Site-Specific Water Quality Guidelines for the Hotham River

Desktop Review and Risk Assessment of Modelled Discharge

Newmont Boddington Gold

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Revision Record

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Basis of Report

This report has been prepared by SLR Consulting Australia (SLR) with all reasonable skill, care and diligence, and taking account of the timescale and resources allocated to it by agreement with Newmont Boddington Gold (the Client). Information reported herein is based on the interpretation of data collected, which has been accepted in good faith as being accurate and valid.

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Executive Summary

Newmont Boddington Gold (NBG) operates the Boddington Gold Mine (BGM), located 17 km northwest of Boddington. The current basement ore mining and processing operation was commissioned in 2009. During the historical oxide mining period (1980s-2001) and during the current mining period tailings have been stored in several co-located Residue Disposal Areas (RDA). Tailings deposition to the current RDA is set to cease from December 2029 and this facility will commence closure in 2030. NBG has commenced detailed studies for a second RDA (RDA2), located in Saddleback Tree farm, to be re-referred to the Environmental Protection Authority (EPA) in early December 2024. The RDA2 is designed to receive all tailings until the end of mine life in 2041. Review of the site water balance is underway and a positive site water balance is forecast over a five year period when pit dewatering aligns with commencement of operation of RDA2 (2029 to 2034), and this period will need to be managed differently. The preferred management strategy includes blended treated seepage and untreated mine dewatering water in existing dams prior to discharge to the Hotham River. As the Hotham River has been identified as a potential discharge location, NBG need an understanding of water quality required for discharge to inform water treatment investigations. Opportunities for seasonal and year-round discharge into Hotham River are being investigated as part of feasibility studies.

As part of feasibility studies, NBG have commissioned SLR to review aquatic fauna values of the Hotham River, describe baseline water quality, develop site-specific guideline values (SSGVs) protective of resident aquatic fauna values, and assess likelihood of exceedances of SSGVs occurring under discharge of current modelled discharge water quality, to identify potential analytes of concern (PAoC). Guideline values presented in this technical report will be provided as supporting information with referral documents required under the Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act) and Part IV of the *Environmental Protection Act 1986* (EP Act).

Key findings of the aquatic fauna desktop review:

A comprehensive desktop assessment of aquatic fauna was conducted, incorporating all available scientific and grey literature, and publicly available fauna records databases. The key feature defining the aquatic ecosystem values of the Hotham River is elevated salinity as a legacy impact of agricultural clearing. Nevertheless, the river supports a diverse assemblage of freshwater fauna tolerant to current levels of salinity, including:

- A moderately diverse assemblage of aquatic macroinvertebrates, including a suite of south-west endemic species;
- Six species of native fish, including the freshwater cobbler *Tandanus bostocki*, southwest Western Australia's largest freshwater fish;
- Three species of native crayfish;
- South-western snake necked turtle *Chelodina oblonga*, listed on the IUCN Redlist of Threatened Species as Near Threatened;
- Rakali (or water rat, *Hydromys chrysogaster*), DBCA Priority 4;
- Estuarine mussel *Fluviolanatus subtorta*, a widespread species known to migrate upstream in rivers affected by secondary salinisation.

Background water quality and development of interim SSGVs:

Long-term monitoring data supplied by NBG was compared against ANZG (2018) default guideline values (DGVs) for 95% species protection, to determine whether any analytes occur at elevated background levels, and thus whether SSGVs are warranted. Recommended DGVs/SSGVs for each analyte were based on the following:

- The ANZG (2018) DGV for 95% species protection, or,
- The 80th percentile value of background water quality data;

Where the 80th percentile concentration of the background water quality is below the ANZG 95% DGV, or data are insufficient, it is recommended to retain the 95% DGV. Where the 80th percentile value is above the 95% DGV, or a DGV is not available, the 80th percentile value is recommended for use as the interim SSGV, to detect deviations from the pre-discharge norm. A summary of key analyte background concentrations elevated against DGVs are presented in Table E1, and interim SSGVs for seasonal and year-round application are presented in section 6.3.7.

