Australian Government
Department of Climate Change, Energy, the Environment and Water

Australian & New Zealand
GUIDELINES FOR FRESH & MARINE WATER QUALITY

Response to public submissions on draft default guideline values for nickel in freshwater

September 2026

Draft default guideline values (DGVs) for nickel in freshwater were published on the Water Quality Guidelines website for a 3-month public consultation period. During this period, comments for the draft DGVs for nickel in freshwater were received via public submission.

Responses to comments and any associated edits to the draft DGV technical brief are outlined in this report, de-identified for public record. In accordance with the formal post public comment DGVs approval process, the responses and revisions have been approved by the jurisdictional technical and policy oversight groups, and noted by the National Water Reform Committee.

The default guideline values for nickel in freshwater are now published as final. For additional information on the publication process, please refer to the pathway for toxicant default guideline value publication.

The Water Quality Guidelines Improvement Program thanks all submissions for their valuable contribution to the development of default guideline values for the protection of aquatic ecosystems.

Attachment A

Response to public submissions on draft default guideline values

Toxicant: Nickel in freshwater

Public comment	Response	Action taken
Submission 1		
1. Errors/inconsistencies with species sensitivity coefficients in Table 2 of the technical brief compared to those reported in Peters et al. (2021)*. * Peters A, Merrington G, Stauber J, Golding L, Batley G, Gissi F, Adams M, Binet M, McKnight K, Schlekat CE & Garman E 2021. Empirical bioavailability corrections for nickel in freshwaters for Australia and New Zealand water quality guideline development. *Environmental Toxicology and Chemistry*. 40(1):113-26.	There were several errors in the Peters et al. 2021 paper which were corrected during the tech brief preparation. These relate to applying assessment factors to LOECs and LC50 values to have EC10 equivalents. The species specific coefficients were recalculated and the ones in the technical brief submitted for public comment are correct. Species for which we used different sensitivity coefficients (SCs) from the Peters et al. (2021) paper and why: • algae *Monoraphidium arcuatum* – there were two SC values. One for a single species assay and one for a multi-species assay. After consulting with Adam Peters, we combined the two tests and calculated an SC for the pooled data instead of using just one SC. • invertebrate *Alona affinis* – The toxicity data were LC50 values that were converted to equivalent EC10 values by dividing by 5 as per Warne et al. (2018) then the SC was rederived. • invertebrate *Ceriodaphnia dubia and Daphnia magna* – Adam Peters has since corrected the LOEC and EC50 values that were wrongly used, and rederived new sensitivity values for *C. dubia* and *D. magna*. • snail *Lymnaea stagnalis* – We changed the SC for this because the water quality parameters for Ca, Mg and DOC had been reported in the database using the wrong units (reported as mM instead of mg/L). Lisa Golding identified the error in May 2019 and corrected the SC value for the correct Ca, Mg and DOC.	No change to SCs used in the final DGV derivation and therefore no change to the final DGV at the index condition. Note that with the new tiered framework, DGVs at the index condition are no longer relevant. A table note was added to Table 2 identifying that corrections were made to the SCs for the 5 species mentioned in the response to the left, and referring to the response to public comments document for more details.
2. Given the natural geology which includes laterites in some tropical regions, it is important to account for ambient or natural Ni background concentrations to	The technical brief acknowledges this issue but it is beyond the scope of the Ni technical brief to address. Guidance on dealing with background concentrations is already provided on the ANZG website, and the DGV technical briefs clearly state that the DGVs need to be used in accordance with the guidance provided on the ANZG website.	No action taken

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better interpret exceedances in GVs related to anthropogenic sources (Wilson et al., 2024). Similar approaches, methodology and guidelines for considering background concentrations should be documented and made available to stakeholders in accordance with the practices of the Commonwealth of Australia."		
Submission 2		
1. Availability of key supporting publications The DGVs are based on a publication by Stauber et al. (2021), who in turn relied on multiple linear regressions (MLRs) developed by Peters et al (2021). Neither of these two key publications was made available via the ANZG website; notably, the latter is behind a paywall.	The Stauber et al. (2021) paper is already open access and freely available. The Peters et al. (2021) was originally not published as Open Access to the general public, but reports important supporting information for the DGVs. Therefore, NiPERA have since paid the required amount to make Peters et al. (2021) Open Access.	Peters et al. (2021) is now available as Open Access.
2. Factors affecting nickel toxicity • Lack of detail on speciation of nickel.	The technical brief is supposed to be that – a brief justifying how the DGVs were determined. It is not supposed to be a complete dossier on nickel speciation.	No change
• Too much information on effect of salinity on nickel toxicity for a freshwater tech brief.	We have deleted some of the text on salinity effects on nickel. However, this text is about Ni and salinity on euryhaline species so it is relevant to Ni in freshwaters too.	Some text on salinity deleted.

