Peer Review Memo of Buller Plateaux Continuation Project-Water Quality Protection Guidelines

Date: 17 August 2026

Job No.: NZL.06355.0000

Executive Summary

1. I have undertaken an independent review of the water quality guideline framework developed for the Buller Plateaux Continuation Project (BPCP), focusing on the scientific defensibility of the proposed water quality guidelines, stream classifications, and Levels of Protection (LoP). Overall, I consider the site-specific turbidity guidelines, stream classification framework, and the use of toxicity-modified site-specific guideline values derived using the Canadian Federal Environmental Quality Guidelines to be scientifically robust and generally consistent with the intent of the ANZG framework. These approaches appropriately recognise the unique water chemistry of the Buller Plateaux streams, including their naturally low pH and elevated dissolved organic carbon concentrations.

2. Given the unique geochemical characteristics of Buller Plateaux streams, I consider the use of site-specific guidelines that account for toxicity-modifying factors, including dissolved organic carbon and pH, to be both appropriate and scientifically justified.

3. However, I have identified two areas that, in my opinion, warrant further consideration. These are:

    (i) the proposal to assess compliance with chronic site-specific guideline values using median concentrations rather than a higher level of compliance,

    (ii) the reliance on the US EPA iron criterion or the newly proposed iron criteria of 0.5 mg/L, which I consider of uncertain relevance to the naturally acidic, fishless plateau stream environments.

4. Subject to these matters being satisfactorily addressed, I consider the assessment provides a sound basis for managing water quality effects associated with the proposed mining activities.

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1 Introduction

5. My full name is Andrew John Rumsby. I am a Principal Environmental Chemist at EHS Support New Zealand.

6. This technical review addresses the water quality guidelines proposed for the Buller Plateaux Continuation Project (BPCP) and provides an independent review of the environmental chemistry and aquatic environmental risk assessment aspects of the Project.

1.1 Qualifications and experience

7. I have the following qualifications and experience relevant to this assessment:

   (a) Master of Science (Hons) Chemistry/Earth Sciences from the University of Waikato

   (b) Bachelor of Science (Chemistry/Geology) from the University of Auckland

   (c) I am a Certified Environmental Practitioner – Contaminated Sites (CEnvP-CS)

   (d) Since February 2020, I have been employed at EHS Support New Zealand. Prior to that:

      (i) Pattle Delamore Partners for 14 years as an Environmental Scientist (2007-2020)

      (ii) URS New Zealand Limited for 7 years as an Environmental Scientist (2000-2007)

(iii) Environmental Science Research (ESR) as an Environmental Data Analyst (1999-2000)

(iv) Taranaki Regional Council as an Environmental Monitoring Officer (1997-1999)

(e) I specialise in environmental risk assessment and have provided environmental and human health risk assessment for the last 25 years in NZ, including assessing the impacts of discharges of mine and quarry waters on the receiving environment at various locations throughout New Zealand. This work has included:

(i) Assessing the environmental and human health risks associated with the historical mercury mine in Puhipuhi for Evolution Mining and the Department of Conservation.

(ii) Assessing environmental and human health risks associated with historical mining in the Coromandel Peninsula (including Maratoto mine and tailings impoundment, Broken Hills, Victoria Battery, Zeehan Battery, Monowai mine, Waikino Concentrate Impoundment) for the Department of Conservation. This work included a high-level data review of the environmental effects of historical mining in the Coromandel and risk ranking of priority sites.

(iii) Undertaken peer review of water quality impacts of discharge consents for various coal mines and quarries in the Waikato Region for the Waikato Regional Council (including Maramarua and Rotowaro Coal Mines and AB Lime limestone quarry, Te Kuiti).

(iv) Undertaking site investigation, remediation support, water quality monitoring, ecological assessment, and long-term post-remediation monitoring of the Tui Mine, Mt Te Ahora, for the Waikato Regional Council and the Matamata Piako District Council.

(v) Assessing water quality impacts for Quarry and Managed Fill in Huntly. This project involved assessing the water-quality impacts of historical coal mining operations at O'Reilly's Mine and the Weaver Pit Overburden disposal sites. This work involved undertaking Whole Effluent Toxicity testing and developing site-specific water quality guidelines.

(vi) Long-term continuous monitoring and assessment of suspended sediment flux for the re-consenting of the Lower Waikato Flood Control Scheme. The work involved the design, installation, and management of a continuous water-quality monitoring network that incorporated flow, turbidity, and other key parameters. This included the development of site-specific flow–sediment rating curves, analysis of continuous monitoring datasets, estimation of suspended sediment loads and fluxes, and the interpretation and reporting of long-term water quality trends to support the re-consenting process and assessment of environmental effects.

(vii) My MSc at the University of Waikato was on the Environmental Geochemistry of Sulphide Mine Wastes.

1.2 Code of conduct

8. I confirm that I have read the Code of Conduct for expert witnesses contained in the Environment Court Practice Note 2023. This review has been prepared in compliance with that Code, as if it were evidence being given in Environment Court proceedings. In particular, unless I state otherwise, this assessment is within my area of expertise, and I have not omitted to consider material facts known to me that might alter or detract from the opinions I express.

1.3 Purpose and scope of review

9. The purpose of this review is to provide independent review of and comment on the following:

(a) Water Quality Guidelines and Level of Protection Document – Final received 14 July 2026

10. My review has focused on the scientific defensibility of the proposed water quality guidelines, including the classification of streams and the appropriateness of the proposed levels of protection. I have not undertaken an independent review of the proposed water quality monitoring programme, hydrology or ecology of the streams, as I understand these matters are being addressed by other experts. Accordingly, my opinions are limited to the derivation of the water quality guidelines and the scientific basis underpinning the proposed stream classifications and protection levels.

11. As part of this review, I have also read the following documents:

(a) NIWA (2020) Water Quality and Aquatic Ecology for the Buller Plateau Project – May 2020

(b) NIWA (2024a) Mt Fred South Aquatic Ecology Field Survey – November 2024

(c) NIWA (2024b) Escarpment Mine Reconsenting Field Survey – November 2024

(d) NIWA (2025) Cascade Creek aquatic ecology baseline survey – Memo – June 2025

(e) RMA Science (2025) Response to Technical Peer Review – Site-Specific Water Quality Protection Levels for Turbidity – 9 December 2025

(f) Mitchell Daysh (undated) Buller Coal Plateaux Continuation Project – Briefing for Technical Peer Reviewers - 21 November 2025

(g) Bathurst Resources (2025) Buller Plateaux Continuation Project – Project Description (version 3) – November 2025

(h) ESNZ (2025) Cobalt and Nickel Toxicity to Algae and Mayfly Species in Mine Receiving Waters.

(i) Unpublished Excel Spreadsheet – Water Quality data for Denniston-Stockton Region-Updated February 2020 - Received December 2025

(j) BPCP: Classification of Streams for Water Quality Protection Levels – memo dated 28 November 2025

(k) Site-specific water quality protection levels for turbidity – memo dated 11 November 2025.

1.4 Assumptions and exclusions in this assessment

12. For my review, I have assumed that the water quality data supplied is accurate and representative of site conditions. I have not independently audited, validated, or otherwise assessed the quality assurance and quality control procedures associated with collecting, analysing, or reporting the water quality data.

13. Unless specifically identified in Paragraph 11, I have not reviewed the supporting ecological, hydrological, Mine Waste Management (MWM) water quality modelling reports and water quality reports referred to in the document listed in Paragraph 9

(Purpose and Scope of Review). My opinions are therefore based solely on the information sources identified in Paragraph 9 and the materials reviewed as part of this assessment in Paragraph 11 and identified (referenced) within this review.

2 The Buller Plateaux Continuation Project

14. The full description for the Project is provided in the AEE. As such, I have not repeated this information in my review.

3 Methodology

3.1 Introduction

15. My approach has been to review all relevant Project reports I have been provided with to develop an understanding of the environmental effects of the project and undertake an independent review of the proposed level of protection at the various monitoring sites and the appropriateness of the proposed water quality guideline values.

16. I undertook a site visit of the Denniston Plateau and Stockton Coal Mine on 29 and 30 June 2026.

4 Independent Review of Documents

4.1 Review of the Site-Specific Water Quality Protection Levels for Turbidity

17. An initial review of the *Site-Specific Water Quality Protection Levels for Turbidity* memorandum was undertaken in November 2025. Following this review, high-level comments and feedback were provided to the author, and these were subsequently discussed to clarify my comments, questions and recommendations.

18. A revised version of the memorandum, together with responses to my initial review comments and additional updates, was provided to me in December 2025.