High salinity measures electrical conductivity (EC) and total dissolved solids (TDS) were the predominant feature defining water quality of the Hotham River, otherwise, waters were generally of good quality. In most cases background analyte concentrations were below the ANZG (2018) DGVs for 95% species protection, applicable to moderately disturbed systems. Concerning toxicants, for all analytes with sufficient background data available, the 80th percentile values were below the DGVs or laboratory limits of reporting (LORs). Most stressors were also below DGV, with the exception of EC, nitrite/nitrate-N (N_NOx) and total nitrogen, for which interim SSGVs for seasonal and year-round discharge were derived.

Since the 2018 guidelines, updated guidance around toxicity modifying factors including pH, dissolved organic carbon (DOC) and hardness have become available for several toxicants, including ammonia, copper, nitrate and zinc. The interim SSGVs derived for the Hotham River include application of relevant toxicity modifying factors, however the available data for these factors (particularly hardness) is limited and should be incorporated into regular monitoring going forward.

Table E1. Summary of analytes above the ANZG (2018) 95% species protection DGV during long-term monitoring at the Hotham River, and recommendations applied to interim SSGVs.

Analyte	Background concentrations	Recommendation
Aluminium	Occasional spot elevations in winter months, likely due to inflows from tributary creeks draining upland catchments rich in bauxite. 80th percentile < 95% DGV	Retain ANZG (2018) 95% DGV (Section 6.3)
Cobalt	80th percentile below DGV (and below LOR, <0.001 mg/L). Occasional records above DGV, with 95th percentile above DGV in both wet and dry season. Records >LOR recorded from Marradong GS614224, indicating a local source. Otherwise, low background concentrations implied.	Retain ANZG (2018) DGV (Section 6.3.1)
Electrical conductivity (and TDS)	EC of the Hotham River is orders of magnitude above the DGV. Nuanced seasonal changes in EC were detected, in accordance with various stages of the flow regime. Seasonal cycles in EC are likely to be ecologically significant, and should be maintained under discharge.	Application of a standard SSGV is not recommended, due to seasonal complexity in background EC. Rather, EC/TDS should be monitored instantaneously against upstream condition (Section 6.3.6).
Cyanide - free	Laboratory LORs mostly too high for comparison to DGVs, or otherwise below LOR.	Retain ANZG (2018) 95% DGV (Section 6.3.1)
Nitrite/nitrate (N_NOx) (stressor)	Seasonal differences detected, likely in accordance with flow regime and catchment runoff. 80th percentiles greater than DGV in both wet and dry.	Seasonal SSGVs based on 80th percentile values for wet and dry seasons. (Section 6.3.3)
Total nitrogen	Seasonal differences detected, likely in accordance with flow regime and catchment runoff. 80th percentiles greater than DGV in wet season, but not during dry season.	Interim SSGV applied to the wet season (based on 80th percentile wet season data); ANZG DGV applied to the dry season. (Section 6.3.3)
Selenium	Occasional spot measurements above DGV, however 80th percentile < DGV in both seasons	Retain ANZG (2018) 95% DGV (Section 6.3)
Zinc	Few elevations above modified DGV, based on draft updated guidance recommending modification using local pH, hardness and dissolved organic carbon (ANZG 2024b)	Modified 95% DGV applied to wet and dry seasons, taking into account seasonality in pH. (Section 6.2; Section 6.3.2) Draft updated Zn guidelines require paired DOC and hardness data be available. Full operational SSGVs possible once sufficient paired DOC and hardness data available.