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• Lack of information on importance of nickel carbonate in aqueous systems.	Some text has been added on the effects of Ni carbonate in aquatic systems:, as follows: "Nickel carbonate can exist as a complex in water, with $NiCO_3(aq)$ 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 reproduction in *C. dubia* in chronic tests at high pH. Similar effects were not reported for other invertebrates."	Text on nickel carbonate added.
• insufficient detail on ligand binding properties of nickel.	We acknowledge that Ni speciation is complex and refer to Peters et al (2021) for more information. We have had to use existing MLRs which only take limited TMFs into account. Further text on ligand binding properties is too detailed for this tech brief.	No change
• Over-generalisation of effect of 'hardness' on nickel toxicity, when in fact "…even a cursory review of the MLRs finds hardness, *per se*, is not correlated to nickel toxicity."	The emphasis on hardness is because previous Ni DGVs only considered hardness (see algorithm in ANZG, 2018). The Meyer paper on nickel toxicity and hardness referred to by the reviewer only discusses acute toxicity which is not relevant to our DGVs based on TMFs on chronic toxicity -both for species endpoints and response thresholds. However, we have added some additional text: "In the algal MLR development, calcium was not a significant parameter. Magnesium was included, as nickel can compete with magnesium in chlorophyll, making it more important than calcium for plants rather than animals (Peters et al. 2021). For invertebrates and fish, calcium and magnesium act independently, and while both compete with nickel for uptake, they have different binding constants in recent biotic ligand models (Santore et al. 2021). They 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."	Additional text added
• Lack of quantified relationship between DOC and pH for some taxonomic groups; as a result, the LRs need further refinement.	We have developed and used the MLRs currently available and published for Ni, based on quality available literature data on TMFs. Peters et al (2021) acknowledge that some studies did not have sufficiently detailed water chemistry information so these studies could not be used in MLR development. We have added the following text to make this clear: "Peters et al. (2021) acknowledge that some studies did not have sufficiently detailed water chemistry information so these studies could not be used in MLR development."	Additional text added
• Lack of data transparency with regards to DOC concentrations – "…the supplementary data files to the '2024 Draft	The supplementary data files contain all the DOC data for each test/species. The reviewer is questioning the use of 0.5 mg/L and 0.25 mg/L DOC in synthetic waters when DOC is estimated. There is always some DOC in synthetic test water due to the addition of organisms and food, including faecal matter and algal exudates. For all algal and invertebrate MLRs, where measured DOC data were not available, Peters et al. used a default DOC for synthetic waters of 0.25 mg/L (considered to be half the DL of 0.5 mg/L). For fish data, DOC was reported as <1 mg/L so half the DL (0.5 mg/L) was used in the fish MLR. Such estimates of DOC concentration in the water are used in all guideline value derivations	No change