19. Having reviewed the revised memorandum and the responses to my comments, I am satisfied that the questions and clarifications raised during my initial review have been adequately addressed in the December 2025 version of the memorandum and subsequently carried through to the assessment report 'Buller Plateaux Continuation Project: Water quality protection guidelines'.

4.2 Review of the BPCP: Classification of Streams for Water Quality Protection Levels

20. An initial review of the BPCP-Water Quality (BPCP-WQ) report was undertaken in November 2025. Following this review, high-level comments and feedback were provided to the report author, and these were subsequently discussed to clarify my comments, questions and recommendations.

21. The BPCP-WQ report underwent multiple revisions between December 2025 and 15 June 2026, and my opinions have been informed by consideration of these successive revisions, together with the responses and explanations provided during those discussions. As a result, a number of my initial concerns were either addressed or clarified through the report revision process and subsequently carried through to the assessment report 'Buller Plateaux Continuation Project: Water quality protection guidelines', while others remain the subject of the opinions expressed in this evidence.

5 Comment on the Methodology Used to Develop the Site-Specific Guideline Values For Turbidity

22. Acute turbidity guidelines were developed based on published peer-reviewed laboratory toxicity studies involving New Zealand native fish and macroinvertebrates exposed to very high turbidity for 24 hours. Based on the response data for turbidity, Hickey (2026) identified smelt (*Retropinna retropinna*) as the most sensitive species tested, with a 10% effect threshold of approximately 550 NTU. Accordingly, a maximum turbidity guideline of 550 NTU was adopted to provide protection for downstream fish communities during intermittent short-duration discharge events.

23. Based on my review of the information supplied, the proposed acute guideline value of 550 NTU appears to be appropriately derived and scientifically defensible. The threshold is based on the most sensitive species identified in the available New Zealand toxicity dataset and is therefore conservative in protecting downstream fish communities. I have not identified any alternative New Zealand studies indicating the need for a more stringent acute turbidity threshold for short-term exposure events.

24. Chronic turbidity guidelines were developed primarily from West Coast field studies[2] that linked sustained increases in turbidity from mining discharges to declines in macroinvertebrate abundance, species richness and EPT (Ephemeroptera

(Mayflies), Plecoptera (Stoneflies) and Trichoptera (Caddiflies)) taxa richness. The guidance is an additive one (background plus) and was derived from the effects of abundance (+5 NTU) and effects on taxa (+20 NTU) identified by Reid & Quinn (2011) and applied as the 95th percentile based on annual data. Additional consideration was given to the resilience of aquatic communities to intermittent high-turbidity events, the effects of turbidity on fish feeding and behaviour, and incorporating recommendations developed for West Coast Streams[1].

25. Based on my review of the information provided, I consider the proposed chronic turbidity guidelines to be scientifically robust and appropriate for the receiving environments affected by the proposed mining activities. The longer-term ecological guidelines, based on site-specific continuous monitoring data and expressed as allowable increases above baseline conditions, provide a scientifically defensible framework for protecting macroinvertebrate communities while recognising the naturally variable turbidity regimes that characterise West Coast catchments. In my opinion, Dr Hickey has applied a rigorous, evidence-based methodology that appropriately integrates laboratory studies, field ecological data, site-specific baseline conditions, and established West Coast sediment guidance to derive the proposed chronic turbidity guidelines.

6 Comment on the Classification of Streams for Water Quality Protection Levels

26. The BPCP Stream Classification memo develops a stream classification for the Escarpment Extension (ESE), Mount Frederick South (MFS) and Upper Waimangaroa Haul Road (UWHR) sub-areas of the Buller Plateaux Continuation Project. This assessment uses a weight-of-evidence approach, consistent with ANZG guidance, to relate water quality conditions and biological community health to appropriate ecosystem protection levels. The framework integrates chemical indicators (including pH, electrical conductivity, sulphate, aluminium, zinc, and nickel) with biological measures, particularly EPT taxa richness and kōura abundance, to classify streams into four condition categories: Excellent, Good, Fair and Poor.

[1] Reid, D and Quinn, J. (2011) Preliminary Information for developing sediment guidelines for stream of the West Coast, New Zealand. NIWA Client Report

27. While I consider a multi-criteria assessment using a weight-of-evidence approach to be an appropriate methodology for identifying an acid mine drainage (AMD) impact gradient, I also agree with Dr Hickey that no single indicator is sufficient to determine stream condition or establish whether a stream has been adversely affected by AMD, particularly in naturally acidic waterways with elevated dissolved organic matter concentrations (operationally measured as dissolved non-purgeable organic carbon[2] (DOC) that also have naturally elevated background concentrations of metals.

28. AMD impacts are generally characterised by changes in water chemistry that include reductions in pH, increases in electrical conductivity and sulphate concentrations, and elevated dissolved metal concentrations, such as aluminium, iron, nickel, and zinc. These changes can adversely affect biological communities, and therefore a combination of chemical, biological, and toxicological assessments are required to evaluate a stream's health when it has been impacted by AMD.

29. In my review dated 9 February 2026 of BPCP: Classification of Streams for Water Quality Protection Levels, I recommended that greater transparency and reproducibility could be achieved by using a structured scoring framework, rather than relying primarily on expert judgement when assigning levels of protection. This would potentially improve the robustness, transparency, and ease of interpretation of the assessment. However, it is unlikely that implementing these recommendations would have materially altered the overall classification outcomes or the broad allocation of protection levels. Rather, they would have provided a clearer and more transparent basis for commissioners and submitters to understand how the selected levels of protection were derived.

30. In my opinion, the available water chemistry and ecological information provided a sufficient basis for identifying a broad acid mine drainage impact gradient and categorising streams according to their relative condition.

7 Comment on the Buller Plateaux Continuation Project: Water Quality Protection Guidelines

[2] Dissolved non-purgeable organic matter (NPDOM) is the fraction of dissolved organic material, primarily comprising humic substances, fulvic acids, tannins, dissolved plant decomposition products, microbial metabolites and other non-volatile organic compounds, that remains in a water sample following the removal of inorganic carbon (carbon dioxide, carbonate and bicarbonate) and volatile organic constituents (i.e. VOCs). The sample is filtered through a 0.45µm filter and acidified to convert all the carbonate and bicarbonate into carbon dioxide before being purged with air or an inert gas to remove the carbon dioxide and volatile organic compounds.

31. The aquatic ecosystems of the Buller Plateaux occur under water chemistry conditions that are atypical of most New Zealand lowland streams. Elevated DOC and organic acids naturally depress stream pH to values as low as 4.6, while hardness concentrations are often less than 5 mg/L as $CaCO_3$. Because metal toxicity is strongly influenced by pH, hardness, and DOC[3], generic water quality guidelines derived under standard reference conditions (circumneutral pH, 30 mg/L as $CaCO_3$ hardness, and 1 mg/L DOC) are not representative of Buller Plateaux streams. Site-specific toxicity-modifying factors are therefore required to account for these natural water chemistry conditions and to derive guideline values that appropriately reflect environmental risk.

32. In streams affected by historical AMD discharges, treatment with lime can significantly increase water hardness, often to concentrations exceeding 100 to 200 mg/L as $CaCO_3$. Elevated hardness reduces the bioavailability and toxicity of many metals, including aluminium, cadmium, copper, lead, nickel, and zinc, through competition between calcium and magnesium ions and dissolved metals for biological uptake sites. Consequently, aquatic organisms in high-hardness waters can generally tolerate higher dissolved metal concentrations than those in soft waters. This effect is recognised internationally and underpins hardness-based toxicity corrections incorporated into many water quality guideline frameworks. Iron is an important exception, as available evidence indicates that iron toxicity is influenced primarily by pH and DOC rather than hardness, meaning that increases in hardness associated with lime treatment are unlikely to substantially reduce iron toxicity.

33. In addition to influencing metal toxicity, low pH can exert direct physiological stress on aquatic organisms by disrupting ion regulation, respiration and metabolic processes. Published studies indicate that sensitive macroinvertebrate taxa, particularly Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, commonly begin to decline where pH falls below approximately 5.5, while adverse effects on fish communities generally become evident below pH 5.5-6.0 for the most sensitive New Zealand species, with severe ecological impairment often occurring below pH 4.5. However, many Buller Plateaux streams are naturally acidic because of peat-derived organic acids and support biological communities adapted to these conditions. Naturally occurring pH values frequently fall below the generic ANZG trigger value range of pH 6.5-8.0 in the absence of mining influences[4].

[3] See Appendix A for more information on how dissolved organic carbon influences metal and iron toxicity.

34. In my opinion, and consistent with Dr Hickey's assessment, an annual 95th percentile compliance range of pH 4.5-7.0 for streams assigned 90% and 95% levels of protection is appropriate for Buller Plateaux streams. This range encompasses the natural background conditions commonly observed (although at times natural streams can be below this value; see paragraph 35) within these ecosystems while remaining above the threshold where severe ecological impairment would generally be expected for acid-tolerant biological communities.

