxicity

Over the past two decades, the body of literature on the toxicity of nickel to freshwater species has increased greatly, with the dataset spanning over 50 species from over 10 taxonomic groups and representing both tropical and temperate species.

For tropical species, reported nickel toxicity values ranged from 1.4 µg/L to 419,000 µg/L (Binet et al. 2018). However, because nickel toxicity varies with water chemistry, comparisons of species sensitivities are most useful when the ecotoxicity data are normalised to the same water chemistry. For freshwater at approximately 30 mg/L $CaCO_3$, DOC of < 1 mg/L, and pH of 6–9, a number of tropical species were sensitive to nickel, with EC10 values of 4.9–175 µg/L and EC50 values of 19–663 µg/L in chronic tests. These included several species of microalgae (*Pediastrum duplex, Pseudokirchneriella* sp., *Spermatozopsis exultans*, *Scenedesmus accuminatus*, *Desmodesmus spinosus*; EC10 values of 4.9–29 µg/L), two duckweeds (*Lemna aequinoctialis*, *Lemna minor*; EC10 values of 8.2–13 µg/L) and a cnidarian (*Hydra viridissima*; EC10 of 175 µg/L).

Tropical bacteria were less sensitive to nickel than other taxa, with an EC50 of 13,000 µg/L (Babich et al. 1986). Likewise, amphibians and fish were lower in sensitivity than most other taxa, with LC50s of 420 µg/L and 2020 µg/L, respectively (Birge 1978). One exception was the Australian endemic rainbowfish *Melanotaenia splendida splendida*, with chronic 12-d EC50 values of 39–185 µg/L, depending on water chemistry (ESA 2014). For tropical crustaceans, EC50 values varied from 250 µg/L to 410 µg/L (Wong et al. 1991; Wong and Pak 2004). Tropical snails were sensitive to nickel (LOEC < 88 µg/L) (Factor and de Chavez 2012). However, some of the tropical studies did not report water quality parameters that are known to affect nickel toxicity (e.g. DOC, pH).

For temperate species, the most sensitive species was the snail *Lymnaea stagnalis*, with an EC10 of 1.1 µg/L (Schlekat et al. 2010), followed by crustaceans *Daphnia magna* and *Ceriodaphnia dubia,* with EC10 values of 21 µg/L and 2–31 µg/L, respectively, at approximately 30 mg/L $CaCO_3$ hardness (Deleebeeck et al. 2008; Nys et al. 2016). There is some uncertainty around the sensitivity of the snail *L. stagnalis* to nickel in different studies, and differences in methods used across the studies have been suggested as possible reasons for this variation (Cremazy et al. 2020). The freshwater microalga *Pseudokirchneriella subcapitata* was also sensitive to nickel, with EC10s ranging from 25 µg/L to 365 µg/L depending on water chemistry. Similar to tropical species, other temperate invertebrates and fish were relatively insensitive to nickel.

Peters et al. (2019) suggested that there is little difference in the sensitivities of tropical versus temperate freshwater species to nickel. They compared nickel toxicity effect concentrations (typically EC10 and NOEC values), overall species sensitivity distributions (SSDs) and closely related groups of species using temperate and tropical freshwater datasets, with and without bioavailability normalisation. While direct comparison of the temperate and tropical SSDs was confounded by the lower taxa diversity, lack of amphidromous species unique to tropical environments, and insensitive species such as molluscs and fish in the tropical SSD, they showed that 95% species protection values for nickel (ranging from 3.5 µg/L to 8.6 µg/L) were similar between tropical and temperate distributions.

3 Factors affecting toxicity

The dissolved forms of nickel (e.g. the free cation and other inorganic species) are the most toxic forms of nickel to freshwater species. In general, the toxicity of nickel increases with increasing pH, decreasing hardness and decreasing DOC (Peters et al. 2018). Deleebeeck et al. (2009b) showed that for the freshwater alga *P. subcapitata*, when the pH increased from 6.45 to 7.92, nickel toxicity increased, with 72-hour EC50 values decreasing from 145 µg/L to 82 µg/L. Deleebeeck et al. (2007) also examined the individual effects of calcium, magnesium and pH on the long-term (17 d) toxicity of nickel to juvenile rainbow trout (*Oncorhynchus mykiss),* using mortality and growth as endpoints. They found that higher concentrations of H^+ (i.e. low pH), Ca^{2+} and Mg^{2+} reduced nickel toxicity, as demonstrated by increased 17-d median lethal concentrations. Similarly, binding nickel to DOC resulted in decreased bioavailability of the dissolved nickel fraction in 21-d reproduction tests with *D. magna* (Deleebeeck et al. 2008).

Variables such as alkalinity (i.e. carbonate/bicarbonate) and other major ions (e.g. sodium, potassium) may also influence nickel bioavailability. In studies with euryhaline temperate species, salinity appeared to significantly alter nickel bioavailability and toxicity, with increasing salinity decreasing nickel toxicity. Blewett et al. (2015) studied the effects of changing salinity on nickel accumulation and physiological mechanisms of nickel toxicity for the euryhaline green crab *Caricinus maenas.* The authors concluded that nickel affects ionoregulatory function in the green crab in a gill-dependent and salinity-dependent manner. Nickel accumulation was greatest at lower test salinity, likely due to the reduced competition between divalent cations and nickel for uptake. Blewett and Wood (2015) studied the effect of salinity on the toxicity of nickel to the euryhaline temperate fish *Fundulus heteroclitus*. Acute 96-hour exposures of this species to sublethal concentrations of nickel (5 mg/L, 10 mg/L and 20 mg/L) in both freshwater (0‰) and seawater (35‰), suggested that seawater (i.e. salinity) was protective against nickel toxicity, with a decrease in nickel accumulation and oxidative stress observed in fish exposed to nickel in seawater compared to fish in freshwater. Thus, salinity in freshwater is also likely to influence nickel toxicity, although quantitative relationships have not been developed.

Schlekat et al. (2010) observed chronic nickel toxicity for three invertebrates and an aquatic plant in five natural waters that varied in pH, calcium, magnesium and DOC. Nickel toxicity for the three invertebrates varied considerably among the test waters: a 14-fold variation of EC50s for *L. stagnalis*; a 3-fold variation in EC20 values for *Chironomus tentans;* and a 10-fold variation in EC20 values for *Brachionus calyciflorus*. Nickel toxicity (EC50) for *L. minor* varied by 6-fold among the test waters. The water type and combination of water quality parameters that led to the most sensitive responses varied across the species; however, a combination of high hardness (136–256 mg/L $CaCO_3$) and high DOC (7 mg/L) generally resulted in lower sensitivity to nickel.