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Nickel Guidelines' do not present all raw DOC data, which must be sought out directly from primary sources. And a review of primary sources finds Peters et al. (2021) incorrectly assign some DOC concentration data."	internationally. Using only measured DOC concentrations and excluding estimated DOC particularly at low DOC concentrations <LOR, would halve the available data and reduce the number of taxa and taxa groups available for inclusion in the SSD. A sensitivity analysis was previously carried out in response to public comments on copper where estimated DOC was not included. There was little difference in resulting DGVs derived with/without estimated DOC, and there would be even lesser effect for Ni as DOC is not as important as a TMF as it is for other metals such as copper for which DOC has strong binding affinity and is particularly ameliorative.	
• Lack of accounting for alkalinity and need for MLRS to be updated to include alkalinity.	No MLRs including alkalinity are currently available for use for nickel. Alkalinity is included as a parameter in the full Ni BLMs, but as shown in Stauber et al (2021), the trophic level specific MLRs (alkalinity not included) were better predictors of Ni toxicity in natural waters than the BLMs. All the papers the reviewer quotes on the effects of alkalinity, pH etc on nickel toxicity are acute studies which are not relevant to chronic nickel toxicity tests that underpin our DGVs. Santore et al. (2021) showed that, for reproduction in *Ceriodaphnia dubia* only, bicarbonate toxicity was one possible explanation for increased sensitivity of this chronic endpoint to nickel at high pH i.e. there is combined toxicity of nickel and bicarbonate. They revised the Ni-HCO_3^- complex parameter in the nickel BLM however this revised BLM did not improve model predictions for other biota. Given that this response is specific to *C. dubia* only, we have not included this in the technical brief. It is likely that alkalinity only becomes an important TMF in waters at very high concentrations, and given that pH and alkalinity co-vary, pH as a TMF is a suitable surrogate to account for alkalinity effects.	No change
3. Multiple linear regressions		
• The draft DGVs "…fail to make clear that only quite limited data was used in the development of MLRs across four trophic levels.".	The technical brief clearly provides the references where more details on the MLR development can be obtained. Nevertheless, we have added some text to the tech brief to make it clear how many species were used in the development and validation of the MLRs: ''The algal MLR was developed from data from *Pseudokirchneriella subcapitata*, and validated with data from 7 different algal species. The aquatic plant MLR was developed using data from *Lemna 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 a local *Melanotaenia splendida splendid.* (See Table 1 in Peters et al., 2021)."	New text added
• Questionable relevance of several species used in MLR development from a New Zealand perspective.	As already described in the tech brief text, 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). No data with local NZ species were available for MLR development. Following Warne et al. (2018), there is no requirement to only use local Aust/NZ species. Rather, use of global data is encouraged to increase the quantity of data available.	No change

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• Incorrect attribution of the fish species used to develop the vertebrate model (cited by Peters et al 2021 as *Pimephales promelas*, but in fact is *Oncorhynchus mykiss*).	The reviewer is confusing the fish BLM of Deleebeeck (based on rainbow trout) with the fish MLR Adam Peters developed (based on *Pimephales promelas).* See record 168 (Deleebeek et al., 2007).	No correction needed
• No MLR for molluscs; application of invertebrate MLR based on crustaceans and a cnidarian "…brings with it uncertainty and an undermining of confidence."; need MLR for molluscs.	No MLRs for molluscs are available for use and there is insufficient data and resources to develop one.	No change
4. The toxicity dataset • Lack of 'reported' (i.e. un-normalised) toxicity values and key WQ parameters in Table 3. Issues with DOC, as reported above, including difficulty in verifying data.	Non-normalised values (190 data) are already reported in the supporting data spreadsheets, so we do not believe that they also need to be reproduced in the tech brief. Several criteria were used to determine which data were used in the final SSD including, reporting of sufficient water chemistry parameters, whether Ni concentrations were measured, whether endpoints were expressed as less than values and using the hierarchy of endpoints as in Warne et al. (2018). Of the 44 species, only 26 species had sufficient water chemistry data to apply the MLRs. Excluded data have now been highlighted in the data spreadsheet. The two species specifically mentioned by the reviewer that were excluded were: *Spermatozopsis exultans* and *Chlorella* (Swedish isolate). These were excluded because neither Ca nor Mg concentrations were reported. We have also further clarified the use of non-preferred data: "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 inclusion of 21 of these NOECs was because they were the only values for those species, including fathead minnow, *Hyallela azteca* and *Clistoronia magnifica*. Two of the three LC/EC50s were also the only values for *Alonas affinis*. Inclusion of these data enabled increased taxa coverage and many different water chemistries for all species for the MLR normalisations."	To make it clearer which data were excluded from normalisation due to a lack of water chemistry parameters, we have highlighted the excluded data in the spreadsheet and added additional text to the tech brief.