35. It is recognised that short-duration reductions in pH below 4.5 in West Coast Streams may occur naturally during high-flow and first-flush events as a result of increased inputs of naturally occurring organic acids (such as humic and fulvic acids) from peatlands, wetlands and forest soils[4,5]. Such episodic events may result in pH values approaching 3.5-4.0 for short periods in both reference and compliance streams. Accordingly, where the relevant reference site records a pH below 4.5 during a concordant monitoring event, the compliance site should not be considered non-compliant solely because pH is below 4.5. In these circumstances, assess compliance relative to the contemporaneous reference condition, and consider whether the discharge has caused a measurable reduction in pH compared with the reference stream as part of any compliance assessment.

36. The BPCP-WQ report develops site-specific water quality guidelines for the Buller Plateaux Continuation Project, focusing primarily on the ESE and MFS mining areas. It applies a weight-of-evidence approach[6] that integrates water chemistry, hydrology, ecotoxicology, and aquatic ecology to derive bioavailability-based guidelines for metals and other contaminants. It then uses the stream classification system outlined in the BPCP Stream Classification memo to assign different levels of ecological protection based on existing AMD impacts. Overall, Dr Hickey's approach is consistent with the intent of the ANZG guidelines.

37. I consider that the use of Toxicity-Modified Site-Specific Guidelines in place of older ANZG guidelines is appropriate because they incorporate toxicity-modifying factors such as pH, hardness, and dissolved organic carbon, and these approaches provide

[4] Greig, H., Niyogi, D., Hogsden, K., Jellyman, P., Harding, J. (2010) Heavy metals; Confounding factors in the response of New Zealand Freshwater fish assemblages to natural and anthropogenic acidity. Science of the Total Environment, pp 3240-3250
[5] Collier, K., Ball, O., Graesser, A., Main, M., Winterbourn, M. (1990) Do organic and anthropogenic acidity have similar effects on aquatic fauna? Oikos, pp 33-38
[6] The ANZG guidelines recommend that water quality guideline values should be used in conjunction with other lines of evidence in a weight-of-evidence process to assess and manage water quality. See Weight of evidence process - https://www.waterquality.gov.au/anz-guidelines/guideline-values.

a more appropriate basis for ecological protection within the naturally acidic Buller Plateau stream environments.

38. My review has identified two principal areas of the BPCP-WQ report where, in my opinion, further consideration or clarification is warranted:

(a) The level of compliance associated with the site-specific guideline values.

This includes an assessment of the proposed compliance framework, the frequency and magnitude of allowable exceedances, and whether the proposed approach is consistent with the intended level of ecological protection and accepted water quality guideline implementation practices.

(b) The Proposed Freshwater Iron Criterion.

In the BPCP-WQ report, Dr Chris Hickey proposed an interim freshwater iron guideline derived from the US EPA freshwater iron criterion by applying a factor-of-two modification. While intended as a pragmatic interim approach, the scientific basis for the magnitude of this adjustment is not clearly documented, and the extent to which it accounts for the unique water chemistry and ecological characteristics of Buller Plateau streams requires further consideration of these criteria.

This section of my review considers the scientific relevance and applicability of both the US EPA criterion, the newly proposed iron criteria and other available freshwater iron criteria to the ESE and MFS plateau streams. In particular, I have evaluated the uncertainties associated with the water chemistry, ecological receptors, iron speciation, and environmental conditions of the water bodies used to derive these criteria.

39. Prior to discussing the two principal areas of the BPCP-WQ report where there is some disagreement between Dr Hickey and my views, I will outline the reasons why I support the replacement of some of the ANZG with newer site-specific guidelines.

7.1 Use of Toxicity-Modified Site-Specific Values

40. As part of this assessment, Dr Hickey replaced the ANZG (2018) default guideline values for metals that have not been updated to consider relevant toxicity-modifying

factors with Canadian FEQG[7] when they have been updated to reflect relevant advances in ecotoxicological science (See Table 7.1). I support this approach and consider it to be consistent with the principles of the ANZG Guidelines[8], which recognise the importance of incorporating site-specific environmental conditions and the best available scientific information when assessing ecological risk.

TABLE 7-1 COMPARISON OF CURRENT ANZG AND CANADIAN FEQG CHRONIC WATER QUALITY CRITERIA FOR METALS[1]

Element	ANZG		Canadian FEQG[2]	
	Year	Toxicity-modifying factors	Year	Toxicity-modifying factors
Aluminium	2000	pH	**June 2021**	**pH, hardness, DOC**
Cadmium	2000	Hardness	**2014**	**Hardness**
Cobalt	2000	Hardness	**May 2017**	**Hardness**
Cobalt[4]	2026? (draft)	Unknown		
Copper	2000	Hardness	May 2021[5]	pH, DOC, Hardness
Copper[3]	**January 2026 (draft)**	pH, DOC		
Iron	2025	None	May 2024	pH, DOC
Manganese	2000	Hardness	**2019**	**Hardness**
Nickel	**2000**	**Hardness**	B.C. Guidelines (pH, Hardness and DOC)[6]	
Nickel[3]	July 2024 (draft)	Hardness, DOC, and pH		
Zinc	2000	Hardness	**2018**	**pH, DOC, Hardness**
Zinc[3]	May 2024 (draft)	DOC, hardness and pH		

Table Notes:

Bold – guidelines recommended in the BPCP-WQ

1. Based on recommendations for bioavailability guidelines in Table 10-1 of the BPCP-WQ report.
2. Does not include older CCME water guidelines, which have not been recommended within the BPCP-WQ report.

[7] With the exception of copper, the Canadian FEQGs are generally derived using the same SSD-based framework now adopted by ANZG, albeit with jurisdiction-specific ecotoxicity datasets. Recent Canadian guideline derivations, including the freshwater iron FEQG, utilise the ShinySSDTools platform that underpins current ANZG guideline development. Accordingly, I consider the selection of Canadian FEQGs to be fully consistent with the hierarchy principles of MfE (2011) Contaminated Land Management Guidelines No. 2. Which recommends that were existing New Zealand guideline values do not exist (or in my opinion do not adequately account for site-specific conditions), it is appropriate to rely on international guideline values derived using comparable contemporary methodologies. In this respect, recent Canadian FEQGs are more closely aligned with current ANZG guideline derivation practices than many older guideline values currently adopted in New Zealand or other overseas jurisdictions such as the US EPA.

[8] See Section of Guidelines Values for Water/Sediment Quality (https://www.waterquality.gov.au/anz-guidelines/guideline-values) which recommends that you should use guideline values that are relevant to your to local conditions or situation. The ANZG default guideline values are recommended to be used if there is an absence of more relevant guidelines values and the ANZG (2018) guidelines do state that the DGVs can be tailored to make them more relevant to local conditions. The ANZG guidelines also state that guideline values should be used in conjunction with other lines of evidence in a weight-of-evidence process to assess and manage water quality.

3. Copper, nickel and zinc freshwater guidelines have been finalised by the National Water Committee (NWC) but have not yet been released. Could potentially be released in July or August 2026. (See Default guidelines values (DGVs) status -https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/draft-dgvs#default-guidelines-values-dgvs-status)
4. Cobalt freshwater guideline has been proposed to be released as a draft in October 2026 (see Default guidelines values (DGVs) status -https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/draft-dgvs#default-guidelines-values-dgvs-status)
5. Canadian FEQG for copper were calculated using the Biotic Ligand Model, which is significantly different to the methodology used by ANZG.
6. British Columbia Guidelines have been used instead of CCME guidelines as they are more up to date.

41. The surface waters of the Buller Plateaux are naturally characterised by low pH and elevated concentrations of dissolved organic carbon (DOC). These water-quality characteristics significantly influence the speciation, bioavailability, and toxicity of many dissolved metals. Contemporary ecotoxicological research has demonstrated that toxicity is not solely a function of the total dissolved metal concentration but is largely governed by the proportion of metal present in biologically available forms and the extent to which environmental factors modify metal uptake by aquatic organisms. Consequently, assessments based solely on total metal concentrations are unlikely to accurately reflect the actual ecological risk posed under site-specific conditions.

