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Australian & New Zealand GUIDELINES FOR FRESH & MARINE WATER QUALITY

Toxicant default guideline values for aquatic ecosystem protection

Cobalt in marine water

Technical brief

June 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* (ANZG 2018).

Cobalt is a transition group metal that is an essential micronutrient for many aquatic and terrestrial organisms. It enters the environment naturally through chemical weathering, leaching, runoff, or volcanic eruptions. The primary anthropogenic sources of cobalt in the environment are mining and mineral processing, municipal and industrial wastewater and sludges, fertiliser additives, emissions from alloy production, and the combustion of fossil fuels.

Cobalt generally occurs in 2 oxidation states: cobalt(II) (Co^{2+}) or cobalt(III) (Co^{3+}) (Blust 2011), with cobalt(II) being the most stable in natural waters, mainly as free Co^{2+}. The concentration of cobalt in marine waters is largely dependent on chelation, adsorption, precipitation, and pH. As with many metals, the solubility and concentration of cobalt generally increases with decreasing pH (Nishikawa and Tominaga 2001). Under oxidising and moderately reducing conditions, the dominant cobalt aqueous species predicted in seawater is Co^{2+} (Krupka and Serne 2002), with lesser amounts of $CoCl^{+}$ and $CoSO_4$. Cobalt can also form strong complexes with organic ligands (Krupka and Serne 2002), such as naturally occurring humic and fulvic acids (Krupka and Serne 2002). These complexes may alter cobalt solubility, reduce adsorption, and thus affect mobility in aqueous environments (Krupka and Serne 2002). Cobalt can co-precipitate with, or be adsorbed onto, manganese and iron oxides, thus reducing mobility and concentration in freshwater systems (Hem et al. 1985).

In aquatic environments, most cobalt is associated with sediments or suspended solids (ATSDR 2024; Nagpal 2004). The adsorption of cobalt to sediments and soils is largely controlled by the presence of iron and manganese oxides, clay minerals and organic matter (ATSDR 2024; Krupka and Serne 2002). Adsorbed phases of cobalt are likely to be retained in deposited sediments (Qian et al. 1998). In deep sediments where water is anoxic and hydrogen sulfide is present, some mobilisation of cobalt from sediment may occur, most likely due to the formation of bisulfides and polysulfides (ATSDR 2024).

Cobalt has a low potential to bioaccumulate in aquatic biota and does not biomagnify within the food chain. The previous default guideline values (DGVs) for cobalt in marine waters were 'high reliability' values based on 8 chronic and converted acute toxicity data points, with the 99% and 95% species protection level values being 0.005 µg/L and 1 µg Co/L, respectively (ANZECC and ARMCANZ 2000). As the dataset included converted acute data, the DGVs were classified to be of moderate reliability (Warne et al. 2018). The 99% species protection level DGV was also well below natural cobalt concentrations in coastal waters (McAlpine et al. 2004). Since ANZECC and ARMCANZ (2000), more data on cobalt chronic toxicity have been published, which has enabled the derivation of improved DGVs.

Very high reliability DGVs for cobalt in marine waters were derived using chronic (EC10, NOEC, NEC) toxicity data for 24 species from 10 taxonomic groups (cyanobacterium, diatom, green microalga, red macroalga, crustacean, annelid, bivalve, echinoderm, cnidarian, and fish), including data from several Australasian species. The DGVs for 99%, 95%, 90% and 80% species protection are 1.6 µg/L, 6.1 µg/L, 12 µg/L and 27 µg/L, respectively. The DGVs relate to dissolved cobalt, operationally defined as the

< 0.45 µm-filtered measurement. The 95% species protection level for cobalt in marine water (6.1 µg/L) is recommended for adoption in the assessment of slightly-to-moderately disturbed ecosystems. If there are ecologically significant and highly sensitive macroalgal communities present, they may not be protected by the 95% species protection DGV, in which case the 99% species protection DGV is recommended.

Background cobalt concentrations in Australian waters (Jahan and Stresov 2017; McAlpine et al. 2004) suggest that it is unlikely that the derived DGVs are exceeded under natural conditions outside of areas with natural mineralisation.

1 Introduction

Cobalt (Co) is a transition group metal. Pure metallic cobalt does not occur in the natural environment; however, the metal is present in different mineral phases (Blust 2011). Cobalt is relatively rare, being the 33rd most abundant element on Earth and comprising 0.0025% of the planet's crust by weight (WHO 2006). Known cobalt minerals include a diversity of sulfides, arsenides, sulfo-arsenides, arsenates, and selenides. Cobalt also exists as oxide and carbonate minerals (Krupka and Serne 2002).

Cobalt is used in many alloys (including steel), magnets, rechargeable batteries (e.g. lithium-ion), as an industrial catalyst, as a drying agent and as a pigment in paints, ceramics, and glass (Diamond et al. 1992; ATSDR 2024; BGS 2019). The main anthropogenic sources of non-radioactive cobalt include metalliferous and coal mining, mineral processing, alloy production, fossil fuel combustion, fertiliser application and inappropriate disposal of municipal or industrial wastes (Diamond et al. 1992; Reinardy et al. 2013; NPI 2017). Radioactive cobalt forms (i.e. ^{60}Co) are released through radioactive waste (from energy and industrial/medical applications) and nuclear weapons detonation and testing (Qian et al. 1998; ATSDR 2024; Blust 2011; Reinardy et al. 2013), of which there are few recorded incidents within Australia or New Zealand (Voigt and Fesenko 2009). The anthropogenic radioisotope ^{60}Co is an important source of gamma radiation with industrial and medical applications (Blust 2011).