Hazard assessment of post-dilution PAoC:

Discharge quality estimates were provided for 21 analytes for use in hazard analyses, and of these, nine are modelled to occur in concentrations above the proposed interim SSGVs at the point of discharge. Order of dilution analyses found that the majority were unlikely to pose actual risk to the receiving environment, under the nominal dilution rate of 5:1[1] and maximum of 2,000kL/hr (555 L/sec). Post-dilution concentrations of aluminium, copper, molybdenum, ammonia, and nitrate N_NO$_3$ as a direct toxicant (not as a stressor) were well below SSGV in

[1] A 5:1 dilution rate refers to discharge equivalent to 20% of natural catchment flows, additional to catchment flows.

each instance, including under elevated (80^{th} percentile) background concentrations. Therefore, these analytes are not considered PAoC, however regular monitoring is strongly advised.

Cobalt and nitrite/nitrate (N_NO_x; as a stressor) remained as PAoC at post-dilution concentrations, and would be likely to cause sustained exceedances under the nominal discharge scenario (5:1 dilution rate), and more conservative scenarios, in both wet and dry seasons.

Key recommendations:

Based on current knowledge of background water quality conditions of the Hotham River, and the hazard assessment using modelled discharge water quality provided by NBG:

- It is recommended that mine discharge only occur during the wet season months (June to October), when flows are above the median recorded at the Hotham Weir. Discharge during the dry season or during low flows would ideally be avoided.
- Two key analytes of potential concern were identified, cobalt (toxicant) and nitrite/nitrate-N (stressor). Reduced concentrations of these analytes in discharge water would reduce potential risk to the receiving environment.
- Further investigation into discharge plume modelling and extent of mixing zones, to determine the spatial extent of discharge influence on the river, taking into account the distinct seasonality of the Hotham River.

Further water quality data requirements to support hazard assessment and future monitoring:

- Additions to the regular water quality monitoring suite for the Hotham River have recently commenced including hardness, dissolved organic carbon, nitrite/nitrate-N, and free cyanide at an LOR < 0.004 mg/L, including monthly replicates from monitoring sites on the Hotham River. These data will be incorporated into future updates of SSGVs. Ideally, the ANZG (2018) recommends 24 months of data for development of baseline SSGVs.
- Laboratory LORs for toxicants have been recently reviewed to ensure sufficient baseline and monitoring data is comparable to ANZG (2018) DGVs.
- To support hazard assessment, modelled discharge concentrations for free cyanide, electrical conductivity, total dissolved solids and total suspended solids should be provided.
- Implementation of an aquatic fauna baseline study including macroinvertebrates, to commence 3 years prior to the commencement of discharge, to underpin monitoring throughout discharge operations. Depending on the discharge location, baseline surveys of fish may also be required.
- Predicted discharge quality is presently under review, however if cobalt concentrations continue to be PAoC then ecotoxicity testing for cobalt on a range of south-west species from a range of trophic levels should be considered. Ecotoxicity testing is the best way to predict likely faunal toxicity thresholds of cobalt in the Hotham River, and to better predict likely consequences of exceeding thresholds.

Table of Contents

Tables in Text

Figures in Text

1.0 Introduction

Newmont Boddington Gold (NBG) operates the Boddington Gold Mine (BGM), located 17 km northwest of Boddington. Open pit oxide mining and stockpile processing was undertaken at the site from 1987 to 2001. After a period of care and maintenance, NBG commenced construction of a large-scale open pit mining operation in 2006 to exploit the hard rock 'basement' ore body. The current mining and processing operation was commissioned in 2009. During the oxide mining period and during the current mining period tailings have been stored in several co-located Residue Disposal Areas. Tailings deposition is currently to the F1/F3 Residue Disposal Area (RDA) (above ground valley-fill tailings storage facilities), and is set to reach maximum storage capacity in Q4 2029 and will enter rehabilitation and closure in 2030. NBG has commenced detailed studies for a second RDA (RDA2), located in Saddleback Tree farm, to be re-referred to the Environmental Protection Authority (EPA) in early December 2024. The RDA2 is designed to receive all tailings until the end of mine life in 2041. A positive site water balance is forecast over a five year period (2029 to 2034) when dewatering of one of the open pits coincides with commissioning of RDA2. Over this period, NBG are investigating options to discharge excess water to the environment. Before discharge to the environment is possible, the location of the discharge and the quality of the water to be discharged are issues for consideration. The preferred management strategy includes blending treated seepage (permeate) water from the proposed RDA2 with untreated mine pit dewatering water in existing dams/pits, and then discharge of the blended water to the Hotham River. The quality and quantity of the water to be discharged, as well as opportunities for seasonal versus year-round discharge into Hotham River are being investigated as part of feasibility studies.