42. The influence of DOC is particularly important for metals such as copper (Cu), nickel (Ni), cobalt (Co), zinc (Zn) and aluminium (Al), all of which occur within receiving environments on the Buller Plateaux. Dissolved organic matter forms stable complexes with many metal ions, reducing the concentration of free metal species available for biological uptake. Numerous laboratory and field studies have demonstrated that increasing DOC concentrations can substantially reduce the toxicity of dissolved metals by decreasing their bioavailability. As a result, guideline values derived under relatively low-DOC concentrations in laboratory toxicity experimental conditions may overestimate ecological risk when applied to naturally organic-rich waters such as those present on the Buller Plateaux. The incorporation of DOC as a toxicity-modifying factor in the ANZG guidance and the Canadian FEQG frameworks, therefore, represents a significant scientific improvement over guideline approaches that do not adequately account for these site-specific conditions.

43. A further advantage of the Canadian FEQG approach is the incorporation of pH as a toxicity-modifying factor when deriving site-specific guidelines. Surface waters on the Buller Plateaux exhibit a wide natural pH range, from strongly acidic to near-neutral conditions. These conditions arise from a combination of naturally occurring organic acids associated with wetlands and native forest ecosystems, rainfall, together with

the influence of historical acid mine drainage at some locations. pH is a key control on metal speciation and bioavailability and therefore substantially influences toxicity. However, it should also be recognised that some of the low-pH conditions encountered on the Buller Plateaux fall outside the pH range represented in the toxicity datasets used to develop the underlying toxicity-modifying factor models. Accordingly, caution is warranted when applying these models to waters with pH values below approximately 6.0. Under such conditions, metal speciation may differ significantly from that represented in the model calibration dataset, and the resulting toxicity-modified guideline values may be associated with greater uncertainty. While the Canadian FEQGs provide a more scientifically robust framework for assessing metal toxicity than older guideline approaches, the limitations of extrapolating beyond the models' validated range should be acknowledged when interpreting results from highly acidic waters.

44. Dr Hickey has used the draft ANZG for copper rather than the Canadian FEQG copper guideline because the latter is based on a Biotic Ligand Model (BLM), which requires a relatively large number of water chemistry inputs, including alkalinity, chloride, calcium, magnesium, sulphate, dissolved organic carbon (DOC) and pH. This reflects the recent direction of ANZG guideline development, which has generally preferred multiple linear regression (MLR) models because they are simpler, require fewer monitoring parameters, and are more readily applied within regulatory and consenting frameworks. Given that ANZG has adopted MLR approaches for several metals, I do not consider an MLR-based bioavailability adjustment scientifically inferior to a BLM-derived approach[9]. Rather, it represents a pragmatic and scientifically defensible methodology that balances predictive performance with practical implementation requirements.

45. In the case of nickel and zinc, Dr Hickey has recommended the use of the Canadian FEQGs for zinc on the basis that they have been finalised and the ANZG (2018) guideline value for nickel (which is the ANZG 2000 hardness-dependent GV). I generally agree with this recommendation, as it is appropriate to rely on published and approved guideline values where available. However, the ANZG website indicates that the draft bioavailability-based guideline values for these metals have been approved by the National Water Quality Management Strategy (NWQMS)

[9] This would be consistent with the recommendations in NIWA (2023) Implementing bioavailability-based toxicity guideline values for copper and zinc in Aotearoa New Zealand: Interim technical guidance for scientists and practitioners, focusing on freshwater applications.

Water Quality Committee and are expected to be published in July to November 2026. As Australian and New Zealand-specific guidelines, they are likely to provide the most relevant regulatory framework for assessing and managing water quality effects within New Zealand and will include any relevant ecotoxicity data on New Zealand species.

46. Depending on when the ANZG guidelines are finalised, they can be potentially adopted either when setting consent conditions or at a specified review date. Dr Hickey has recommended that all guidelines are reviewed after 6 years to coincide with other specified investigations after that period.

47. In summary, I consider that the recalculation of toxicity-modified, site-specific guideline values using the Canadian ecotoxicological database represents an appropriate and scientifically defensible approach that is consistent with the best available scientific knowledge and with the framework and recommendations of the ANZG (2018) Guidelines. The approach recognises the importance of toxicity-modifying factors, including pH, hardness, and dissolved organic carbon (DOC), which influence contaminant bioavailability and toxicity in freshwater environments. By explicitly accounting for the unique water chemistry characteristics of the Buller Plateaux streams, the resulting guideline values are likely to provide a more ecologically relevant and scientifically robust basis for environmental assessment and compliance monitoring than directly applying older guideline values that do not incorporate these factors.

7.2 The Level of Compliance Required with the Site-specific Guideline Values.

48. Within the BPCP-WQ Report, Section 10.1.1, Dr Hickey proposes that the median dissolved metal concentration be compared with the bioavailability-adjusted site-specific guideline value applicable to the LoP assigned to each monitoring site. In discussions with Dr Hickey, he indicated that the rationale for this recommendation is that the ANZG and Canadian FEQG guideline values are chronic, long-term guidelines. While I agree that the ANZG guideline values are intended to protect against long-term exposure effects, I disagree that a compliance framework based on only 50% compliance with the site-specific guideline values is appropriate for the reasons outlined below.

49. The ANZG guidelines are not intended to be routinely exceeded. Rather, exceedances are expected to be relatively infrequent and should trigger an assessment of the magnitude, frequency, duration and ecological significance of the exceedance. While ANZG recognises that natural variability occurs in environmental systems, the framework does not suggest that chronic guideline values should be exceeded as often as they are achieved. The monitoring section of the ANZG website recommends "that the 95th percentile of the monitoring data for a toxicant be compared with the default guideline value". The guideline value shall be regarded as exceeded if the 95th percentile of the monitoring data, after rounding in accordance with the Australian Standard SAA 2706-2003, to the number of significant figures in the DGV". This recommendation implicitly recognises that guideline values are intended to be met for the majority of observations and that exceedances should be relatively infrequent.

50. A similar principle is evident in the US EPA aquatic life criteria framework. The chronic criterion, referred to as the Criterion Continuous Concentration (CCC), is intended to protect aquatic organisms from long-term exposure effects and is generally expressed as a 4-day average concentration that should not be exceeded more than once every 3 years. This approach combines a concentration threshold with an allowable exceedance frequency, recognising that isolated short-term excursions may occur while preventing prolonged or repeated exposures that could cause ecological harm. Consequently, the US EPA framework anticipates that chronic criteria will be attained for the overwhelming majority of observations and does not support routine exceedance of chronic protection thresholds.

7.3 The Use of the US EPA Iron Criterion

51. I disagree with Dr Hickey that the United States Environmental Protection Agency (US EPA, 1976) iron criterion, whether applied directly (1.0 mg/L) or modified by a factor of two to produce a guideline value of 0.5 mg/L, provides an appropriate basis for establishing a site-specific iron guideline for Buller Plateau streams for the following reasons;

    (a) It is an old guideline value that has not been updated since 1976 and is based on limited field observations from a single, iron-polluted stream in Colorado in 1967, where trout and other fish were absent at iron concentrations above 1.0 mg/L. However, this study is not available online, and information on other water quality

parameters (such as dissolved organic matter (DOM), pH, and other potentially toxic elements) is unavailable.

(b) The scientific relevance of the US EPA iron criterion to the ESE and MFS plateau streams is questionable. The Canadian FEQG review notes that the US EPA criterion is based largely on field observations linking iron concentrations to impacts on fish communities rather than on modern toxicity-based guideline derivation methods. Fish are absent from the ESE and MFS streams and are therefore not the critical receptors requiring protection in these environments. In these circumstances, the use of a fish-based iron criterion is not well aligned with the ecological characteristics of the receiving environment. A guideline derived from site-relevant ecological receptors and contemporary toxicological evidence would provide a more scientifically defensible basis for assessing potential ecological effects in the Buller Plateaux streams

(c) Based on the information available to me, it is unclear whether the water chemistry characteristics of the streams used to derive the US EPA iron criterion are representative of the receiving environments on the Buller Plateaux. The criterion was developed from field observations of unnamed streams in Colorado affected by mining activities rather than a contemporary toxicity-based assessment, and I have not identified sufficient information to determine whether the unnamed Colorado waterways exhibited the same naturally low pH, very low hardness, and elevated DOM concentrations that characterise many of the Buller Plateaux streams.

(d) It is not clear what forms of iron were present in the streams from which the field observations underpinning the US EPA criterion were derived. Iron toxicity, bioavailability, and ecological effects are strongly influenced by speciation, including the relative proportions of dissolved and particulate iron, oxidation state, and the degree of complexation with organic matter. Without this information, it is difficult to determine whether the ecological responses observed in those studies are applicable to the water chemistry conditions present within the Buller Plateaux streams.

(e) In addition, the field-based nature of the US EPA criterion introduces uncertainty regarding the extent to which other environmental stressors may have contributed to the observed ecological effects. From the information available, it is unclear whether the sites used to derive the criterion contained elevated

concentrations of other contaminants, were influenced by acidity or other water quality stressors, or were subject to habitat-related factors that may also have affected fish distributions. Consequently, there is uncertainty regarding the degree to which the observed absence or impairment of fish populations can be attributed solely to iron concentrations.