Australia is currently the fifth largest supplier of cobalt, producing approximately 6,000 tonnes in 2015 (USGS 2016). Cobalt is mainly mined with other mineral commodities (BGS 2019; Reinardy et al. 2013). In New Zealand, it is produced as a by- or co-product during the exploitation of many nickel-sulfide deposits (Christie and Brathwaite 2016).

Cobalt generally occurs in 2 oxidation states: cobalt(II) (Co^{2+}) or cobalt(III) (Co^{3+}) (Blust 2011). Under most geochemical conditions the cobaltous ion (Co^{2+}) is the stable oxidation state in water, and the ion and its various complexes are the principal dissolved forms (Krupka and Serne 2002; Nagpal 2004). Cobalt(III) is a strong oxidising agent, is not thermodynamically stable, and has a low solubility, but can be stabilised in organic complexes in the presence of dissolved oxygen (Baars and Croot 2015). Cobalt (II) organic complexes in coastal waters are strong and thought to involve phytoplankton exudates (Ellwood and van den Berg 2001; Wong et al. 2022) but have been difficult to study because of low dissolved cobalt concentrations.

Natural cycling of cobalt mainly occurs through chemical weathering, leaching, erosion, seawater spray, runoff, or volcanic eruptions (WHO 2006; Gál et al. 2008). Anthropogenic activities, particularly mining and processing of coal and metalliferous deposits, have been linked to the alteration of cobalt cycling and the elevation of cobalt concentrations in the receiving environment (WHO 2006; Gál et al. 2008; NPI 2017).

Cobalt is found naturally in the environment and is an essential element for algae, cyanobacteria and other nitrogen-fixing organisms (Saito and Moffett 2001), and for animals and humans, as a component of vitamin B12 (ATSDR 2024). Cobalt concentrations in oceans lie between 0.002 µg/L in the north central Pacific Ocean and 0.39 µg/L in the Indian Ocean (Nagpal 2004). Concentrations are lower in the photic zone than at depth due to cobalt uptake by phytoplankton (Saito and Moffett 2001). Although concentrations increase with depth due to the absence of phytoplankton, they can

subsequently decrease in deeper waters due to oxidation by manganese oxidising bacteria (Moffett and Ho 1996). Higher concentrations (2–5 µg/L) are found in estuarine and coastal waters (Florence and Batley 1980). Jahan and Stresov (2017) reported maximum cobalt concentrations of 1–2 µg/L in 2 ports in New South Wales (Port Yamba, Port Eden) based on 4 months of data. However, in most samples collected from these 2 ports and 4 others (Port Jackson, Port Botany, Port Kembla, Port Newcastle), cobalt concentrations were below the detection limit (Jahan and Stresov 2017). McAlpine et al. (2004) also reported low background concentrations (≤ 0.008–0.014 µg/L) for coastal marine waters of Western Australia. These studies were limited in assessment and longer-term sampling programs would typically be required to properly characterise water quality.

As an essential element, cobalt is known to be taken up by organisms. Bioaccumulation factors (BAFs) for marine fish have been reported to be ~100–4,000 (ATSDR 2024) and 100–1000 (Karlsson et al. 2002), which translate to log BAF values of 2–3.6. These values indicate that cobalt does not have a high potential to bioaccumulate and, therefore, that secondary poisoning is not an issue. Supporting this conclusion, Chouvelon et al. (2019) examined trace metal bioaccumulation and trophic transfer in a marine food web, reporting that cobalt concentrations progressively decreased along the food web from the smallest phytoplankton up to fish. Consistent with this finding, ATSDR (2024) reported that cobalt does not biomagnify within the food chain.

The previous Australian and New Zealand default guideline values (DGVs) for cobalt in marine waters (ANZECC and ARMCANZ, 2000) were 'high reliability' values. They were based on chronic and converted acute toxicity data for 8 species from 4 taxonomic groups, with the 99% and 95% species protection level values being 0.005 µg/L and 1 µg/L, respectively (ANZECC and ARMCANZ 2000). There are now more data on cobalt chronic toxicity to marine species available, including numerous Australasian species. This new data has enabled the derivation of improved DGVs compared to those in ANZECC and ARMCANZ (2000). The updated cobalt marine DGVs reported here supersede the ANZECC and ARMCANZ (2000) DGVs.

2 Aquatic toxicology

2.1 Mechanism of toxicity

The mode of action of cobalt toxicity is thought to be related to various biochemical mechanisms. Cobalt stimulates the generation of a range of reactive oxygen species in the presence of water and biological chelators (e.g. glutathione) (Leonard et al. 1998; Zou et al. 2001). Reactive oxygen species can cause cell injury by damaging lipids, proteins, and DNA (Mates 2000) and can induce programmed cell death (Zou et al. 2001). This is supported by several studies that have demonstrated changes in indices of oxidative stress in fish and mammal tissues after exposure to soluble cobalt (e.g. Lewis et al. 1991; Kubrak et al. 2011). The genotoxicity of cobalt is thought to be caused by the combined effects of DNA damage and mechanisms that inhibit DNA repair (Kasten et al. 1997; Asmuß et al. 2000; Lison et al. 2001; Figgitt et al. 2010; Reinardy et al. 2013). Chronic cobalt exposure in fish has similar pathology to chronic hypoxia in mammals, including increased red blood cell counts and cardio-ventilatory dysfunction (Morelli et al. 1994; Majmudar and Burleson 2006; Atamanalp et al. 2010). As with other cationic metals, Co^{2+} can bind to negative sites on the gill surface (Richards and Playle 1998), potentially inhibiting ionic regulation, gas transfer and acid-base balance. Cobalt is also a calcium antagonist, blocking cell membrane calcium channels and thereby

interfering with cellular metabolism (Henquin and Lambert 1975; Yamatani et al. 1998). Csatorday et al. (1984) suggested that cobalt competes with iron for the active site on the chlorophyll molecule, with the subsequent accumulation of protoporphyrin inhibiting chlorophyll synthesis and hindering algal growth.