Nickel carbonate can exist as a complex in water, with soluble $NiCO_3$ being the dominant complex. The presence of nickel carbonate can also affect several other water quality parameters such as pH, alkalinity, and the buffering capacity of the water. Nickel can also adsorb onto carbonate minerals such as hydroxyapatite, potentially reducing its bioavailability in the environment. In freshwaters in equilibrium with atmospheric carbonate dioxide, bicarbonate concentrations increase with increasing pH. Santore et al. (2021) identified potential mixture toxicity effects of nickel and

bicarbonate on the reproduction of *C. dubia* in chronic tests at high pH. Similar effects were not reported for other invertebrates.

Some of these water quality parameters have been incorporated into nickel bioavailability models such as the BLM and MLR (Deleebeeck et al. 2008, 2009a; NiPERA 2012; Brix et al. 2017; Peters et al. 2021), which enable bioavailable concentrations of nickel to be estimated or for guideline values to be adjusted based on water chemistry.

A chronic BLM for nickel has been developed and modified by Nys et al. (2016) that incorporates 10 parameters (Ca, Mg, Na, K, SO_4, Cl, pH, DOC, temperature, and alkalinity). Simplified tools, which require input of only three or four parameters but are underpinned by the full BLM, have also been developed and used to derive bioavailability-based guideline values (e.g. nickel in the European Union) (NiPERA 2012; Merrington et al. 2016). Peters et al. (2018) showed that this nickel BLM was applicable to freshwater with > 50 mg/L $CaCO_3$ hardness in Australia and New Zealand. However, for soft water, an increased competitive effect of calcium and magnesium with nickel for binding to the biotic ligand was found, so modifications were made to the BLM for its application to water with < 50 mg/L $CaCO_3$ hardness (Peters et al. 2018). For invertebrates and fish, calcium and magnesium act independently, and while both compete with nickel for uptake, they have different binding constants in recent BLMs (Santore et al. 2021). These authors found that the ratios of Ca:Mg may also affect calcium amelioration of nickel toxicity, so studies conducted at very high ratios have generally been excluded from model development.

Nickel MLRs based on relationships between toxicity (as EC10) and pH, hardness (Ca and Mg) and DOC, have recently been developed for freshwater microalgae, aquatic plants, invertebrates and fish, and validated for use in Australia and New Zealand (Peters et al. 2021). Peters et al. (2021) acknowledged that some toxicity studies did not have sufficiently detailed water chemistry information, and so not all studies could be used in MLR development. The algal MLR was developed from data from *P. subcapitata* and validated with data from 7 different algal species. In the algal MLR development, calcium was not a significant parameter. Magnesium was included because nickel can compete with magnesium in chlorophyll, making it more important than calcium for plants rather than animals (Peters et al. 2021). The aquatic plant MLR was developed using data from *L. minor* and validated with a different *Lemna* species (*L. aequinoctialis*). Three different invertebrates were used to develop the invertebrate MLRs and then validated with 4 different invertebrates. The fish MLR was developed with data from *Pimephales promelas* and validated with the local (i.e. Australian) *M. splendida splendida.* (See Table 1 in Peters et al., 2021).

Each of the MLRs included data for at least one Australian species as part of either the development or the validation of the models, while the performance of the models was tested for 10 Australian and 10 New Zealand freshwaters with varying water chemistries (Peters et al. 2021). Stauber et al. (2021) recommended using these trophic-level-specific MLRs to derive bioavailability-based guideline values for Australia and New Zealand, as they were better predictors of nickel toxicity and were easier to use than existing nickel BLMs or a pooled MLR, and were developed/validated for Australian and New Zealand species and water chemistry conditions. The trophic-level-specific MLRs are shown in Table 1.

When adjusting toxicity values to account for bioavailability, the MLRs were applied as follows: the algal MLR was applied to the chronic algae data, the aquatic plant MLR to chronic duckweed data, the invertebrate MLR to chronic invertebrate data and the fish MLR to chronic fish data.

More details on the development, validation and comparison of the various models are discussed by Peters et al. (2021) and Stauber et al. (2021).

Table 1 Trophic-level-specific multiple linear regressions (MLRs)

Trophic level	MLR[a]
Algae	$Log_e(EC10) = Sensitivity^{b} + 0.28 \times log_e[DOC] + 0.50 \times log_e[Mg] - 0.20 \times pH$
Aquatic plants	$Log_e(EC10) = Sensitivity + 0.96 \times log_e[DOC] - 1.44 \times pH$
Invertebrates	$Log_e(EC10) = Sensitivity + 2.09 \times log_e[DOC] + 0.19 \times log_e[Ca] + 0.40 \times log_e[Mg] - 0.40 \times pH - 0.24 \times log_e[DOC] \times pH$
Fish	$Log_e(EC10) = Sensitivity - 1.05 \times log_e[DOC] + 3.55 \times log_e[Mg] - 0.07 \times pH + 0.19 \times log_e[DOC] \times pH - 0.42 \times log_e[Mg] \times pH$

Note: Table sourced from Peters et al. (2021).

[a] The parameters in the MLR equations have been rounded to 2 decimal places.

[b] Sensitivity is a species-specific coefficient calculated as the difference between the observed (experimental) value (log_eEC10_{expt}) and the MLR predicted value (log_eEC10_{pred}) for each species at each water chemistry combination tested (see Table 2).

Each MLR includes a sensitivity coefficient calculated as the difference between the observed (experimental) value (log_eEC10_{expt}) and the MLR predicted value (log_eEC10_{pred}) for each species at each water chemistry combination tested. The geometric mean of the calculated sensitivity coefficients for an individual species across all the available tests for that species is used as the species-specific sensitivity value in the MLR. These are given for each species in Table 2.