Baseline aquatic fauna surveys of the Hotham River were completed from 2011 to 2012 focussed on ecological water requirements and environmentally sustainable yields for the river (WRM 2011). Ongoing Hotham River environmental monitoring includes biennial aquatic fish surveys and streamflow at the Water Intake (HRBP1) and Marradong Road Bridge (GS614224) gauging stations (Figure 1). As part of feasibility studies, NBG have commissioned SLR to review aquatic fauna values of the Hotham River, describe baseline water quality of the Hotham River and develop site-specific guideline values (SSGVs) that protect resident aquatic fauna values. Then, a hazard analysis was conducted based on modelled concentrations of potential analytes of concern (PAoCs) in mine derived water, and assessment of likelihood of exceedances of SSGVs occurring under seasonal and year-round discharge scenarios.

2.0 Scope of Work

As the Hotham River has been identified as a potential discharge location, Newmont need an understanding of water quality required for discharge to inform water treatment investigations. This technical report outlines trigger values to be provided as supporting information with referral documents required under the Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act) and Part IV of the WA Environmental Protection Act 1986 (EP Act).

Specifically, this scope of work included:

- Desktop review of the aquatic ecosystem values of the Hotham River based on published studies, baseline field survey and aquatic fauna monitoring reports documenting water quality, habitat, and aquatic fauna.
- Analysis of long-term water quality monitoring data for the Hotham, with an assessment of these data against default water quality guideline values for ecosystem protection (DGV)

(ANZG 2018), and derivation of interim site-specific guideline values (SSGVs) for the Hotham River as a receiving environment.

- Recommend interim SSGVs for the Hotham River to protect aquatic ecosystem values under seasonal and year-round discharge to the Hotham River.
- A hazard analysis of potential impacts to aquatic fauna of mine water discharge based on modelled quantity and quality of mine discharge, using default DGVs and interim SSGVs. The assessment also includes evaluation of seasonal risk by order of magnitude dilution tests of discharge against Hotham River streamflow flow data (sensitivity analysis).
- Recommendations on quantity, quality and seasonality of mine discharge to the Hotham River.