2.2 Toxicity

Cobalt is an important element in marine ecosystems, with vitamin B12 (cobalamin) being essential for the growth of some phytoplankton, particularly cyanobacteria (Saito and Moffett 2002; Panzeca et al. 2008). There is generally little overlap between background concentrations of cobalt in seawater (< 0.5 µg/L) and concentrations that cause effects in aquatic organisms.

Saili et al. (2021) conducted chronic toxicity tests on 10 marine organisms comprising *Champia parvula* (red macroalga), *Neanthes arenaceodentata* (polychaete), *Americamysis bahia* (mysid/opossum shrimp), *Skeletonema costatum* (diatom), *Dendraster excentricus* (sand dollar), *Mytilus galloprovincialis* (mussel), *Strongylocentrotus purpuratus* (purple sea urchin), *Crassostrea gigas* (oyster; now known as *Magallana gigas*), *Dunaliella tertiolecta* (marine flagellate or green microalga), and *Cyprinodon variegatus* (sheepshead minnow). All were static tests, except for the polychaete, which used static renewal, and the mysid and sheepshead minnow tests, which were both flow-through tests.

The most sensitive chronic effect of cobalt resulted in an EC10 of 1.23 µg/L for 48-hour reproduction of the red macroalga *C. parvula* (Saili et al. 2021). This species is at least an order of magnitude more sensitive than any other species tested. The least sensitive effect was an EC10 of 31,800 µg/L for 28-days biomass of fertilised eggs of the sheepshead minnow *C. variegatus* (Saili et al. 2021). Tests for this fish involved concentrations close to the solubility limit of cobalt in seawater (Krauskopf 1956; Cosovic et al. 1982).

The chronic effect of cobalt on the Australian sea anemone, *Exaiptasia diaphana* (formerly known as *Aiptasia pulchella*), resulted in an EC10 of 42 µg/L for asexual reproduction via the formation of lacerates (Howe et al. 2014). Diatoms and green algae generally showed low sensitivities to cobalt, with EC10 values ranging from 170 µg/L for *Odentella mobiliensis* and 370 µg/L for *Cosinodiscus centralis* (Karthikeyan et al. 2019) to 590 µg/L for *S. costatum* and 12,000 µg/L for *D. tertiolecta* (Saili et al. 2021), and EC50 values between 3,800 µg/L and 12,000 µg/L for *S. costatum, Nitzschia closterium*, *Chaetoceros curvicetus*, *Platymonas subcordiforus* (Rosko and Rachlin 1975, Chen et al. 2018). Copepods also exhibited low sensitivity to cobalt with EC10s of 220 µg/L and 470 µg/L for *Acartia danae* and *Oithona similis* respectively (Karthikeyan et al. 2019).

Markich (2021) determined no effect concentrations (NECs) for the 48-hour embryo larval development of 10 Australian bivalve species comprising 2 oysters, one mussel, 2 cockles, one scallop and 4 clams. NEC values ranged between 18.0 µg/L and 75.3 µg/L. It is worth noting the large difference in sensitivity between the 2 bivalves (*C. gigas* and *M. galloprovincialis*) tested by Saili et al. (2021), for which the EC10s were 2,760 µg/L and 1,160 µg/L, respectively, and those tested by Markich (2021). The reasons for this large difference are unclear.

There is generally little overlap between background concentrations of cobalt in seawater (< 0.5 µg/L) and concentrations that cause effects in aquatic organisms.

3 Factors affecting toxicity

The dissolved form of cobalt, particularly the free cation Co^{2+}, is the most toxic form of cobalt. Increases in salinity result in lower concentrations of Co^{2+}, due to complexation with chloride (Byrne 2002). Changes in cobalt speciation with differing pH and salinity affect the toxicity to aquatic organisms. Typically, metal toxicity decreases as salinity increases (Hall and Anderson 1995). There is little information available on the influence of temperature on cobalt toxicity. A single study by Moleiro et al. (2022) has investigated the influence of both increasing temperature and cobalt exposure to the mussel (*M. galloprovincialis*) and found that the combination of both stressors can be synergistic, with regard to oxidative damage and neurotoxicity.

In marine waters, the key factor affecting cobalt bioavailability is dissolved organic carbon (DOC), since pH and salinity do not vary substantially among sites. In estuarine and coastal waters, over 60% of cobalt is complexed with dissolved organic matter (Zhang et al. 1990). While organic cobalt complexes are generally soluble (WHO 2006), the bioavailability, and hence toxicity, of complexed cobalt are currently not clearly understood. The uptake of organic colloids by marine organisms has been demonstrated (e.g. Guo et al. 2001), indicating the potential for colloidal bound cobalt to be bioavailable. Watanabe et al. (2017) reported confounding results about the influence of humic substances on the toxicity of cobalt to the freshwater species *Ceriodaphnia dubia*, with high humic concentrations leading to almost 100% mortality while lower concentrations increased the effects of cobalt on reproduction. More research on the effects of DOC on cobalt toxicity is needed.

4 Default guideline value derivation

The DGVs were derived in accordance with the method described in Warne et al. (2018) and using Burrlioz 2.0 software.

4.1 Toxicity data used in derivation

The derivation of DGVs for cobalt involved searching the international literature for published studies that assessed the toxicity of cobalt in marine water. Data from these studies were then collated, screened, and analysed according to the methods described by Warne et al. (2018) and the rationale described by Batley et al. (2018).