(f) No clear scientific basis has been provided for applying a safety factor of two to the US EPA (1976) iron criterion. In my opinion, the application of an arbitrary assessment factor is not an adequate substitute for a guideline derived using contemporary ecotoxicological methods. A more scientifically defensible approach would be to establish a guideline value based on an SSD-derived toxicity threshold and then evaluate whether adjustment is required to account for site-specific modifiers such as pH, dissolved organic carbon, iron speciation, water chemistry and the naturally elevated background concentrations observed within Buller Plateau streams. This approach would provide a stronger scientific basis for guideline derivation than applying an unsubstantiated safety factor to an existing criterion.

52. Contemporary guideline frameworks, such as the Canadian FEQG approaches, recognise the importance of toxicity-modifying factors and increasingly incorporate water chemistry adjustments when deriving ecological protection values. The US EPA iron criterion does not contain any equivalent adjustment methodology for pH, DOM, or other relevant water chemistry variables. Consequently, there is considerable uncertainty regarding its applicability to the ESE and MFS receiving environments. In my opinion, greater weight should be given to guideline approaches that explicitly account for site-specific water chemistry and the resulting effects on contaminant bioavailability and ecological risk.

7.3.1 Rationale for Not Using the ANZG (2025) Freshwater Guideline for Iron

53. The recently derived ANZG guideline for iron in freshwater[10] was not proposed in the BPCP-WQ report as the basis for a bioavailability-based guideline for iron. The rationale for the exclusion was not provided within the BPCP-WQ report, but discussion with Dr Hickey revealed that the primary reasons for the exclusion were;

[10] ANZG. (2025). Toxicant default guideline values for aquatic ecosystem protection: Iron in fresh water. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Governments and Australian state and territory governments, Canberra, ACT, Australia

(a) The guideline did not allow site-specific modification for highly acidic/high DOC waters like those encountered in Buller Plateaux streams.

(b) Most of the ecotoxicity studies that were incorporated within the ecotoxicity dataset used to derive the ANZG iron guideline were from ecotoxicity tests undertaken on a limited circumneutral pH range, which may not be appropriate for the site.

54. I acknowledge and generally concur with Dr Hickey's observation that exceedance of the ANZG default guideline value for iron does not necessarily indicate that adverse ecological effects will occur. Based on my own experience conducting water quality and ecological assessments in the lower Waikato catchment, including the Whangamarino Wetland, I have observed freshwater ecosystems that appear to maintain healthy, diverse aquatic communities despite dissolved iron concentrations exceeding the current ANZG default guideline value.

55. In my opinion, this highlights one of the limitations of applying generic guideline values to environments with unusual water chemistry characteristics. In particular, DOC is known to influence iron speciation, bioavailability, and toxicity. Water bodies with elevated DOC concentrations may support higher dissolved iron concentrations while exhibiting a lower level of biological effect than would be predicted from dissolved iron concentrations alone. This observation is consistent with the broader scientific understanding that metal toxicity is not solely a function of concentration but is also influenced by site-specific water chemistry.

56. To better understand why pH and DOC multiple linear regression (MLR) models were not incorporated into the ANZG (2025) iron guideline, I contacted Dr Graeme Batley, who was the principal technical author of the ANZG freshwater iron guideline. Dr Batley advised that there were two primary reasons why ANZG did not adopt the iron bioavailability model developed by Brix et al. (2023)[11], despite that model subsequently being incorporated into the Canadian Federal Environmental Quality Guidelines (FEQGs) for iron.

(a) First, the MLR model was derived from a relatively limited ecotoxicity dataset comprising only three species representing an alga, an invertebrate, and a fish. In Dr Batley's opinion, this dataset was insufficient to provide a sufficiently robust

[11] Brix, K., Tear, L., DeForest, D., Adams, W. (2023) Development of multiple linear regression models for predicting chronic toxicity to aquatic organisms. Environmental Toxicology and Chemistry, 42(6), 1386-1400. https://doi.org/10.1002/etc.5623

basis for incorporation into the ANZG guideline framework, particularly given the importance of demonstrating that bioavailability relationships are broadly applicable across a range of taxonomic groups.

(b) Second, the underlying toxicity dataset did not include Australian species. As ANZG guideline development places considerable emphasis on ensuring that toxicity datasets are representative of Australian and New Zealand aquatic ecosystems, the absence of Australian test species reduced confidence in the resulting bioavailability model's full applicability to Australian and New Zealand receiving environments.

57. It should be noted that Brix et al. (2023) found pH was not a statistically significant modifier of iron toxicity for the freshwater invertebrate (*Ceriodaphnia dubia*), while only a relatively modest pH effect was observed for the alga (*Raphidocelis subcapitata*). In contrast, pH exerted a much stronger influence on iron toxicity for the fish species (*Pimephales promelas*).

58. Given that fish have not been recorded in the upper reaches of Buller Plateaux streams, these findings may suggest that DOC could be a more influential toxicity-modifying factor than pH for the aquatic communities (noting potential effects relating to invertebrates) present at these sites, particularly within the pH range of approximately 5 to 8[12]. However, considerable uncertainty remains because the Brix et al. (2023) analysis was based on a limited number of test species, and the applicability of these relationships to New Zealand stream macroinvertebrates has not been confirmed. Accordingly, toxicity testing with ecologically relevant New Zealand species would be desirable to determine whether pH exerts a comparable influence on iron toxicity in these systems.

59. While I acknowledge these concerns, it is also notable that the Canadian FEQG process considered the available evidence sufficient to support adoption of the Brix et al. (2023) iron bioavailability model. This highlights a key distinction between the two frameworks. The ANZG decision not to adopt the model does not necessarily indicate that the model is scientifically invalid; rather, it reflects a judgement that the available dataset was not yet sufficiently comprehensive or regionally representative to support incorporation into the ANZG guideline framework. In my opinion, this

[12] See Section 8.4.3, iron geochemistry changes below pH 5, with shifts in iron speciation and solubility that may alter both bioavailability and toxicity. Below pH 3 iron (II) becomes the predominant iron species and that may significantly alter iron toxicity.

limitation should be weighed against the fact that the existing ANZG iron guideline does not explicitly account for the influence of pH and DOC on iron bioavailability and toxicity, despite both factors being widely recognised as important controls on iron behaviour in freshwater systems.

60. Overall, given the limitations of the ANZG (2025) iron guideline, particularly its inability to account for the influence of recognised toxicity-modifying factors such as pH and dissolved organic carbon (DOC) on iron bioavailability and toxicity, I understand and largely agree with Dr Hickey's reluctance to recommend the ANZG iron guideline for assessment of the Buller Plateaux streams.

7.3.2 Rationale for Not Using the Canadian (FEQC) Freshwater Guideline for Iron

61. The recently published Canadian FEQG[13] for iron was not adopted within the BPCP-WQ report as the basis for deriving a bioavailability-adjusted iron guideline. While the rationale for its exclusion is not explicitly discussed in the report, discussions with Dr Hickey indicated two principal reasons for this decision.

(a) First, Dr Hickey expressed concerns regarding the inclusion of the mesocosm study undertaken by Cadmus et al. (2018) in the derivation of the Canadian guideline. In his view, the 10-day duration of the mesocosm experiments was insufficient to represent chronic ecological exposure and therefore did not provide an appropriate basis for deriving a long-term guideline value. He considered that chronic toxicity studies should generally be longer, such as 28 days or more, to adequately assess chronic ecological responses. There were also a number of other reasons that Dr Hickey expressed concerns regarding the mesocosm experiments, including the pH of the tests, lack of information on the DOC within the system, the chemical form of the iron being added to the test, and whether it precipitated or remained in the dissolved phase.

(b) Second, Dr Hickey noted that the multiple linear regression (MLR) model incorporated within the Canadian FEQG is only validated for waters with pH values between 6.0 and 8.5. Many of the Buller Plateaux streams have natural pH values below 6.0, commonly occurring between pH 4.5 and 5.5. Dr Hickey also acknowledges that DOC plays a more important role in influencing iron

[13] Environment and Climate Change Canada (ECCC). (2024). Federal Environmental Quality Guidelines – Iron. Gatineau, Quebec: Environment and Climate Change Canada

toxicity than pH. Consequently, application of the Canadian model to these streams would require extrapolation beyond the pH range for which the model was developed and validated. In Dr Hickey's opinion, this introduced sufficient uncertainty to render the model of limited utility for the plateau stream environments

62. The ANZG technical brief for iron also considered the Cadmus et al. (2018)[14] study and provides specific reasons why it was excluded from the ANZG guideline derivation.