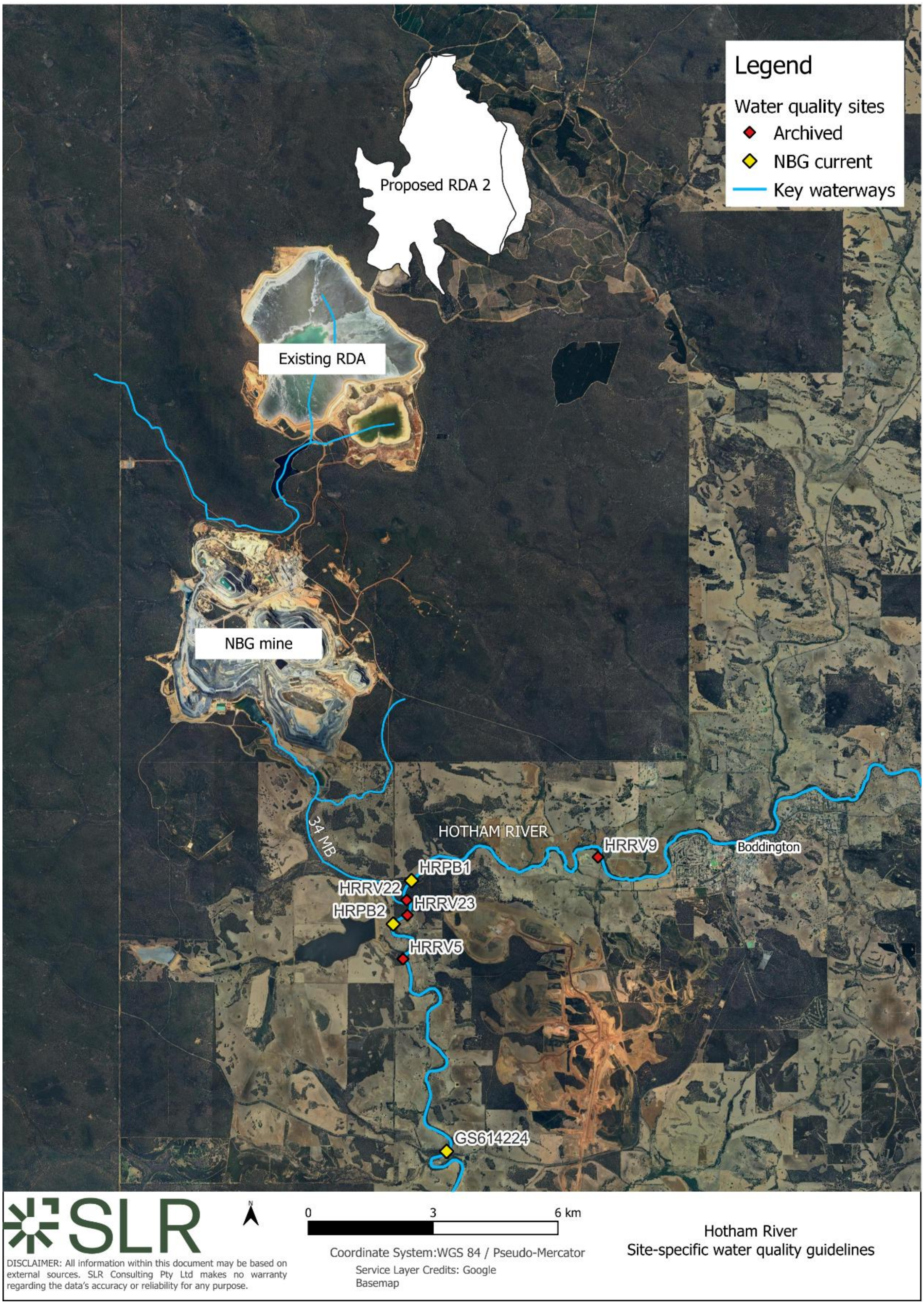

Figure 1. The Hotham River study area in vicinity of the NBG mine and RDAs. Water quality monitoring sites shown, including historic archived data included in long term analysis (pre – 1990) and current NBG monitoring sites. HRPB1 = Hotham Weir gauging station, GS614224 = Marradong Weir gauging station.

3.0 Guidance and general approach

3.1 EPA Environmental Factor Guideline: *Inland Water*

Derivation of water quality guidelines and hazard analyses for the Hotham River were conducted in accordance with the EPA Environmental factor guideline *Inland Water,* broadly defined as encompassing "the occurrence, distribution, connectivity, movement, and quantity (hydrological regimes) of inland water including its chemical, physical, biological and aesthetic characteristics (quality)" (EPA 2018).

Inland waters are considered to include groundwater systems, wetlands, estuaries, and any river, creek, stream or brook (and its floodplain), including systems that "flow permanently, for part of the year or occasionally, and parts of waterways that have been artificially modified" (EPA 2018). Thus, the EPA factor is considered to include all inland waterways irrespective of duration, frequency or volume of flow or inundation. The objective of this factor is "to maintain the hydrological regimes and quality of groundwater and surface water so that environmental values are protected" (EPA 2018). Environmental value is defined under the Environmental Protection Act 1986 as a beneficial use or an ecosystem health condition. Aquatic fauna and the ecological processes that support them are specifically listed in the revised Environmental Factor Guideline as one of the ecosystem health values that must be considered as part of the EIA process (EPA 2018).