Cobalt toxicity data for marine species were obtained from the scientific literature by conducting searches using the US EPA Ecotox database (US EPA n.d.), the Australasian Ecotoxicology Database (Markich et al. 2002; Langdon et al. 2009) and the Google Scholar search engine. The data used to derive the ANZECC and ARMCANZ (2000) DGVs for cobalt were also reviewed. There was sufficient chronic toxicity data to derive DGVs using the species sensitivity distribution (SSD) approach, therefore, data from acute toxicity studies were not considered. All accepted data scored greater than 77% using the quality assessment scheme outlined by Warne et al. (2018), with high quality data scoring ≥80%.

A total of 24 chronic toxicity values (10 NECs, 10 EC10s, 4 NOECs) for 24 species from 10 taxonomic groups (cyanobacterium, diatom, green microalga, red macroalga, annelid, crustacean, bivalve,

echinoderm, cnidarian, fish) were obtained (Appendix A). Although Warne et al. (2018) recommends excluding NOEC values if there are more than 8 NEC/EC10 values, the 4 NOECs were included in the final dataset. This increased the species and taxonomic representation (especially through the inclusion of crustaceans) and resulted in an improved fit of the SSD.

Saili et al. (2021) reported both EC10 and EC20 data for cobalt for 10 marine species, with the EC10 data being used for the derivation of the DGVs reported here. Of the data published by Saili et al. (2021), the EC10 for the mysid *A. bahia* was excluded from the derivation of the DGVs because the salinity of the test solutions (i.e. 19–20 ppt) was below the recommended lower limit of 25 ppt for marine waters (Warne et al. 2018).

Measured concentration data were not available for 3 diatoms (*N. closterium*, *C. curvicetus*, *S. costatum*) and a green alga (*P. subcordiforus*) from the studies by Chen et al. (2018) and Rosko and Rachlin (1975). These values were initially considered acceptable due to Saili et al. (2021) reporting that all measured concentrations were close to nominal values, and losses are generally less in saline waters. However, both studies reported only EC50 values and, as such, the data were not used in the final dataset.

The inclusion of the cobalt toxicity data for 10 bivalve species published by Markich (2021) resulted in 11 of the 24 species in the final dataset being bivalves (i.e. 46% of the total species). Appendix B presents an assessment of the effect of inclusion of the Markich (2021) bivalve species on the SSD and associated protective concentration values. This assessment provided no strong argument to exclude the Markich (2021) bivalve data on the basis that they resulted in an over-representation of this taxonomic group. Consequently, the Markich (2021) data were included in the final dataset used to derive the DGVs.

An EC10 value for larval abnormalities of 2,760 µg/L was obtained for the oyster *C. gigas* (Saili et al. 2021). This species has been renamed *M. gigas* for which Markich (2021) reported a NEC value of 18 µg/L. Because the measures of toxicity were different (i.e. EC10 and NEC), it was inappropriate to average the values as a geometric mean, so the lower value was used in the SSD.

Given the above considerations, the final dataset used for the derivation of the DGVs was based on cobalt chronic toxicity data for 24 species from 10 taxonomic groups, and included 10 NECs, 10 EC10s and 4 NOECs. An assessment of modality of the final dataset confirmed that it was unimodal (Appendix C). Therefore, the full dataset was used to construct the SSD and derive the DGVs. A summary of the toxicity data (consolidated to one value per species) used to derive the DGVs for cobalt in marine water is provided in Table 1. Further details for all the data used to derive the DGVs, and associated test conditions, are presented in Table A1 , Appendix A. Details of the data quality assessment and the data that passed the quality assessment are provided as supporting information.

Table 1 Summary of chronic toxicity data used to derive the default guideline values for cobalt in marine water

Taxonomic group	Species	Life stage	Duration (d)	Toxicity measure [a] (endpoint)	Toxicity value (µg/L)
Cyanobacterium	*Cyanobium* sp.	Exponential growth phase	3	EC10 (growth rate)	300
Diatom	*Skeletonema costatum*	Exponential growth phase	3	EC10 (growth rate)	590
Diatom	*Odentella mobiliensis*	Exponential growth phase	4	NOEC (growth rate)	170
Diatom	*Coscinodiscus centralis*	Exponential growth phase	4	NOEC (growth rate)	370
Green microalga	*Dunaliella tertiolecta*	Exponential growth phase	4	EC10 (growth rate)	12,000
Red macroalga	*Champia parvula*	Adult branch	2	EC10 (growth rate)	1.23
Crustacean (copepod)	*Oithona similis*	Adult	4	NOEC (mortality)	470
Crustacean (copepod)	*Acartia danae*	Adult	4	NOEC (mortality)	220
Annelid	*Neanthes arenaceodentata*	Juvenile (24 h)	113	EC10 (reproduction)	206
Echinoderm (sand dollar)	*Dendraster excentricus*	Embryo	3	EC10 (abnormalities)	968
Echinoderm (sea urchin)	*Strongylocentrotus purpuratus*	Embryo	4	EC10 (abnormalities)	1,790
Bivalve (cockle)	*Anadara trapezia*	Embryo	2	NEC (abnormalities)	32.7
Bivalve (cockle)	*Fulvia tenuicostata*	Embryo	2	NEC (abnormalities)	43.3
Bivalve (clam)	*Barnea australasiae*	Embryo	2	NEC (abnormalities)	53.2
Bivalve (clam)	*Hiatula alba*	Embryo	2	NEC (abnormalities)	46.3
Bivalve (clam)	*Irus crenatus*	Embryo	2	NEC (abnormalities)	75.3
Bivalve (clam)	*Spisula trigonella*	Embryo	2	NEC (abnormalities)	64.4
Bivalve (oyster)	*Magallana gigas* [b]	Embryo	2	NEC (abnormalities)	18.0
Bivalve (oyster)	*Saccostrea glomerata*	Embryo	2	NEC (abnormalities)	18.1
Bivalve (scallop)	*Scaeochlamys livida*	Embryo	2	NEC (abnormalities)	24.7
Bivalve (mussel)	*Mytilus galloprovincialis*	Embryo	2	EC10 (abnormalities)	1,660
Bivalve (mussel)	*Xenostrobus securis*	Embryo	2	NEC (abnormalities)	22.8
Cnidarian	*Exaiptasia diaphana* [c]	Lacerate	28	EC10 (asexual reproduction)	42
Fish	*Cyprinodon variegatus*	Freshly fertilised egg	28	EC10 (biomass)	31,800