Table 2 Species-specific sensitivity coefficients to use in MLRs to normalise ecotoxicity data

Trophic level	Species	Sensitivity coefficient[b]
Algae	*Navicula pelliculosa*	4.50
	Chlorella sp. (Kakadu isolate) [a]	5.97
	Chlorella sp. 12 (PNG isolate)[a]	3.91
	Monoraphidium arcuatum[a]	5.40
	Nannochloropsis sp.[a]	3.53
	Pediastrum duplex[a]	5.69
	Pseudokirchneriella subcapitata	5.01
Aquatic plants	*Lemna aequinoctialis* [a]	12.5
	Lemna minor[a]	13.6
Invertebrates	*Brachionus calyciflorus*[a]	7.71
	Hyalella azteca	5.82
	Alona affinis	4.14
	Ceriodaphnia dubia	3.25
	Ceriodaphnia pulchella	5.48
	Ceriodaphnia quadrangula	5.18
	Daphnia longispina	6.54
	Daphnia magna	5.14
	Peracantha truncata	5.75
	Simocephalus serrulatus	6.21
	Simocephalus vetulus	5.54
	Lymnaea stagnalis	2.04
	Hydra viridissima[a]	6.38
	Chironomus tentans	6.98
	Clistoronia magnifica	6.43
Fish	*Melanotaenia splendida splendida*[a, c]	3.35
	Pimephales promelas	5.83

Note: Data sourced from Peters et al. (2021), with corrections to the sensitivity coefficients for *Monoraphidium arcuatum*, *Alona affinis*, *Ceriodaphnia dubia*, *Daphnia magna* and *Lymnaea stagnalis* (see Response to public submission documents for details).

[a] Tropical species.

[b] Rounded to 3 significant figures.

[c] *Melanotaenia splendida splendida* is a subspecies of the *Melanotaenia splendida* complex.

4 Default guideline value derivation

The DGVs for dissolved nickel in freshwater were derived in accordance with the method described in Warne et al. (2018), except that the ssdtools package (V 2.3.0, Thorley et al. 2025) for R (R Core Team 2024) was used instead of the Burrlioz 2.0 software, as recommended by Warne et al. (2025). Additional steps that were required to derive bioavailability-based DGVs for nickel are also captured in this Section.

4.1 Collation and screening of toxicity data

The derivation of the DGVs for nickel in freshwater relied heavily on the nickel freshwater toxicity dataset compiled and assessed by Stauber et al. (2021). Due to the large size of the nickel toxicity dataset, it was initially divided into temperate species data and tropical species data. Temperate species were isolated from temperate regions and/or had a natural geographical distribution outside of the Tropics of Cancer and Capricorn, and toxicity tests were conducted at temperatures < 25°C. Tropical species had a natural geographical distribution between the Tropic of Cancer and the Tropic of Capricorn, and the toxicity tests were conducted at ≥ 25°C. Chronic nickel toxicity data that passed the quality assessment and screening process were available for 20 temperate species from 6 taxonomic groups and for 24 tropical species from 6 taxonomic groups. A comparison of the temperate and tropical datasets by Stauber et al. (2021) showed a large overlap in the toxicity values and found that the two datasets resulted in similar protective concentrations (i.e., for 80%, 90%, 95% and 99% species protection) with overlapping confidence intervals. Consequently, there was insufficient justification to derive climatic zone-specific DGVs, and the two datasets were combined for the DGV derivation. The combined dataset included more taxonomic groups (Stauber et al. 2021), which ultimately improved the confidence in the DGVs.

The combined dataset comprised chronic toxicity data for 44 species from 9 taxonomic groups. The toxicity data included LC/EC10, NOEC, LOEC and LC/EC50 values. However, of the 44 species in the combined dataset, only 26 species from 9 taxonomic groups (diatom, green microalgae, duckweed, rotifer, crustaceans, gastropod, cnidarian, insects and fish) had sufficient test water chemistry data to apply the MLR bioavailability corrections. Of the 190 individual tests used in the nickel ecotoxicity database for normalisation, there were 156 EC10s, 29 NOECs, 2 LOECs and 3 LC/EC50s. The 2 LOECs, 2 LC50s and 1 EC50 were converted to EC10 equivalents by dividing each value by 2.5, 5 and 5 respectively, in accordance with Warne et al. (2018). Twenty one NOEC values were included because they were the only values for those species, including *P. promelas* (fathead minnow), *Hyalella azteca* and *Clistoronia magnifica*. Two of the three LC/EC50s were also the only values for *Alonas affinis*. Inclusion of all these “non preferred” data enabled either increased taxa coverage and/or many different water chemistries for all species for the MLR normalisations. These benefits were considered to outweigh the recommendation of Warne et al. (2018) to not use chronic NOEC, LOEC or EC/LC50 data if sufficient preferred negligible effect data (e.g. LC/EC10s) are available. A summary of all the quality assessed nickel toxicity data, together with the water chemistry for each test for each species used to develop, validate and apply the nickel MLRs, is provided as supporting documentation. Details of the data quality assessment are also provided as supporting documentation.

4.2 Selection of default guideline values that represent a high bioavailability condition

Adopting a bioavailability-based approach for the nickel freshwater DGVs enables the adjustment of the DGVs to account for DOC, pH, calcium and magnesium in local waters. This requires the DGVs to be set at a conservative, high bioavailability water chemistry condition, as the DGVs can then be conservatively applied to situations where there are no local water chemistry data. The process for doing this for nickel is described in detail by Gadd et al. (2026a, b) and, thus, is only summarised below.

The high bioavailability condition upon which the DGVs were based was determined following an extensive collation and analysis of data from freshwaters around Australia and New Zealand. A matched dataset with pH, DOC, calcium and magnesium measured in the same sample and within the applicability range for the nickel MLR models included 10,607 values for Australia and 6,436 values for New Zealand. This included data where hardness was measured, but not calcium and magnesium (or they were not reported) and these two variables were estimated from the reported hardness. Full details of the water chemistry datasets are provided in Gadd et al. (2026a, b).

The collated water chemistry datasets, together with the nickel freshwater ecotoxicity dataset and trophic level-specific MLRs described in Section 3, were used in the ssdtools package with model averaging to construct probability distributions of nickel concentrations that were protective of 99%, 95%, 90% and 80% of species in each individual water sample. These bioavailability-adjusted protective concentrations (BAPCs) were calculated for each available water chemistry sample for all sites in each state and mainland territory in Australia and each region in New Zealand. Only those data within the boundary conditions of the bioavailability models were used to calculate the BAPCs. The 99%, 95%, 90% and 80% species protection level BAPCs corresponding to the water chemistry conditions at the weighted 5th percentile of the BAPCs in the most sensitive state or territory in both Australia (Australian Capital Territory) and the most sensitive region in New Zealand (Bay of Plenty) were selected as the DGVs. Full details of the process are provided in Gadd et al. (2026b).