3.1.1 Technical guidance

Desktop review, water quality and hazard analyses, and development of proposed SSGVs for the Hotham River were conducted using the following guidance:

- 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 (ANZG 2018);
- Australian Government 2018, Charter: National Water Quality Management Strategy (NWQMS), Department of Agriculture and Water Resources, Canberra, March. CC BY 3.0 (AG 2018);
- Batley, GE, van Dam, RA, Warne, MStJ, Chapman, JC, Fox, DR, Hickey, CW and Stauber, JL 2018. Technical rationale for changes to the Method for Deriving Australian and New Zealand Water Quality Guideline Values for Toxicants. Prepared for the revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Governments and Australian state and territory governments, Canberra, ACT, 49 pp (Batley et al. 2018);
- Warne MStJ, Batley GE, van Dam RA, Chapman JC, Fox DR, Hickey CW and Stauber JL 2018. Revised Method for Deriving Australian and New Zealand Water Quality Guideline Values for Toxicants – update of 2015 version. Prepared for the revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Governments and Australian state and territory governments, Canberra, 48 pp (Warne et al. 2018).

In addition, Australia's NWQMS provides authoritative guidance on the management of water quality in Australia and New Zealand (ANZG, 2018). To protect the community values of waterways (aquatic ecosystems and cultural and spiritual values), the Water Quality Management Framework (WQMF) applies a weight of evidence (WoE) process to collect, analyse and evaluate a combination of different qualitative, semi-quantitative or quantitative lines of evidence (LoE) to make an overall assessment of water quality and its associated management. Therefore, in accordance with the WQMF (ANZG 2018), water quality (physical and chemical stressors and toxicants) and aquatic fauna receptors (e.g. phytoplankton, diatoms, hyporheic fauna, microinvertebrates, macroinvertebrates, and fish) can be used to characterise and monitor ecosystem health condition.

4.0 Environmental Setting

4.1 Climate and Rainfall

The Hotham River is located in the south-west region of Western Australia (SWA). The region has a mediterranean climate, typified by hot dry summers and mild winters. Rainfall tends to be highly seasonal, falling primarily in the winter to early spring months (June through September), with little rainfall over the summer dry season with the exception of occasional summer storms. As such, many low order (i.e. headwater) streams and rivers in the region tend to have seasonally intermittent flow regimes.

Rainfall in the study area is best represented by Bureau of Meteorology (BOM) stations Bannister (009507; approx. 9.5km from centre of study area) and Boddington North (109516; approx. 16.5km from study area). As is typical for the region, rainfall predominantly occurs between June and September, and annual totals for the catchment vary between 560 – 635 mm (Figure 2). A trend of declining winter rainfall has been observed across the south-west since approximately 1970, and is expected to continue as climate change intensifies (Andrys et al., 2017; McFarlane et al., 2020; CSIRO & BOM 2022). This is expected to increase intermittency of flow regimes, including prolonged summer low/no flow periods.