[a] The measure of toxicity being estimated/determined: EC10: 10% effect concentration; NEC: no effect concentration; NOEC: no observed effect concentration.

[b] Formerly known as *Crassostrea gigas*.

[c] Formerly known as *Aiptasia pulchella.*

4.2 Species sensitivity distribution

The cumulative frequency (species sensitivity) distribution (SSD) of the 24 marine cobalt chronic toxicity values reported in Table 1 is shown in Figure 1. The SSD was plotted using the Burrlioz 2.0 software. The model was judged to provide a good fit to the data (Figure 1).

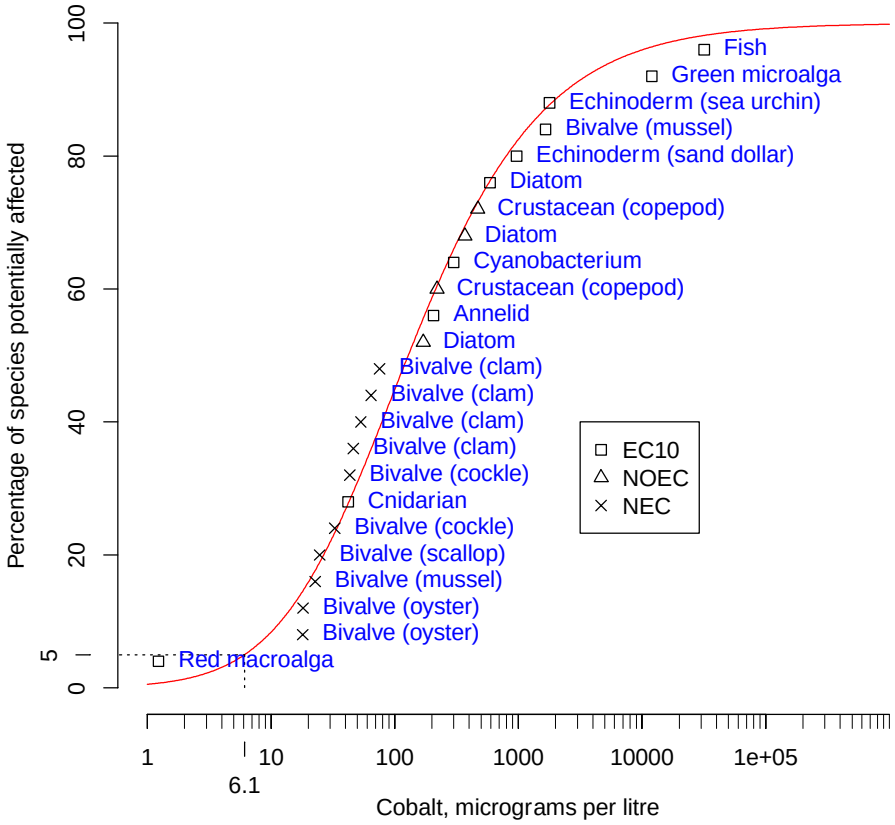


Figure 1 Species sensitivity distribution (from Burrlioz 2.0) for cobalt in marine water

4.3 Default guideline values

It is important that the DGVs (Table 2) and associated information in this technical brief are used in accordance with the detailed guidance provided in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* (ANZG 2018).

The DGVs for 99%, 95%, 90% and 80% species protection are shown in Table 2. The DGVs relate to dissolved cobalt, operationally defined as the < 0.45 µm-filtered measurement. The 95% species protection DGV for cobalt in marine water (6.1 µg/L) is recommended for application to slightly-to-moderate disturbed marine ecosystems. The DGVs are based on toxicity data generated at pH 7.2–8.4, temperature 16–26°C and salinity 25–38 ppt, where reported. The DGVs are expected to be applicable to DOC concentrations considered average for seawater (median 0.5 mg/L, Thurman 1985). At present, there are insufficient empirical data to enable the development of DGVs that incorporate the effects of toxicity modifying factors. Where it is suspected that local water quality may affect the toxicity of cobalt, site-specific guideline values may need to be considered. Guidance on the application of DGVs to estuarine waters is provided in ANZG (2018).

Table 2 Toxicant default guideline values for cobalt in marine water with very high reliability [a]

Level of species protection (%)	DGV for cobalt in marine water (µg/L)
99	1.6
95	6.1
90	12
80	27

[a] The DGVs were derived using the Burrlioz 2.0 software and rounded to 2 significant figures.