The above approach aims to ensure that the DGVs are protective of 95% of waters in the most sensitive region in each country and, therefore, more than 95% of waters in each country. The associated, high bioavailability water chemistry condition is termed the ‘reference condition’, and these are as follows for Australia and New Zealand:

- Australian reference condition – pH 7.7, 4 mg/L Ca, 4 mg/L Mg and 2.2 mg/L DOC
- New Zealand reference condition – pH 7.6, 14 mg/L Ca, 5 mg/L Mg and 0.5 mg/L DOC

The final chronic toxicity dataset described in Section 4.1, normalised to the Australian and New Zealand reference conditions, is shown in Table 3.

Table 3 Summary of chronic toxicity values normalised to the Australian and New Zealand reference conditions, used to derive the default guideline values for dissolved nickel in freshwater

Taxonomic group	Species	Life stage	Duration	Toxicity endpoint	Toxicity value normalised to Australian reference condition[a] (µg/L)	Toxicity value normalised to NZ reference condition[b] (µg/L)
Diatom	*Navicula pelliculosa*	–	72 h	Growth rate	47	36
Green microalga	*Chlorella* sp. (Kakadu isolate)[a]	–	72 h	Growth rate	203	155
	Chlorella sp. 12 (PNG isolate)[a]	–	72 h	Growth rate	26	20
	Monoraphidium arcuatum[c]	–	72 h	Growth rate	115	88
	Nannochloropsis sp.[c]	–	72 h	Growth rate	18	14
	Pediastrum duplex[c]	–	72 h	Growth rate	154	118
	Pseudokirchneriella subcapitata	–	72 h	Growth rate	78	60
Macrophyte (duckweed)	*Lemna aequinoctialis*[c]	–	96 h	Growth rate	8.6	2.3
	Lemna minor[c]	–	7 d	Growth rate	25	6.6
Rotifer (rotifer)	*Brachionus calyciflorus*[c]	Neonates	48 h	Population growth rate	277	270
Crustacean (amphipod)	*Hyalella azteca*	7–8 d old	14 d	Survival	41	41
Crustacean (cladoceran)	*Alona affinis*	Neonates	16 d	Survival	7.8	7.6
	Ceriodaphnia dubia	Neonates	7 d	Reproduction	3.2	3.1
	Ceriodaphnia pulchella	Neonates	17 d	Reproduction	30	29
	Ceriodaphnia quadrangula	Neonates	17 d	Reproduction	22	21
	Daphnia longispina	Neonates	21 d	Reproduction	85	83
	Daphnia magna	Neonates	5 broods	Reproduction	21	21
	Peracantha truncata	Neonates	17 d	Reproduction	39	38
	Simocephalus serrulatus	Neonates	17 d	Reproduction	61	60
	Simocephalus vetulus	Neonates	21 d	Reproduction	31	31
Gastropod	*Lymnaea stagnalis*	< 24 h old	30 d	Growth rate	1.0	0.9
Cnidarian	*Hydra viridissima*[c]	–	96 h	Population growth	73	71
Insect (chironomid)	*Chironomus tentans*	Larvae	10 d	Growth	132	129
Insect (caddisfly)	*Clistoronia magnifica*	Larvae	19 w	Survival	77	75
Fish	*Melanotaenia splendida splendida*[c]	Embryos	12 d	Hatching	35	22
	Pimephales promelas	Swim-up fry	17 d	Survival	421	267

[a] Australian reference condition – pH 7.7, 4 mg/L Ca, 4 mg/L Mg and 2.2 mg/L DOC.

[b] New Zealand reference condition – pH 7.6, 14 mg/L Ca, 5 mg/L Mg and 0.5 mg/L DOC.

[c] Tropical species; the remaining species are temperate species.

Bimodal or multimodal toxicity was determined to be unlikely, based on the available knowledge on the mechanism of chronic toxicity of nickel to aquatic biota. Although nickel may have an additional mode of action in plants and algae (i.e., displacement of magnesium from the chlorophyll molecule), toxicity values for these phototrophic groups were spread evenly across the dataset. The data were not indicative of bimodality based on histogram statistics (not shown here), a bimodality coefficient of 0.35 and 0.42 for the log-transformed datasets for Australia and NZ (less than the threshold criterion for bimodality of 0.55), and the spread of taxa in the dataset (Figure 1; normalised for the New Zealand reference condition only).

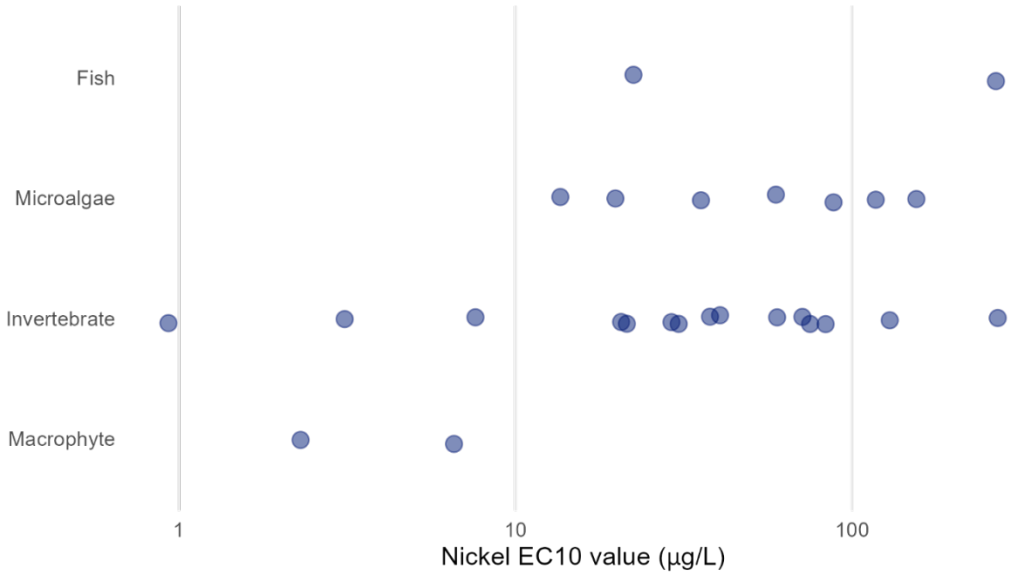


Figure 1 Comparison of dissolved nickel toxicity between taxonomic groups. Nickel toxicity data normalised to the New Zealand reference condition of pH 7.6, 14 mg/L Ca, 5 mg/L Mg and 0.5 mg/L DOC.