63. In my opinion, the key scientific issue is that both approaches have limitations. The ANZG iron guideline does not account for the recognised influence of pH and dissolved organic carbon on iron bioavailability and toxicity, whereas the Canadian FEQG explicitly incorporates these toxicity-modifying factors, but it was developed and validated over a pH range that does not encompass naturally acidic plateau streams. Accordingly, neither approach can be regarded as free from uncertainty when applied to the BPCP receiving environments.

64. While I acknowledge Dr Hickey's concern that the Canadian FEQG iron MLR model has not been validated across the full pH range observed on the Buller Plateau. As noted above, Buller Plateaux streams are characterised by naturally elevated DOC concentrations, making this factor particularly relevant to the assessment of iron toxicity. Further, this limitation is not unique to the iron guideline, as several of the toxicity-modifying factor models presented in Table 10.1 of the BPCP-WQ report were developed and validated over relatively restricted water chemistry ranges. In my opinion, the key consideration is whether the underlying toxicity relationships adequately represent site conditions. Given the importance of DOC in controlling iron speciation and bioavailability, I consider the iron MLR to provide a more scientifically defensible basis for guideline derivation than fixed guideline values that do not account for site-specific water chemistry. As discussed in paragraph 62, there is also evidence that pH may not be a strong modifier of iron toxicity for macroinvertebrate species relative to other factors such as DOC

65. However, of the evaluable freshwater guideline values for iron in freshwater, it is my view that the Canadian FEQG are superior to either the ANZG (2025) or US EPA

[14] Cadmus, P., Brinkman, S., May, M. (2018) Chronic Toxicity of Ferric Iron for North American Aquatic Organisms: Derivation of a Chronic Water Quality Criterion Using Single Species and Mesocosm Data. Archives of Environmental Contamination and Toxicology, 74(4), 605–615. https://doi.org/10.1007/s00244-018-0505-2

(1976), and therefore should have been selected in preference to the US EPA iron guideline.

7.3.3 Iron Guideline Value

66. Iron geochemistry (summarised in Appendix B) should be a primary consideration in the selection of an appropriate iron guideline value for the Buller Plateaux streams. The solubility, speciation (ferrous or ferric iron), bioavailability and toxicity of iron are strongly controlled by pH, DOC and redox conditions, and therefore the toxicity associated with a given total iron concentration cannot be assumed to be constant across different water bodies. Given the naturally acidic and high-DOC characteristics of the plateau streams, the applicability of iron guideline values derived from circumneutral waters is uncertain. In my opinion, any iron guideline adopted for these streams should explicitly account for the influence of local geochemical conditions on iron bioavailability and toxicity.

67. At present, the Canadian FEQG framework is the only freshwater iron guideline that incorporates both pH and DOC as toxicity-modifying factors.

68. Dr Hickey has raised concerns regarding the inclusion of mesocosm-derived studies within the ecotoxicological dataset used to derive the Canadian FEQG iron guideline and associated MLR model. In my opinion, this concern does not undermine the underlying scientific rationale for incorporating pH and DOC into the derivation of guidelines. If the inclusion of mesocosm data is deemed inappropriate, the affected studies could be removed, and the guideline recalculated using only studies that meet the agreed-upon data quality and study selection criteria. Such an approach would remove the influence of the mesocosm data on the derived guideline values and MLR relationships, while retaining the benefit of incorporating recognised toxicity-modifying factors into the assessment framework.

69. If modification of the Canadian FEQG approach is not practicable, an alternative would be to apply site-specific geochemical speciation modelling using a combined WHAM–PHREEQC framework[15]. WHAM can be used to estimate the partitioning of iron between free ionic, organically complexed, and colloidal forms, while PHREEQC can be used to evaluate iron speciation, iron–sulphate complexation, and mineral

[15] This approach is consistent with the recommendations in the ANZECC (2000) guidelines and Chapman, J., Warne, M., Patra, R (2001) Considerations when applying the revised toxicant guidelines. Australian Journal of Ecotoxicity, 7:154-174

saturation states across a range of pH and DOC conditions. Together, these models could be used to develop site-specific lookup tables describing the expected concentrations of dissolved iron, and precipitated iron phases under representative plateau stream conditions. Such an approach would provide a more mechanistic assessment of iron bioavailability and solubility than a single concentration-based guideline and would explicitly account for the strong influence of pH, DOC, and competing metals on iron chemistry in these naturally acidic waters.

70. In the BPCP-WQ report, Dr Hickey proposes a dissolved iron criterion of 0.5 mg/L. However, the memorandum does not provide the toxicological basis, geochemical rationale, or derivation methodology used to determine this value. As a result, I am unable to independently evaluate whether 0.5 mg/L represents an appropriate site-specific trigger value for iron in the Buller Plateaux streams.

71. However, it is notable that Appendix A, Table A1 of the ANZG (2025) Technical Brief for Iron in Freshwater reports that exclusion of the mesocosm-derived insect toxicity data from Cadmus et al. (2018) increases the resulting chronic guideline value from approximately 0.25 mg/L to 0.499 mg/L. Modifying the Canadian FEQG to remove the Cadmus et al. (2018) value would likely increase the iron guideline value, perhaps to approximately 0.5 mg/L.

72. Consequently, it is conceivable that a dissolved iron guideline value of approximately 0.5 mg/L could be scientifically defensible in waters characterised by elevated DOM, such as the Buller Plateaux streams. However, in the absence of a documented derivation, supporting ecotoxicological analysis, or geochemical modelling demonstrating how the proposed value was determined, I am unable to conclude whether the proposed criterion is justified for these receiving environments.

73. I agree with Dr Hickey that further iron toxicity studies would be valuable and support the recommendation that additional work be undertaken. In particular, I recommend:

(a) Undertaking bench-scale studies to determine iron floc formation thresholds in Buller Plateau waters across a range of pH, dissolved organic carbon (DOC), hardness and temperature conditions. The kinetics of iron floc formation should also be assessed to determine the time-response of floc development and ensure the results are applicable to transient discharge and mixing events;

(b) Undertaking toxicity testing using native aquatic species (for example, Deleatidium spp.) to establish iron toxicity thresholds under environmentally relevant exposure conditions representative of Buller Plateau receiving waters.

If recommendation (b) is adopted, testing should cover the range of pH and DOC conditions characteristic of Buller Plateaux streams, as these parameters are expected to significantly influence iron speciation, bioavailability, and toxicity. I further recommend that additional studies be undertaken to quantify the interaction between iron and naturally occurring dissolved organic matter, including estimation of iron-organic matter binding relationships. Such information would improve the scientific basis for assessing iron bioavailability in Buller Plateaux waters and could be incorporated into future WHAM-PHREEQC modelling to support site-specific guideline development and effects assessments.

8 Summary

74. In my opinion, the methodology used to derive the site-specific turbidity guideline values is scientifically robust and appropriate, having been developed using a weight-of-evidence approach that incorporates laboratory toxicity studies, field ecological observations, site-specific baseline data, and relevant New Zealand guidance.

75. I consider the use of toxicity-modified site-specific guideline values for metals, incorporating pH and dissolved organic carbon as toxicity-modifying factors, to be an appropriate and scientifically defensible approach for the Buller Plateaux streams. This methodology is consistent with contemporary ecotoxicological science and the site-specific assessment principles of the ANZG water quality framework.

76. However, I have concerns regarding the proposal to assess compliance with chronic site-specific guideline values using median concentrations, effectively allowing exceedance of the guideline values in approximately 50% of monitoring results. In my opinion, this is inconsistent with the intent of the ANZG, the CFEQG, and the US EPA's chronic aquatic life protection frameworks. An alternative approach is to use rolling 30-day median values (including high-bypass events) with no value exceeding the acute criteria, and a rolling 95th-percentile annual average (excluding high-bypass events).

77. Finally, I have concerns regarding the reliance on the USEPA (1976) iron criterion or the newly proposed iron guideline of 0.5 mg/L. In my opinion, these criteria are of uncertain relevance to the naturally acidic, high-DOM, fishless stream environments of the Buller Plateaux upper catchments. The US EPA criterion was developed from limited field observations and has been criticised in subsequent reviews for not being based on a modern ecotoxicological dataset. More recent assessments note that the US EPA criterion was derived largely from observations of fish community responses in an iron-impacted stream in Colorado, rather than from contemporary species-sensitivity or bioavailability-based approaches.

78. Furthermore, there is insufficient information available to determine whether the water chemistry and ecological characteristics of the receiving environments used to derive the US EPA criterion are representative of the Buller Plateaux streams. In contrast, contemporary guideline development frameworks, including the Canadian FEQG approach, explicitly recognise the influence of site-specific water chemistry, including pH and dissolved organic carbon, on iron bioavailability and toxicity.