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• Counter-intuitive normalised toxicity value for *Lemna aequinoctialis*.	Normalised EC10 values with the MLRs (particularly the aquatic plant MLR) do not always equate to the measured values, and at the EC10 level may vary by up to a factor of 3, which is still considered acceptable. It is not surprising then that the normalised EC10 is 0.0028 mg/L versus a measured EC10 of 0.0035 mg/L. This is a very reasonable prediction at low DOC and varies by only a factor of 1.25, well within MLR prediction acceptability.	
• Unclear why some data in the spreadsheet were not used.	See above comment and criteria. We have also added text identifying which species were excluded due to failing the 50% QA criterion. "Several test data were not used as they failed the QA criteria i.e. scored <50%. These were: *Salmo gairdneri, Micropterus salmoides* and *Ambystoma opacum* (Birge et al, 1978); *Pimephales promelus* (Kszos et al. (1992); *Aeromonas hydrophila (Babich et al., 1986) and a mixed phytoplankton community (*Chakraborty, P., Babu, P. V. R., Acharyya, T. and Bandyopadhyay, D. (2010)."	Excluded data now highlighted in the spreadsheet. Test data that failed QA are now stated in the text.
• Issues/errors associated with the reporting of literature data for the snail, *Lymnaea stagnalis*; concerns over the use of data for this species given conclusions of Cremazy et al. (2020).	The draft Ni DGVs were developed before the Cremazy review was published in 2020. We acknowledged in the tech brief that there was some controversy around the snail EC10 value used, especially given that it is the most sensitive species in the SSD and its sensitivity varies widely. However, the tests with this species passed the quality assurance criteria (score of 94%) and this species has been used in developing other guidelines e.g. for Zn in Australia and New Zealand, and for Ni in the EU. Nevertheless, we have rerun the SSD excluding the snail value as a comparison. See below Burrlioz plot. Excluding the snail increases the PC95 from 2.0 at the index condition to 3.7 µg/L. However, if we plot SSDs for both with and without the snail using the now recommended ssdtools, the PC95 only changes from 2.1 to 2.9 µg/L at the index condition used in the draft technical brief. See below. Note also that these DGVs at the index condition will no longer be used in the new tiered framework. We conclude therefore that there is no justification to exclude the snail data from the ecotoxicity dataset.	New SSDs with and without the snail were run but no change recommended. Updated DGVs using new ssdtools.

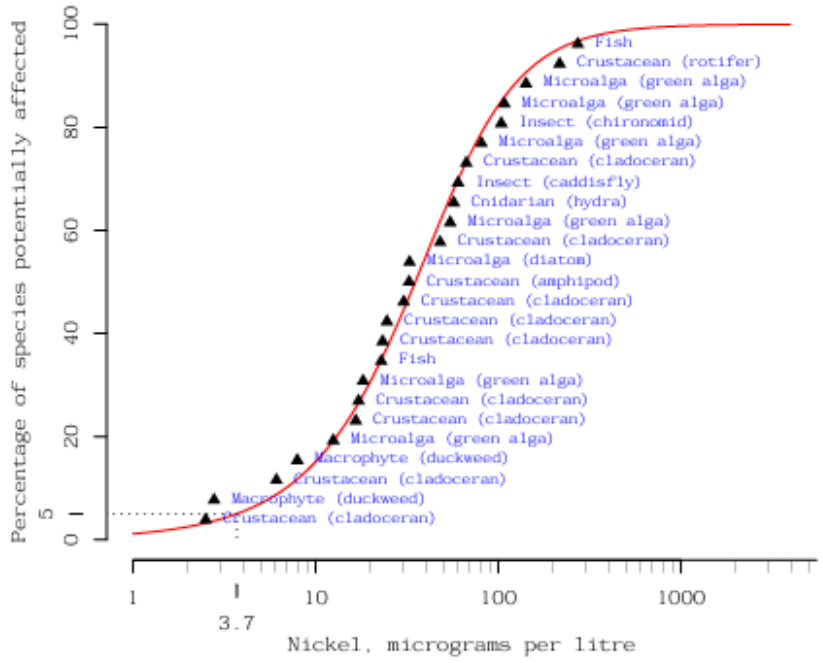
Attachment A

Toxicant: Nickel
Input file: C:\Users\goll2c\Documents\Projects\2018\NiPERA NiWQ_2018\Freshwater Ni\SSD I
Time read: Wed Jun 11 10:00:24 2025
Units: micrograms per litre
Model: Burr type III