4.3 Species sensitivity distributions

The SSDs of the chronic toxicity data for nickel for 26 species in Australian and New Zealand freshwaters at the respective reference conditions are shown in Figure 2 and Figure 3. The SSDs were plotted using the ssdtools software (v 2.3.0) (Thorley et al. 2025) using the function to fit the 6 distributions recommended for the ANZG toxicant DGVs (Gamma, log-Gumbel, log-logistic, log-normal, log-normal/log-normal and Weibull) (Fox et al. 2024), with the model averaging method used to calculate the DGVs. The models were judged to provide a good fit to the data for both the Australian and New Zealand reference conditions. Further details on the SSD fitting process for the Australian and New Zealand SSDs are provided in Appendix A and Appendix B, respectively.

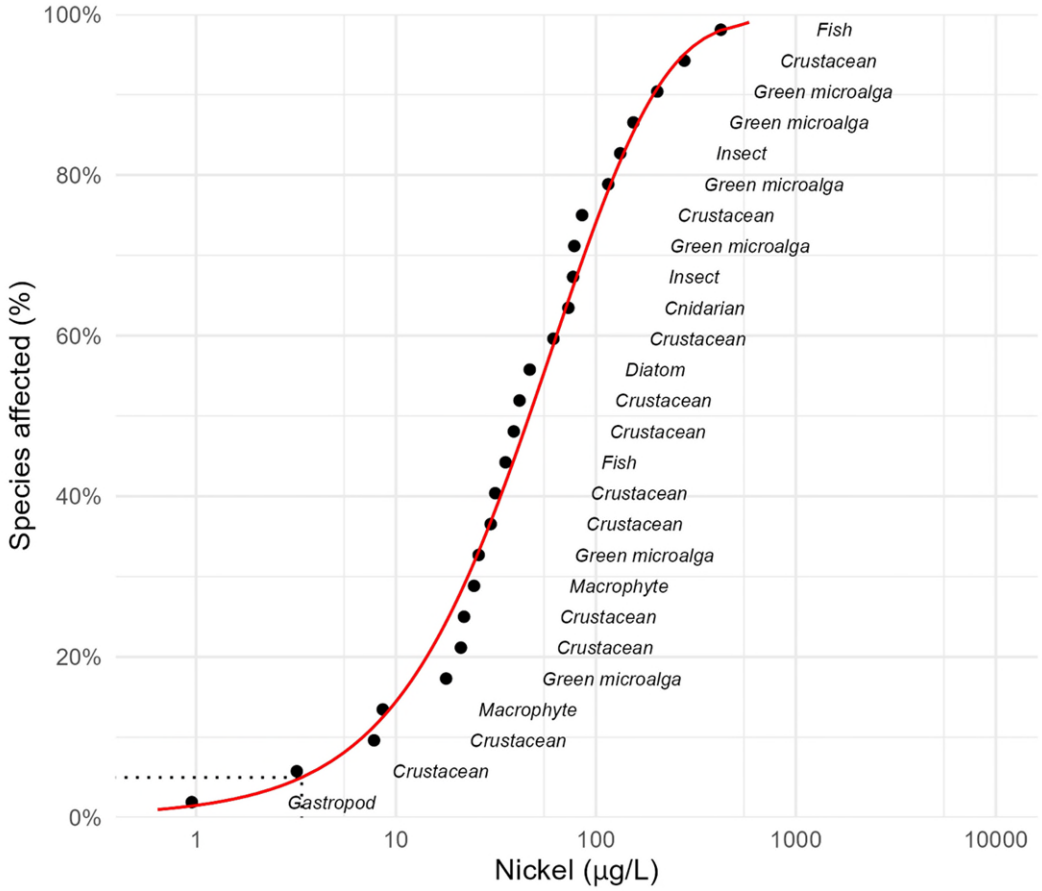


Figure 2 Species sensitivity distribution for dissolved nickel in freshwater at the reference condition for Australia (pH 7.7, 4 mg/L Ca, 4 mg/L Mg and 2.2 mg/L DOC)

Dotted line indicates the nickel concentration at which there is 95% species protection and 5% of species are potentially affected (3.4 µg/L).

4.4 Default guideline values

It is important that the DGVs and associated information in this technical brief are used in accordance with the detailed guidance provided on the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* website (ANZG 2018).

The DGVs for 99%, 95%, 90% and 80% species protection at the reference conditions for Australia and New Zealand are shown in Table 4. The nickel DGVs are slightly lower for New Zealand than for Australia, which reflects the different overall water chemistries, especially the lower DOC in New Zealand. The DGVs supersede the ANZECC and ARMCANZ (2000) DGVs for nickel in freshwater and should be applied in accordance with the tiered assessment framework for nickel in freshwater (see Section 4.5).