79. Accordingly, I consider that greater weight should be given to guideline derivation approaches that account for these toxicity-modifying factors and the characteristics of the receiving environment (such as the Canadian FEQC for iron). While a dissolved iron criterion of 0.5 mg/L may ultimately prove to be appropriate for the Buller Plateaux streams, the methodology, toxicological basis, and supporting evidence used to derive this value have not been provided. Consequently, I am unable to independently determine whether the proposed criterion is sufficiently protective of the aquatic ecosystems of the Buller Plateaux streams.

Andrew John Rumsby

17 August 2026

Appendix A: Influence of Dissolved Organic Matter on Metal Speciation, Iron Solubility and Aquatic Toxicity

Introduction

A1. Dissolved organic matter (DOM) is a major component of the water chemistry of Buller Plateau streams and is likely to play a fundamental role in controlling the speciation, solubility, mobility, bioavailability and toxicity of dissolved metals. DOM in peatland and forested catchments is derived primarily from the decomposition of plant material and consists largely of humic substances (humic acids and fulvic acids), together with lesser quantities of tannins and other plant-derived organic compounds. These compounds contribute to the naturally acidic character of Buller Plateau waters and strongly influence the environmental behaviour of metals, particularly iron and aluminium.

A2. Metal binding by DOM occurs through a variety of functional groups, including carboxylic and phenolic groups, which form complexes with dissolved metals. These interactions can substantially reduce the concentration of free metal ions, thereby reducing metal bioavailability and toxicity to aquatic organisms. Dissolved organic matter can also maintain metals in solution at concentrations that would otherwise be expected to precipitate, particularly under acidic conditions.

Major Components of Dissolved Organic Matter

Table A1 Summary of the Major Components of Dissolved Organic Matter

Property	Humic Acids	Fulvic Acids	Tannins
Origin	Decomposition and humification of plant and microbial material	Decomposition and humification of plant and microbial material	Directly derived from vegetation including leaf litter, bark, wood and peat-forming plants
Molecular size	Large, complex molecules	Smaller and less complex molecules	Small to intermediate-sized polyphenolic compounds
Solubility	Soluble at neutral to alkaline pH; precipitation may occur under acidic conditions	Soluble across virtually all environmental pH conditions	Highly water soluble, particularly in fresh plant leachates

Property	Humic Acids	Fulvic Acids	Tannins
Mobility	Moderate	High	High
Persistence	Highly persistent	Persistent	Generally, less persistent and more readily biodegraded than humic substances
Metal binding	Strong metal complexation and sequestration through multiple binding sites	Strong metal complexation and transport of dissolved metals	Moderate to strong metal complexation, particularly with iron and aluminium
Role in metal transport	Often immobilises metals in soils, sediments and organic colloids	Enhances dissolved transport of metals through aquatic systems	Can enhance dissolved transport through organic-metal complexes
Effect on iron chemistry	Stabilises Fe(III) through complexation and colloid formation	Maintains dissolved iron over a broad range of pH conditions	Forms Fe-tannin complexes that may increase iron solubility
Environmental significance	Major component of soil organic matter and sediment carbon storage	Major component of dissolved organic carbon (DOC) in natural waters	Important contributor to brown-water colour, peatland runoff and forest-derived DOM
Typical abundance in peatland streams	High	High	Variable but locally abundant in some streams

Importance for Buller Plateau Streams

A3. Buller Plateaux streams are naturally acidic and receive substantial inputs of organic matter from peat soils, wetlands and native forest vegetation. Under these conditions, DOM is expected to strongly influence dissolved metal chemistry.

A4. Of the major DOM fractions, fulvic acids are likely to be particularly important because they remain soluble at the naturally low pH values characteristic of Buller Plateaux streams. Fulvic acids can therefore maintain iron and aluminium in dissolved form through complexation, reducing the proportion present as free metal ions. Humic acids also contribute to metal binding but may occur to a greater extent in colloidal or particulate phases. Tannins may provide an additional source of metal

binding and are likely to contribute to the characteristic brown colour of many Plateau streams.

A5. Naturally occurring organic acids associated with DOM also influence stream chemistry by releasing hydrogen ions and organic anions. Consequently, they contribute to the naturally low pH observed in many streams while also increasing conductivity. However, their contribution to conductivity is generally much smaller than that associated with inorganic ions such as sulphate, chloride and dissolved metals commonly associated with acid mine drainage.

Influence on Metal Toxicity

A6. The toxicity of many metals is largely controlled by their chemical speciation rather than their total dissolved concentration. Numerous studies have demonstrated that low pH, hardness, alkalinity and dissolved organic carbon are major controls on metal bioavailability, with DOC often being one of the most important modifying factors in organic-rich waters.

A7. Complexation reactions between dissolved metals and DOM generally occur rapidly, often within seconds to minutes, such that metal speciation in natural waters commonly approaches equilibrium conditions.

A8. pH is a key factor controlling the ability of DOM to bind and complex dissolved metals. The metal-binding capacity of DOM is primarily associated with negatively charged functional groups, particularly carboxylic and phenolic groups present in humic substances, fulvic acids and tannins. As pH increases, these functional groups progressively deprotonate, increasing their negative charge and enhancing their capacity to bind metal cations. Conversely, at low pH, hydrogen ions compete with metals for the same binding sites, reducing the number of available complexation sites and decreasing the overall metal-binding capacity of DOM.

Uncertainty Regarding the Nature of DOM on the Buller Plateau

A9. At present, the composition and characteristics of dissolved organic matter within Buller Plateaux streams have not been comprehensively characterised. Consequently, the relative importance of humic acids, fulvic acids, tannins and other organic compounds remain uncertain.

A10. I understand that research undertaken by Earth Sciences New Zealand (formerly NIWA) has indicated that DOM derived from mānuka-dominated vegetation may exhibit substantially greater metal-binding capacity than some DOM derived from other plant species. If confirmed for Buller Plateaux streams, this would have important implications for metal speciation, bioavailability and toxicity and could further support the need for site-specific approaches that explicitly consider DOM characteristics rather than relying solely on generic guideline values.

Application of Geochemical Models

A11. For naturally acidic streams with elevated dissolved organic matter, such as many peatland, forested and historic mining catchments, geochemical models that explicitly simulate metal-DOM interactions provide a valuable tool for understanding metal speciation and bioavailability.

A12. The Windermere Humic Aqueous Model (WHAM) is widely recognised as one of the most appropriate approaches for predicting metal speciation in waters where DOM complexation is a dominant control on metal chemistry. WHAM is an equilibrium model and therefore assumes that metal-organic binding reactions have reached equilibrium. However, this assumption is generally consistent with the rapid kinetics observed for many metal-DOM complexation reactions under natural stream conditions. Furthermore, most contemporary ecotoxicological datasets, bioavailability models and multiple linear regression (MLR) toxicity models similarly assume equilibrium conditions or are based on toxicity tests conducted under conditions that approximate equilibrium. Accordingly, the use of equilibrium-based approaches such as WHAM is, in my opinion, appropriate for assessing the influence of dissolved organic matter on metal bioavailability and toxicity in Buller Plateaux streams.

Conclusion

A13. In my opinion, dissolved organic matter is likely to be one of the principal controls on metal speciation, bioavailability and toxicity in Buller Plateaux streams. Its influence is particularly important for iron and aluminium, where complexation by organic matter can substantially modify solubility and toxicity. Given the naturally acidic, organic-rich nature of these receiving environments, consideration of DOM is essential when interpreting ecological risk and deriving site-specific water quality guideline values. The current lack of detailed characterisation of Buller Plateaux