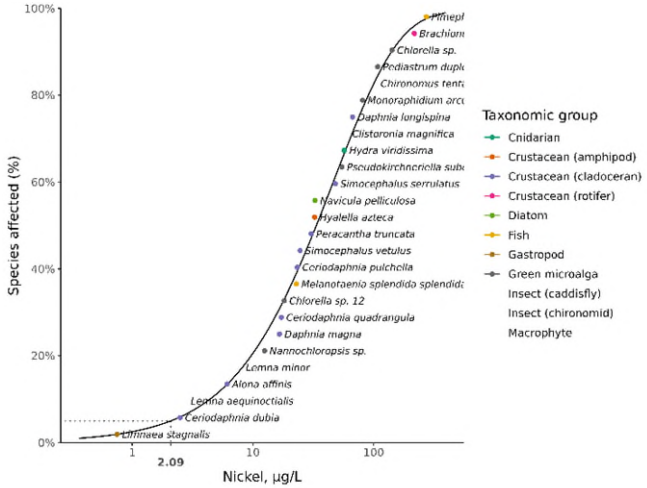
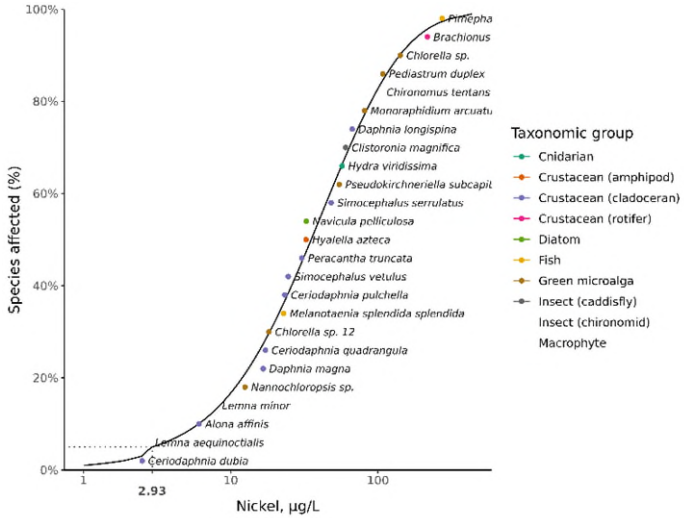
Protection level information

Protect. level	Guideline Value	lower 95% CI	upper 95% CI
99%	0.9	0.095	6.5
95%	3.7	1.2	11
90%	6.9	3.2	15
80%	13	7.5	23

notes:



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	 Above – with snail. Below without snail. 	

Attachment A

Public comment	Response	Action taken
5. The Final DGVs and Practical Considerations Various comments including: • The DGVs "…rely on much data from species that have little relevance to New Zealand."	If no data are available for NZ species, then the onus is on NZ to undertake tests with nickel on local species so that more NZ species can be included in future. The acceptable approach outlined in Warne et al (2018) is to include global data so that sufficient taxa are included in the SSD. This includes *Chlorella* sp. from PNG which was the 6th most sensitive species to Ni.	No change
• Concerns over pooling of data from tropical and temperate systems.	We refer the reviewer to the paper by Peters et al (2019) in ET&C 38, 1211-1220 https://doi.org/10.1002/etc.4384 which showed that there was no difference in the sensitivity of tropical and temperate freshwater species to nickel and recommended combining datasets for GV derivation. In marine waters, this is also the case (see Gissi et al. (2020) in ET&C 39, 2540-2551 (https://doi.org/10.1002/etc.4880).	No change
• Difficulties in accurately measuring trace concentrations of nickel in freshwaters – will increase the risk of exceedances due to sample contamination.	We agree with the reviewer that it is difficult to measure dissolved Ni at ultratrace concentrations and that avoiding contamination is paramount. However, at the µg/L level, modern day labs should be able to achieve appropriate detection limits for Ni using ICP-MS etc. We have added the below text: "During the public consultation process, concerns were raised regarding the difficulty in the analytical detection of nickel at trace concentrations e.g. below 1 µg/L in freshwater samples. More importantly, erroneous results can be obtained if specialised trace metal sampling techniques are not followed. Sampling details are provided in the Australian guidelines for water quality monitoring and reporting (ANZECC/ARMCANZ 2000b). Analytical methods for trace nickel should be well within the scope of qualified analytical laboratories, and certified reference water samples are available for use in quality assurance."	Additional text added
6. The Look-up table approach Various comments including, but not limited to: • Look-up tables represent a 'black box' approach and do not specify DGVs at intermediate TMF values. • Obvious failing is with the inclusion of	These look up tables have been superseded and will be removed. In the proposed new tiered framework, an on-line tool will enable calculation of Ni bioavailability-adjusted GVs at the matching water chemistry of the sample. This tool includes the need for Ca and Mg concentration data (not hardness) as these parameters affect Ni bioavailability. Guidance and training materials have been developed to assist users with the new tool and framework.	Look up tables deleted.

Attachment A

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hardness, which is not needed at all. • Too limited a set of Ca and Mg concentrations. Suggest an alternative to look-up tables.		