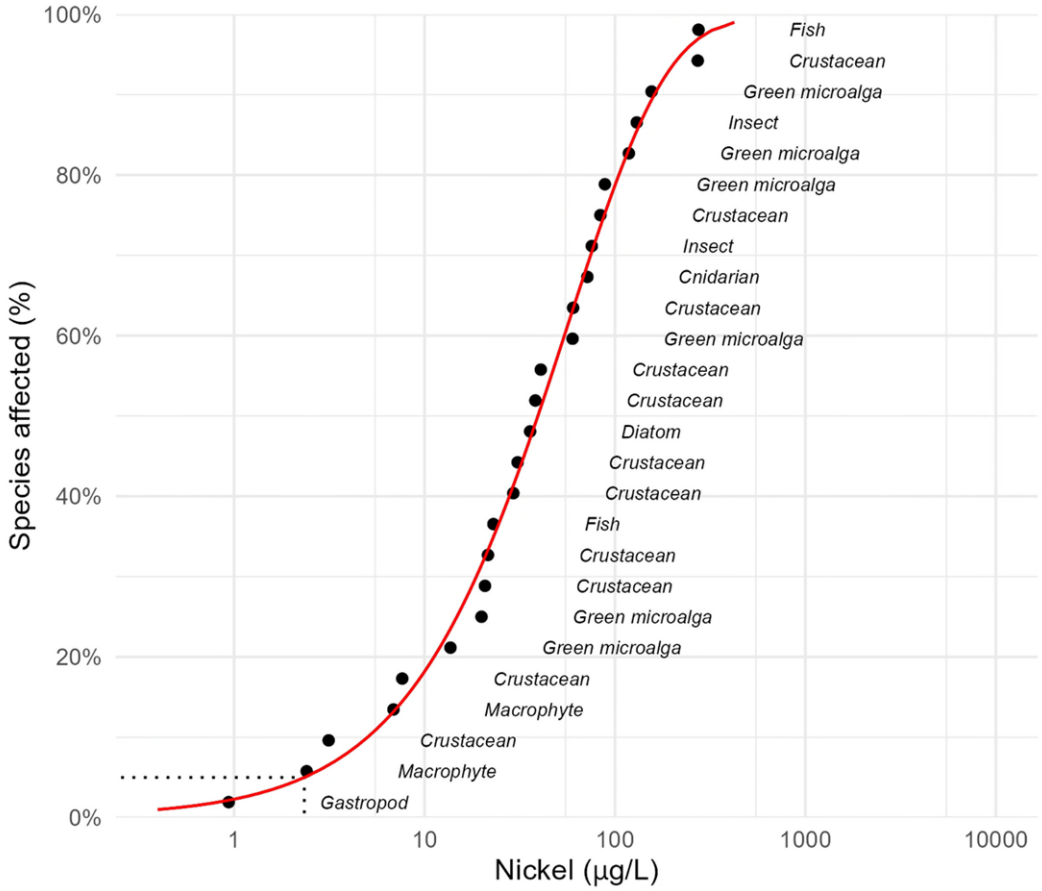


Figure 3 Species sensitivity distribution for dissolved nickel in freshwater at the reference condition for New Zealand (pH 7.6, 14 mg/L Ca, 5 mg/L Mg and 0.5 mg/L DOC)

Dotted line indicates the nickel concentration at which there is 95% species protection and 5% of species are potentially affected (2.3 µg/L).

Table 4 Default guideline values for dissolved nickel in freshwater for Australian and New Zealand reference conditions, with very high reliability.

Level of species protection (%)	DGV for dissolved nickel in freshwater for Australia (µg/L)[ab]	DGV for dissolved nickel in freshwater for New Zealand (µg/L)[ac]
99	0.66	0.39
95	3.4	2.3
90	6.9	5.0
80	14	11

[a] DGVs were derived using the ssdtools package (V 2.3.0) in R (Thorley et al. 2025) and have been rounded to 2 significant figures. They can be adjusted for different pH, calcium, magnesium and DOC concentrations.

[b] Australian reference condition – pH 7.7, 4 mg/L Ca, 4 mg/L Mg and 2.2 mg/L DOC.

[c] New Zealand reference condition – pH 7.6, 14 mg/L Ca, 5 mg/L Mg and 0.5 mg/L DOC.

The DGVs are expected to be protective of at least 95% of waters in both Australia and New Zealand. The DGVs apply to dissolved nickel (operationally defined as the < 0.45 µm filtered measurement) and can be applied in waters with pH 6.0–8.0, calcium 3–88 mg/L and magnesium 3–72 mg/L (equivalent to a hardness range of 20–516 mg $CaCO_3$/L), and DOC 0.5–17 mg/L. These applicability ranges are based on the ranges where all the MLRs for algae, invertebrates, and vertebrates were applicable. The aquatic plant MLR was excluded from this assessment because it has a particularly narrow applicability range and applied only to two of the 26 species included in the SSD. BAGVs can only be calculated for water chemistries within these ranges. The 95% species-protection level DGVs and BAGVs are recommended for application to slightly-to-moderately disturbed ecosystems. Where the local water chemistry (pH, calcium, magnesium, DOC) is outside the above-mentioned applicability ranges, the DGVs and BAGVs may not be applicable, and site-specific guideline values may be needed (Stauber et al. 2026). ANZG (2018) provides guidance on the derivation of site-specific guideline values, and local jurisdictions should always be consulted on appropriate methods.

The 99% species protection DGVs may be close to analytical detection limits routinely used by commercial laboratories (e.g. 1 µg/L); however, under most water chemistries, BAGVs are likely to be above typical limits of detection. ANZG (2018) provides advice on data analysis where the DGV is less than the limit of detection.

Under the tiered assessment framework (see Section 4.5), if the dissolved nickel concentration exceeds the relevant DGV, the DGV can be adjusted to a BAGV based on the pH, calcium, magnesium and DOC concentrations of ambient water, using the ANZG metals bioavailability tool.

4.5 Use of the default guideline values in the tiered assessment framework

The DGVs should be applied in accordance with the tiered assessment framework for dissolved nickel in freshwater. The tiered assessment framework, depicted as a flow chart in Appendix C, represents a decision tree to guide the user through the process of accounting for nickel bioavailability when comparing monitoring data with guideline values, and incudes a metals bioavailability tool to adjust DGVs to BAGVs based on local water chemistry. Details about the tiered assessment framework can be found in Stauber et al. (2026) and ANZG (2018). Briefly, the tiered assessment framework comprises a screening level at Tier 1, where the measured dissolved nickel concentration in a sample should be compared with the DGV at the required level of species protection. If the DGV is exceeded at Tier 1, then at Tier 2, metal bioavailability for the local water quality conditions can be taken into account. This can be done either by (i) calculating a nickel BAGV based on the sample's pH, calcium, magnesium and DOC concentrations and comparing the dissolved nickel concentration in the sample to the BAGV, or (ii) adjusting the nickel concentration in the sample to represent the bioavailable fraction, and comparing this concentration with the DGV. If the sample still exceeds the relevant guideline value, then, at Tier 3, additional lines of evidence are required to determine the implications of the exceedance, which may include toxicity testing or bioavailability modelling.

4.6 Reliability classification

The dissolved nickel in freshwater DGVs for both Australia and New Zealand have a very high reliability classification (Warne et al. 2018) based on the outcomes for the following three criteria:

- sample size—26 (preferred)

- type of toxicity data—chronic, including EC10, NOEC and converted EC50, LC50 and LOEC
- SSD model fit—good.