Saili et al. (2021) derived a 95% species protection value for cobalt of 7.1 µg/L using largely EC10 data fitted to a log-normal distribution, although the fit of the SSD to the data was poor. This value is similar to the 95% species protection DGV of 6.1 µg/L derived here. Notably, the 95% species protection DGV does not theoretically protect the red macroalgal species (*C. parvula*); however, this species appears to be an exceptionally sensitive species relative to other species tested. For a dataset of 24 toxicity values, it is not unexpected for at least one of the toxicity values to lie below the 95% species protection DGV. However, if there are ecologically significant and highly sensitive macroalgal communities present, they may not be protected by the 95% species protection DGV, in which case the 99% species protection DGV is recommended.

4.4 Reliability classification

The DGVs for cobalt in marine water have a very high reliability classification (Warne et al. 2018) based on the outcomes for the following 3 criteria:

- Sample size — 24 (preferred)
- Type of toxicity data — chronic EC10, NEC and NOEC values
- SSD model fit — good (Burr Type III)

Glossary and acronyms

Term	Definition
Acute toxicity	A lethal or adverse sub-lethal effect that occurs as the result of a short (relative to the organism's life span) exposure to a chemical. Refer to Warne et al. (2018) for examples of acute exposures.
ANZECC	Australian and New Zealand Environment and Conservation Council.
ARMCANZ	Agricultural and Resource Management Council of Australia and New Zealand.
Chronic toxicity	A lethal or sub-lethal 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 sub-lethal effect on a sensitive early life stage. Refer to Warne et al. (2018) for examples of chronic exposures.
Default guideline value (DGV)	A guideline value recommended for generic application in the absence of a more specific guideline value (e.g. a site-specific value), in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Formerly known as 'trigger values'.
ECx	The concentration of a substance in water or sediment that is estimated to produce an x% change in the response being measured or a certain effect in x% of the test organisms, under specified conditions.
Endpoint	The specific response of an organism that is measured in a toxicity test (e.g. mortality, growth, a particular biomarker).
Guideline value (GV)	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 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.
Humic substances	Organic substances only partially broken down that occur in water mainly in a colloidal state. Humic acids are large-molecule organic acids that dissolve in water.
NEC (no-effect concentration)	The maximum concentration of a substance that causes no adverse effect in a target organism, based on a threshold parameter in a concentration–response model.
NOEC (no-observed-effect concentration)	The highest concentration of a substance used in a toxicity test that does not have a statistically significant ($p > 0.05$) adverse effect on the exposed population of test organisms as compared with the controls.
Site-specific	Relating to something that is confined to, or valid for, a particular place. Site-specific guideline values are relevant to the specific location or conditions that are the focus of a given assessment.
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.
Species sensitivity distribution (SSD)	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: toxicity data that passed the screening and quality assessment and were used to derive the default guideline values

Table A1 Summary of the chronic toxicity data that passed the screening and quality assurance processes and were used to derive the cobalt in marine water default guideline values

Taxonomic group	Species	Life stage	Exposure duration (days)	Toxicity measure (Test Endpoint)	Test medium	Temperature (°C)	Salinity (ppt)	pH	Concentration [a] (µg/L)	Reference
Cyanobacterium	*Cyanobium* sp.	Exponential growth phase	3	EC10 (growth rate)	Seawater	25	33	8.0	300	Alquezar and Anastasi 2013
Diatom	*Odentella mobiliensis*	Exponential growth phase	4	NOEC (growth rate)	Seawater	24	29	—	170	Karthikeyan et al. 2019
Diatom	*Coscinodiscus centralis*	Exponential growth phase	4	NOEC (growth rate)	Seawater	24	29	—	370	Karthikeyan et al. 2019
Diatom	*Skeletonema costatum*	Exponential growth phase	3	EC10 (growth rate)	Seawater	19–20	38	8.0	590	Saili et al. 2021
Green microalga	*Dunaliella tertiolecta*	Exponential growth phase	4	EC10 (growth rate)	Synthetic seawater	19–20	30	8.1	12,000	Saili et al. 2021
Red macroalga	*Champia parvula*	Adult branch	2	EC10 (growth rate)	Seawater	23–24	31	8.3	1.23	Saili et al. 2021
Crustacean (copepod)	*Oithona similis*	Adult	4	NOEC (mortality)	Seawater	24	29	—	470	Karthikeyan et al. 2019
Crustacean (copepod)	*Acartia danae*	Adult	4	NOEC (mortality)	Seawater	24	29	—	220	Karthikeyan et al. 2019

Taxonomic group	Species	Life stage	Exposure duration (days)	Toxicity measure (Test Endpoint)	Test medium	Temperature (°C)	Salinity (ppt)	pH	Concentration [a] (µg/L)	Reference
Annelid	*Neanthes arenaceodentata*	Post-emergent (< 24 h) juvenile	113	EC10 (reproduction)	Synthetic seawater	19–20	29–34	7.7–8.4	206	Saili et al. 2021
Bivalve (oyster)	*Magallana gigas* [b]	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	18.0	Markich 2021
Bivalve (oyster)	*Saccostrea glomerata*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	18.1	Markich 2021
Bivalve (mussel)	*Mytilus galloprovincialis*	Embryo	2	EC10 (proportion abnormal)	Seawater	16–17	29–30	8.0–8.1	1,660	Saili et al. 2021
Bivalve (mussel)	*Xenostrobus securis*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	22.8	Markich 2021
Bivalve (cockle)	*Anadara trapezia*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	32.7	Markich 2021
Bivalve (cockle)	*Fulvia tenuicostata*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	43.3	Markich 2021
Bivalve (clam)	*Barnea australasiae*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	53.2	Markich 2021
Bivalve (clam)	*Hiatula alba*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	46.3	Markich 2021
Bivalve (clam)	*Irus crenatus*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	75.3	Markich 2021
Bivalve (clam)	*Spisula trigonella*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	64.4	Markich 2021
Bivalve (scallop)	*Scaeochlamys livida*	Embryo	2	NEC (abnormalities)	Seawater	21	30	7.9	24.7	Markich 2021
Echinoderm (sand dollar)	*Dendraster excentricus*	Embryo	3	EC10 (proportion abnormal)	Seawater	15–16	30–31	8.2	968	Saili et al. 2021
Echinoderm	*Strongylocentrotus purpuratus*	Embryo	4	EC10 (proportion abnormal)	Seawater	15–16	30–31	8.0	1,790	Saili et al. 2021
Cnidarian	*Exaiptasia diaphana* [c]	Lacerate	28	EC10 (asexual reproduction)	Seawater	25–26	34	—	42	Howe et al. 2014
Fish	*Cyprinodon variegatus*	Freshly fertilised eggs	28	EC10 (biomass)	Synthetic seawater	25–26	27–29	7.2–8.0	31,800	Saili et al. 2021