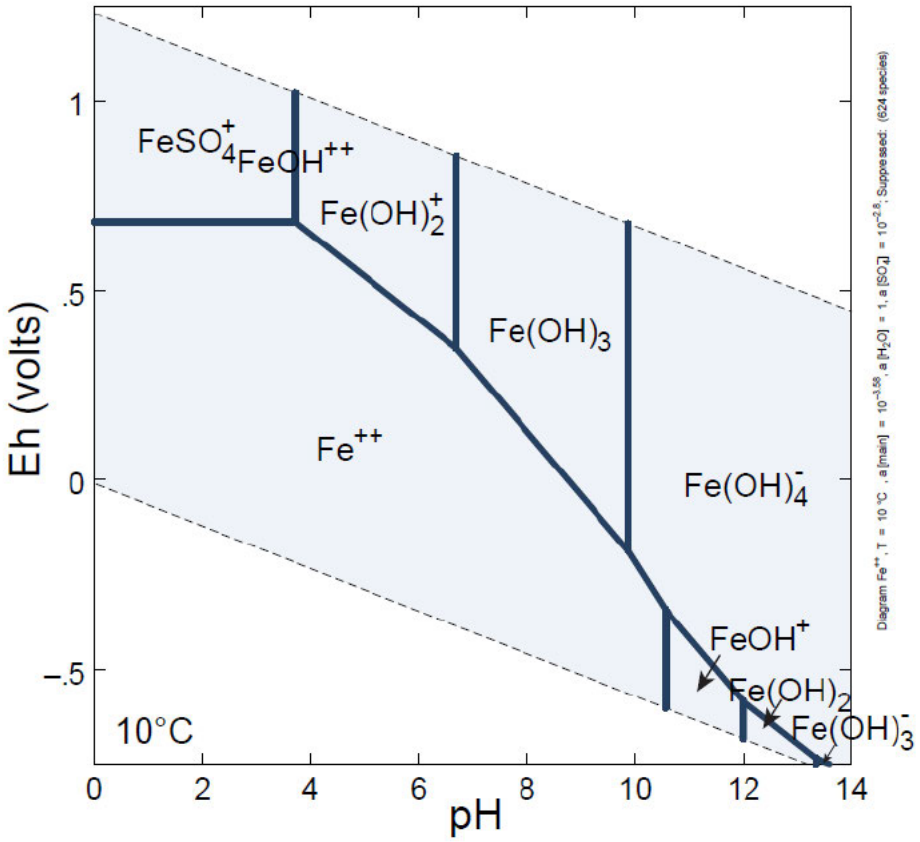
DOM represents a source of uncertainty but does not diminish the importance of incorporating DOM effects into guideline derivation and ecological assessments.

Appendix B: Iron geochemistry in natural waters that may influence toxicity

B1. Iron is an essential micronutrient for aquatic organisms; however, adverse effects can occur when concentrations become sufficiently elevated. Under strongly acidic conditions (typically pH <3), dissolved iron is predominantly present as ferrous iron (Fe(II)). As pH increases under oxic conditions, ferric iron (Fe(III) becomes the dominant oxidation state, and ferric hydroxides and oxyhydroxides become thermodynamically stable at approximately pH 3–4 and above (See Figure A1). Consequently, in most well-oxygenated surface waters, iron occurs primarily as Fe(III) species, although the distribution among dissolved, colloidal, organically complexed, and particulate forms is strongly influenced by pH, redox conditions, and dissolved organic matter.

Figure B1 – Pourbaix Diagram for Inorganic Iron Speciation within the Buller Plateaux Streams based on site geochemistry (Generated using Geochemical Workbench-note assumed conditions on side of figure)[16].

[16] Note; Dissolved Organic Carbon cannot be displayed on Eh-pH diagrams using Geochemical Workbench; the binding of the DOC depends on the particular functional groups present on the DOC and the type of natural organic present, which is too complex to display on Pourbaix diagrams



B2. The oxidation of pyrite associated with acid mine drainage (AMD) initially releases ferrous iron (Fe(II)) to solution (See equation 1). However, acidophilic microorganisms commonly present in AMD environments, including Acidithiobacillus ferrooxidans, Leptospirillum ferrooxidans, and Acidithiobacillus thiooxidans, can substantially accelerate iron cycling by oxidising ferrous iron to ferric iron (Fe(III)). These microbial processes increase iron oxidation rates by several orders of magnitude compared with purely abiotic reactions and are a key factor controlling iron speciation in AMD-affected waters.

$$FeS_2+(7/2)O_2+H_2O \rightarrow Fe^{2+} + 2SO_4^{2-}+2H^+ \quad \text{equation 1}$$

B3. Consequently, while Fe(II) may be the primary form of iron initially released from pyrite weathering, it is unlikely to remain the dominant dissolved species for extended periods in well-oxygenated plateau streams. Once pH increases above approximately 3 and dissolved oxygen is available, biologically mediated oxidation

can rapidly convert Fe(II) to Fe(III). The ferric iron subsequently undergoes hydrolysis and forms ferric hydroxide and oxyhydroxide precipitates, although the extent of precipitation may be moderated by dissolved organic matter through complexation and colloid stabilisation. As a result, the iron present within the plateau streams is likely to comprise a mixture of dissolved Fe(II), dissolved and organically complexed Fe(III), colloidal iron, and precipitated ferric hydroxides, with the relative proportions controlled by pH, redox conditions, dissolved oxygen concentrations, and DOM.

B4. Dissolved iron may contribute to direct toxic effects, whereas colloidal and precipitated iron can cause physical impacts, including coating of biological surfaces, smothering of benthic habitats, impairment of gill function, and reduced habitat quality for aquatic organisms. Consequently, measurements of total or dissolved iron alone may not adequately represent ecological risk because toxicity is influenced not only by concentration but also by iron speciation, bioavailability, and the presence of precipitated iron phases.

B5. The bioavailability, solubility and toxicity of Fe(III) are influenced by several water chemistry variables, including pH, redox conditions, DOM, and sulphate concentrations. DOM, particularly humic and fulvic substances, can strongly complex Fe(III), reducing the activity of free ferric ions and potentially reducing bioavailability while simultaneously increasing apparent dissolved iron concentrations through complexation and colloid stabilisation. Experimental and field observations indicate that iron solubility may increase substantially in waters containing moderate to high concentrations of DOM relative to predictions based solely on inorganic Fe–Eh–pH relationships[17]. Under low-DOM conditions, Fe(III) concentrations are generally constrained by ferric hydroxide solubility above pH 4–5. However, in waters with elevated DOM, dissolved and colloidal iron concentrations may range from hundreds of micrograms per litre to several milligrams per litre. Consequently, dissolved iron concentrations within the range of 500–2,000 µg/L are plausible in peat-influenced plateau streams containing elevated dissolved organic carbon, and dissolved iron concentrations of up to 350 µg/L indicate that iron-DOM complexes are being formed within the Buller Plateau streams (See Attachment 8 in BPCP-WQ report).

[17] See Appendix D of the BPCP-WQ, which shows the relationship of dissolved iron and DOC in several streams on the Buller Plateaux.

B6. Although Fe(III) has a strong affinity for humic and fulvic binding sites, aluminium can also form stable complexes with DOM and may compete with iron for available ligands. In AMD-affected waters containing elevated dissolved aluminium concentrations, competition for DOM binding sites may reduce the proportion of iron present in organically complexed forms and promote the formation of inorganic iron species. As aluminium concentrations increase, particularly under acidic conditions where both metals are relatively soluble, the relative importance of competitive complexation processes is expected to increase. In addition, increasing pH will favour hydrolysis and precipitation of Fe(III), further limiting dissolved iron concentrations.

B7. At low pH (approximately 3 to 4), competition from hydrogen ions progressively reduces the availability of ionised functional groups on dissolved organic matter (DOM), thereby reducing the capacity of DOM to bind both Fe(III) and aluminium. Under these acidic conditions, sulphate also becomes an increasingly important ligand, forming aqueous complexes with both iron and aluminium. Consequently, the influence of DOM on iron speciation, solubility and bioavailability is expected to decline as pH decreases below approximately 4, while inorganic iron-sulphate complexes become increasingly important between pH 4 and 5. This has implications for the application of the Canadian FEQG iron MLR model. The MLR model was developed principally to account for the influence of pH and dissolved organic carbon on iron bioavailability and does not explicitly account for elevated sulphate concentrations or iron-sulphate complexation that may occur in AMD-affected waters. Accordingly, the applicability of the MLR under Buller Plateau conditions may warrant further evaluation. In my opinion, geochemical modelling should be undertaken to determine the significance of iron-sulphate complexes within the pH range of approximately 4.5[18] to 5.0. Under strongly oxidising conditions and at pH values below about 4, iron-sulphate complexes may become an increasingly important component of dissolved iron speciation (See Figure B1).

B8. To understand the solubility and bioavailability of iron in low pH, high aluminium, high sulphate and high DOM waters, geochemical modelling using Windermere Humic

[18]The proposed minimum compliance pH is 4.5. Consequently, the relevance of geochemical modelling that focuses on iron-sulphate speciation at pH values below 4.5 is limited, as such conditions would fall outside the proposed operating range. As pH increases above approximately 4.5 under oxic conditions, iron-sulphate stability fields progressively contract and iron hydrolysis becomes increasingly important, promoting the formation of ferric oxyhydroxide precipitates such as schwertmannite and, ultimately, goethite, while dissolved sulphate increasingly occurs as a free ion.

Acid Model (WHAM version 6 or 7) combined with PHREEQC for detailed iron sulphate speciation is recommended.