Glossary

Term	Definition
Acute toxicity	A lethal or adverse sublethal effect that occurs as the result of a short exposure period to a chemical relative to the organism's life span.
BAGV (bioavailability-adjusted guideline value)	A default guideline value that has been adjusted to account for one or more specific toxicity modifying factors using one or more established models.
BAPC (bioavailability-adjusted protective concentration)	A protective concentration value derived from a species sensitivity distribution that has been adjusted to account for one or more specific toxicity modifying factors using one or more established models. These values were derived to identify and select the DGVs (based on ensuring at least 95% of waters were protected).
Benthic	Refers to organisms living in or on the sediments of aquatic habitats (e.g. lakes, rivers, ponds).
BLM (biotic ligand model)	Models that consider the effect of multiple parameters, including water hardness, pH, and dissolved organic carbon, to model the bioavailability of metals.
Chronic toxicity	A lethal or sublethal adverse effect that occurs after exposure to a chemical for a period of time that is a substantial portion of the organism's life span or an adverse effect on a sensitive early life stage.
DGV (default guideline value)	A guideline value recommended for generic application in the absence of a more specific guideline value (e.g. site-specific guideline value) in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Formerly known as 'trigger values'.
DOC (dissolved organic carbon)	Refers specifically to the mass of carbon in the dissolved organic material, operationally defined as that which can pass through a filter with a pore size typically between 0.22 and 0.7 µm.
EC50 (median effective concentration)	The concentration of a substance in water or sediment that is estimated to produce a 50% change in the response being measured or a certain effect in 50% of the test organisms relative to the control response, under specified conditions.
Endpoint	The specific response of an organism that is measured in a toxicity test (e.g. mortality, growth, a particular biomarker).
Euryhaline	Describes organisms that are capable of osmo-regulating over a wide range of salinities.
Guideline value	A measurable quantity (e.g. concentration) or condition of an indicator for a specific community value below which (or above which, in the case of stressors such as pH, dissolved oxygen and many biodiversity responses) there is considered to be a low risk of unacceptable effects occurring to that community value. Guideline values for more than one indicator should be used simultaneously in a multiple lines of evidence approach.
LC50 (median lethal concentration)	The concentration of a substance in water or sediment that is estimated to be lethal to 50% of a group of test organisms, relative to the control response, under specified conditions.
LOEC (lowest observed effect concentration)	The lowest concentration of a material used in a toxicity test that has a statistically significant adverse effect on the exposed population of test organisms as compared with the controls.
Macrophyte	A member of the macroscopic plant life of an area, especially of a body of water; large aquatic plant.
MLR (multiple linear regression)	Regression equations where the y parameter is modelled using 2 or more x parameters.

Term	Definition
NOEC (no observed effect concentration)	The highest concentration of a toxicant used in a toxicity test that does not have a statistically significant adverse effect on the exposed population of test organisms as compared with the controls.
Reference condition	The high metal bioavailability water chemistry condition at the default guideline values. At least 95% of waters in Australia and New Zealand would have lower bioavailability conditions than the reference condition. It is based on an extensive collation and analysis of data from freshwaters around Australia and New Zealand.
Species (biological)	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.
SSD (species sensitivity distribution)	A method that plots the cumulative frequency of species' sensitivities to a toxicant and fits a statistical distribution to the data. From the distribution, the concentration that should theoretically protect a selected percentage of species can be determined.
Toxicity	The inherent potential or capacity of a material to cause adverse effects in a living organism.
Toxicity test	The means by which the toxicity of a chemical or other test material is determined. A toxicity test is used to measure the degree of response produced by exposure to a specific level of stimulus (or concentration of chemical) for a specified test period.

Appendix A: Details of the ssdtools SSD fitting process for nickel in freshwater – Australian reference condition

The SSD software package ssdtools (V 2.3.0) uses a model averaging approach to generating a SSD (as described by Fox et al. 2024). Figure A1 shows the full set of default distributions fitted to the nickel freshwater toxicity dataset at the Australian water chemistry reference condition (pH 7.7, 4 mg/L Ca, 4 mg/L Mg and 2.2 mg/L DOC), while Table A1 provides the relevant goodness of fit and weighting estimates for each of the distributions. The final model-averaged SSD is shown in Figure 2 of the main report.

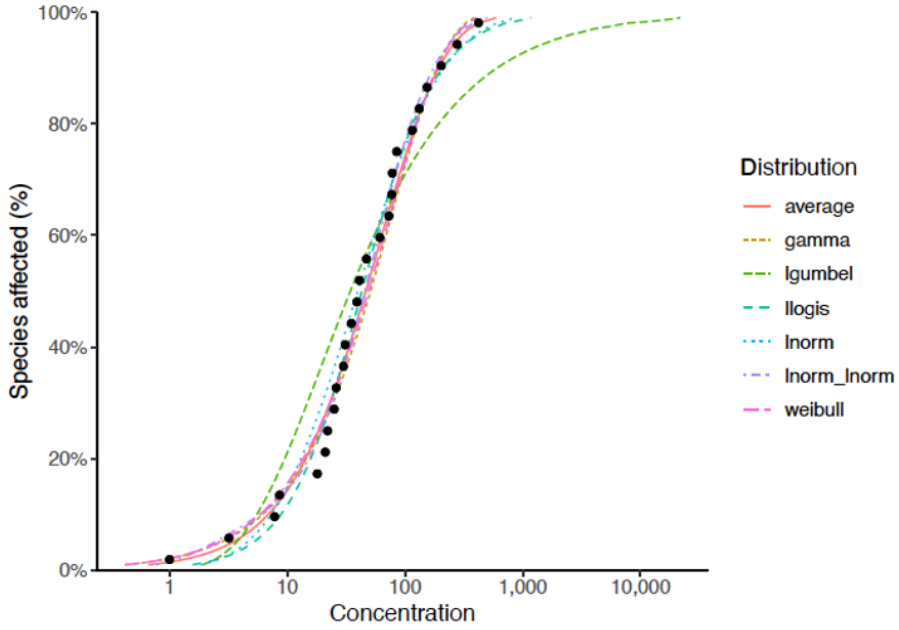


Figure A1 SSDs for six default distributions used to model the nickel freshwater toxicity dataset at the Australian water chemistry reference condition

Table A1 Goodness of fit and weighting estimates for six default distributions used in model averaging for the nickel freshwater toxicity dataset at the Australian water chemistry reference condition

Distribution	Akaike's Information Criterion (AIC)	Corrected Akaike's Information Criterion (AICc)	Delta	Weight
Weibull	282	283	0.000	0.325
Gamma	282	283	0.275	0.284
Log logistic	283	283	0.673	0.232
Log normal	284	284	1.580	0.148
Log normal - Log normal	287	290	7.040	0.010
Log gumbel	293	294	11.200	0.001

Appendix B: Details of the ssdtools SSD fitting process for nickel in freshwater – New Zealand reference condition

The SSD software package ssdtools (V 2.3.0) uses a model averaging approach to generating a SSD (as described by Fox et al. 2024). Figure B1 shows the full set of default distributions fitted to the nickel freshwater toxicity dataset at the New Zealand water chemistry reference condition (pH 7.6, 14 mg/L Ca, 5 mg/L Mg and 0.5 mg/L DOC), while Table B1 provides the relevant goodness of fit and weighting estimates for each of the distributions. The final model-averaged SSD is shown in Figure 3 of the main report.