[a] All values reported here used in the SSD as is.

[b] Formerly known as *Crassostrea gigas*.

[c] Formerly known as *Aiptasia pulchella.*

Appendix B: comparison of cobalt toxicity datasets with and without Markich (2021) bivalve data

The inclusion of the cobalt toxicity data for 10 bivalve species published by Markich (2021) resulted in almost half of the dataset (i.e. 11 of the 24 species) being bivalves, thus, potentially resulting in an unacceptable bias towards this taxonomic group. Consequently, an assessment of the effect of inclusion of the Markich (2021) bivalve species on the SSD and associated protective concentration (PC) values was undertaken. Without the Markich (2021) bivalve data, the dataset already contained data generated by Saili et al. (2021) for 2 bivalve species: *Crassostrea gigas* (now referred to as *Magallana gigas*) and *Mytilus galloprovincialis* (used in Figure B1 A, below). However, the toxicity values for these 2 species were > 1,500 µg/L, at least 20 times higher than those for the 10 species assessed by Markich (2021). Thus, *C. gigas* and *M. galloprovincialis* could not be considered representative of the other bivalve species. The SSDs based on datasets with and without the Markich (2021) bivalve data are shown in Figure B1 , with the derived PC values in Table B1.

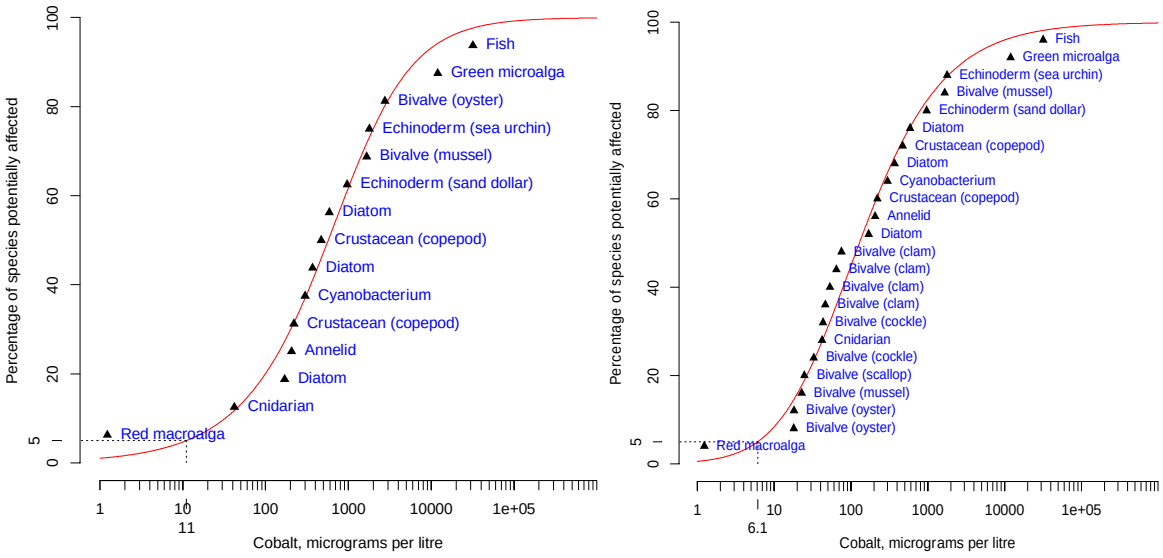


Figure B1 Species sensitivity distributions for datasets (A) without (n = 15) and (B) with (n = 24) the Markich (2021) bivalve species

Table B1 Toxicant protective concentration (PC) values for cobalt in marine water [a]

Percent species protection (%)	Dataset excluding Markich (2021) bivalve data (n = 15) (μg/L)	Dataset including Markich (2021) bivalve data (n = 24) (μg/L)
99	0.95	1.6
95	11	6.1
90	32	12
80	99	27

[a] The PC values were derived using the Burrlioz 2.0 software, They have been rounded to 2 significant figures.

Both SSDs were determined to result in good fits to the datasets, meaning that both sets of PC values would be assigned a very high reliability classification. There were considerable differences between the PC values for the 2 datasets, with the values for the dataset including the Markich (2021) data being approximately 2 to 4 times more conservative Table B1. This appears to be reflective of the relatively high sensitivity of the 10 bivalve species assessed by Markich (2021), all of which lie in the lower 50th percentile of the dataset. To assess whether the predominance of bivalve data was overly influencing the PC values, an alternative dataset was assessed that involved consolidating the bivalve toxicity values from 11 separate species into values for 5 taxonomic sub-groupings (i.e. oysters, mussels, cockles, clams, scallop), using geometric means of the single or lowest value for each species. This dataset comprised 18 toxicity values of which only 28% (5) were bivalves and, thus, was less biased towards this taxonomic group. The SSD for this dataset yielded PC99, PC95, PC90 and PC80 values of 0.9 μg/L, 5.8 μg/L, 14 μg/L and 38 μg/L, respectively (SSD not shown here). All of the PC values were similar to those based on the dataset that included all the individual bivalve species toxicity values. Consequently, it was concluded that the high proportion of bivalve species represented in the dataset was not having a disproportionately large influence on the SSD and associated PC values.

Overall, the results did not provide any compelling argument to exclude the Markich (2021) bivalve data on the basis that they resulted in an over-representation of this taxonomic group. Consequently, the Markich (2021) data were included as individual species values in the final dataset used to derive the DGVs.

Appendix C: modality assessment for cobalt toxicity to marine species

A modality assessment was undertaken for cobalt according to the 4 questions stipulated in Warne et al. (2018). These questions and their answers are listed as follows.

1. Is there a specific mode of action that could result in taxa-specific sensitivity?

As discussed in Section 2.1, cobalt appears to have numerous potential mechanisms of toxicity that are applicable to a range of taxonomic groups, particularly by stimulating the generation of reactive oxygen species. Although there may be a plant/algal-specific mechanism of toxicity, based on the greater sensitivity to cobalt exposure reported for the red macroalga, there is otherwise no evidence to suggest that the toxicity of cobalt would target one taxonomic group more than another.

2. Does the dataset suggest bimodality?

Visual representation of the data, calculation of the bimodality coefficient (BC), and consideration of the range in the effect concentrations are recommended lines of evidence for evaluating whether bimodality or multimodality of the dataset is apparent. For the cobalt dataset:

- The histogram of the natural log transformed toxicity data (Figure C1) is partly right skewed but does not indicate bimodality.
- Datasets that span large ranges (i.e. > 4 orders of magnitude) indicate potential for underlying bimodality or multimodality (Warne et al. 2018). The cobalt marine toxicity dataset spans ~5 orders of magnitude.
- When the BC is < 0.555, it indicates that the data do not follow a typical normal distribution and may be bimodal. The BC for the log transformed data is 0.293, which does not support a conclusion of bimodality.

Based on the lines of evidence described above, the distribution of the log transformed dataset is likely to be unimodal.

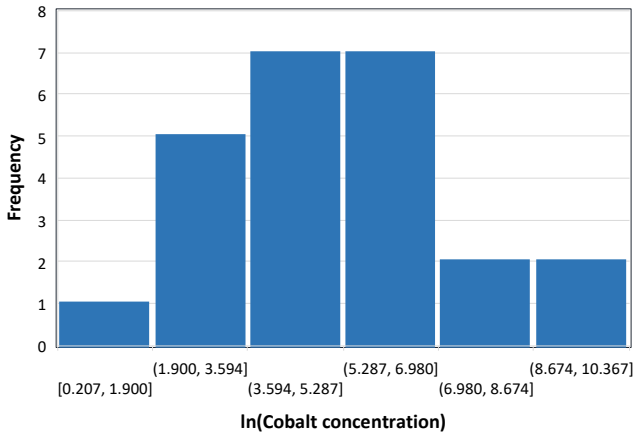


Figure C1 Histogram of natural-log-transformed data for cobalt marine toxicity

3. Do data show taxa-specific sensitivity (i.e. through distinct groupings of different taxa types)?

Figure C2 suggests that the macroalga may be more sensitive to cobalt and fish less sensitive to cobalt, than other taxonomic groups. However, the sample sizes for these and most other taxonomic groups are very small (n = 1–2) and, thus, it is not possible to draw any conclusions about taxa-specific sensitivity.

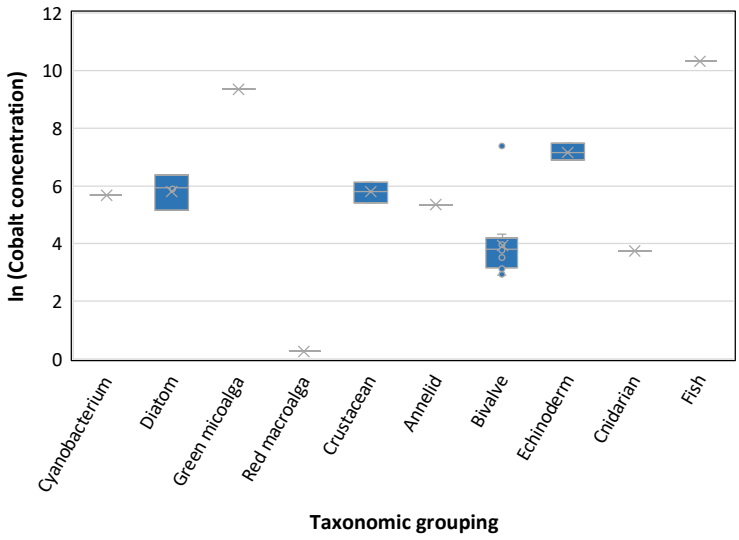


Figure C2 Box plot of natural-log-transformed data for cobalt marine toxicity by taxonomic group

4. Is it likely that indications of bimodality or multimodality or distinct clustering of taxa groups are not due to artefacts of data selection, small sample size, test procedures, or other reasons unrelated to a specific mode of action?

None of the factors assessed (mode of action, indications of bimodality of dataset, and taxa-specific sensitivity) provided strong evidence of bimodality. Generation of additional toxicity data for macrophytes would be useful for incorporating further evidence towards the modality of the dataset. However, on the basis of the available evidence, the dataset appears to be unimodal, which supports the use of the data for the 24 species identified in the preparation of the DGVs derivation.

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