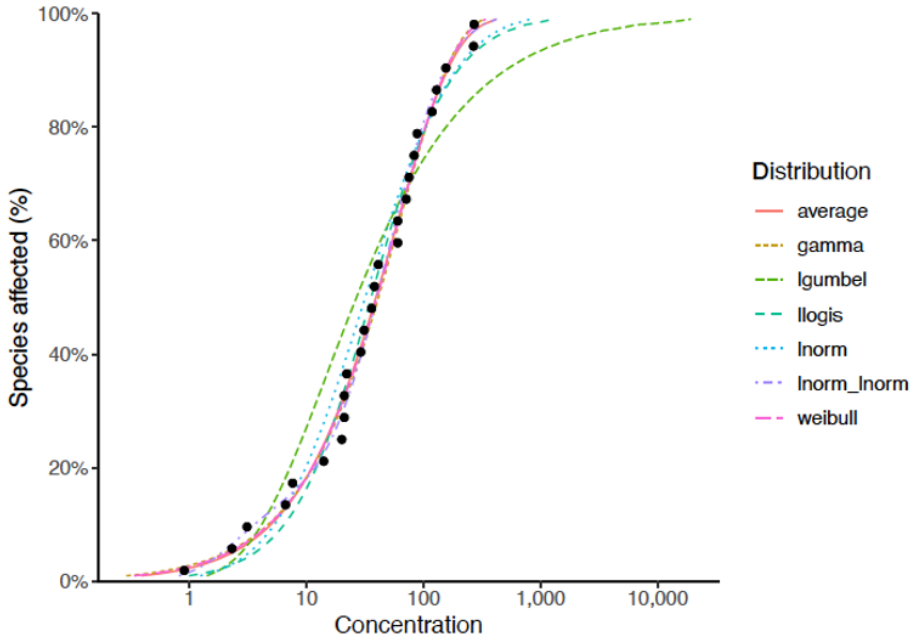


Figure B1 SSDs for six default distributions used to model the nickel freshwater toxicity dataset at the New Zealand water chemistry reference condition

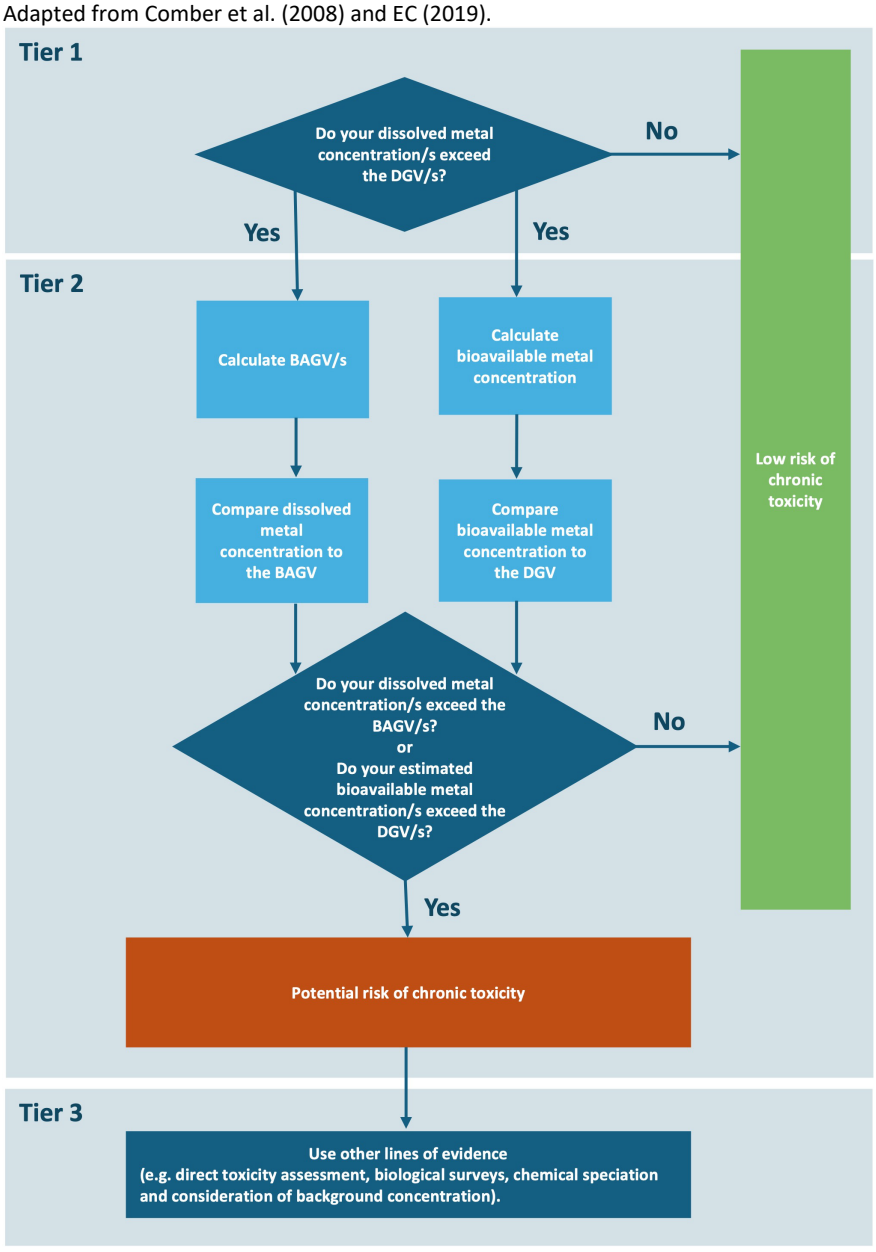
Table B1 Goodness of fit and weighting estimates for six default distributions used in model averaging for the nickel freshwater toxicity dataset at the New Zealand water chemistry reference condition

Distribution	Akaike's Information Criterion (AIC)	Corrected Akaike's Information Criterion (AICc)	Delta	Weight
Weibull	272	272	0.000	0.408
Gamma	272	272	0.077	0.392
Log logistic	275	275	2.890	0.096
Log normal	275	275	2.960	0.093
Log normal - Log normal	277	280	7.380	0.010
Log gumbel	283	284	11.700	0.001

Appendix C: Flow chart of tiered assessment framework for bioavailability-based default guideline values

Figure C1 depicts the tiered assessment framework for bioavailability-based default guideline values, including those for dissolved nickel in freshwater. Refer to Section 4.5, Stauber et al. (2026) and ANZG (2018) for further details of how the framework is implemented.

Figure C1 The tiered assessment framework for bioavailability-based default guideline values
Adapted from Comber et al. (2008) and EC (2019).



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