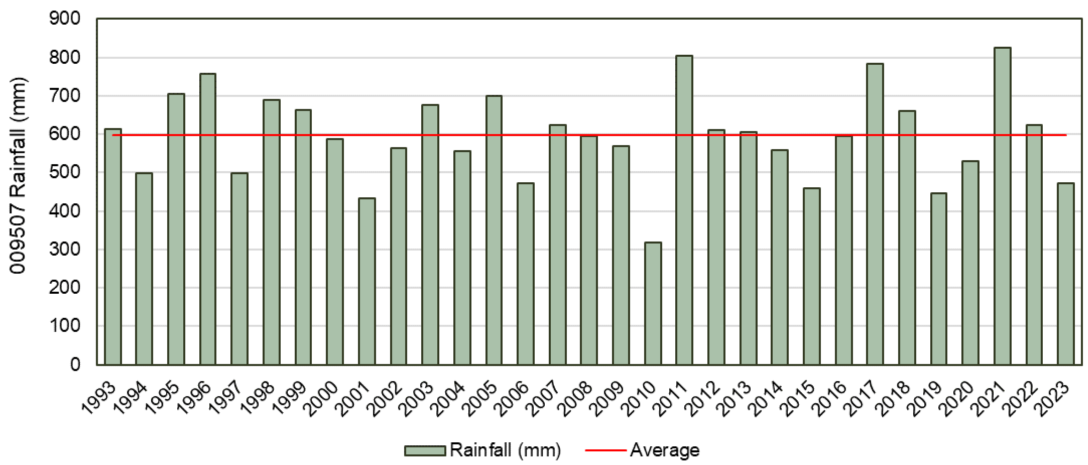

Figure 2. Annual rainfall recorded from Bannister between 1993 – 2023 (BOM 009507). Average (1993 – 2023) total is also given.

4.2 Flow regime

The Hotham River is a major sub-catchment of the Murray River. The sub-catchment is predominantly cleared, being part of the Wheatbelt Region, and is affected by secondary salinisation. The flow regime is strongly seasonal, with an increase from baseflows usually commencing in May/early June, reflecting the onset of winter rainfall (Figure 3 and Figure 4). The large soil-water storage capacity of the region continues to generate flows well after winter and spring rainfall ceases, which usually continues into November followed by return to baseflow. Summer and autumn are typified by a prolonged period of low or no flows, and disconnection of summer refuge pools. The stretch of river in the vicinity of the Hotham Weir (HRB1; Figure 1) appears to be a losing[2] reach, and often experiences periods of zero flow between November and May. The reach from Hotham Weir downstream to Marradong Bridge is regarded as a gaining reach, and experiences low flows throughout the dry season (minimums *circa.* 10 L/sec or 36 kL/hr). The drying climate is expected to prolong the dry season, and may result in reduced baseflows as groundwater levels decline (Petrone et al., 2010; McFarlane et al., 2020). Prolonged dry periods would intensify physical-chemical changes that accompany seasonal drying (e.g. evapo-concentration of solutes, oxygen stress), and increase the critical importance of permanently inundated summer refuge pools (Boulton 2003; Gómez et al., 2017). These changes will increase pressure on resident aquatic fauna.

For the purposes of this hazard assessment and development of SSGVs, the wet and dry seasons were defined using the median flow rate per *N*th calendar day 2015 to 2023, using the median flow recorded at the Hotham Weir (Figure 5). Because the Marradong Weir is situated on a gaining section of the river, and downstream of likely proposed discharge locations, flow data from this gauge would not be representative of the no flow conditions further upstream, and temporal comparisons would be confounded by the addition of discharge. Therefore data from the Hotham Weir (at HRPB1) was used in all following calculations and assumptions.

Wet and dry seasons were defined as follows:

- The overall median flow rate per calendar day is 139 L/sec or 500 kL/hr, measured at the Hotham Weir.
- The commencement of the wet season was defined as the *N*th day where the median flow rate had exceeded the overall median for the prior consecutive 10 *N*th days (day 151, 31st of May). At least 10 days of consecutive flow increases above were chosen to differentiate the rising limb of the annual flow regime, from fluctuating flows in autumn.
- The commencement of the dry season was defined during the falling limb of the flow regime, as the *N*th day occurring 10 days prior to the first day with flows below the median (day 313, 9th of November).
- Importantly, the dry season has commenced during October in some years, for example in 2023 the first day below median flow was the 18/10/2023, and zero flow commenced on the 22/11/2023.
- The wet season is defined as the 1st of June to the 31st of October, and the dry season the 1st of November to the 31st of May.

[2] "Gaining" and "losing" reaches refer to the vertical linkage of groundwater and the stream channel. In gaining (effluent) reaches, the water table slopes downward to the channel and generates baseflows via groundwater contribution. Losing (influent) reaches lie above the water table, and depending on streambed permeability lose surface water vertically to the water table (Boulton et al., 2014).

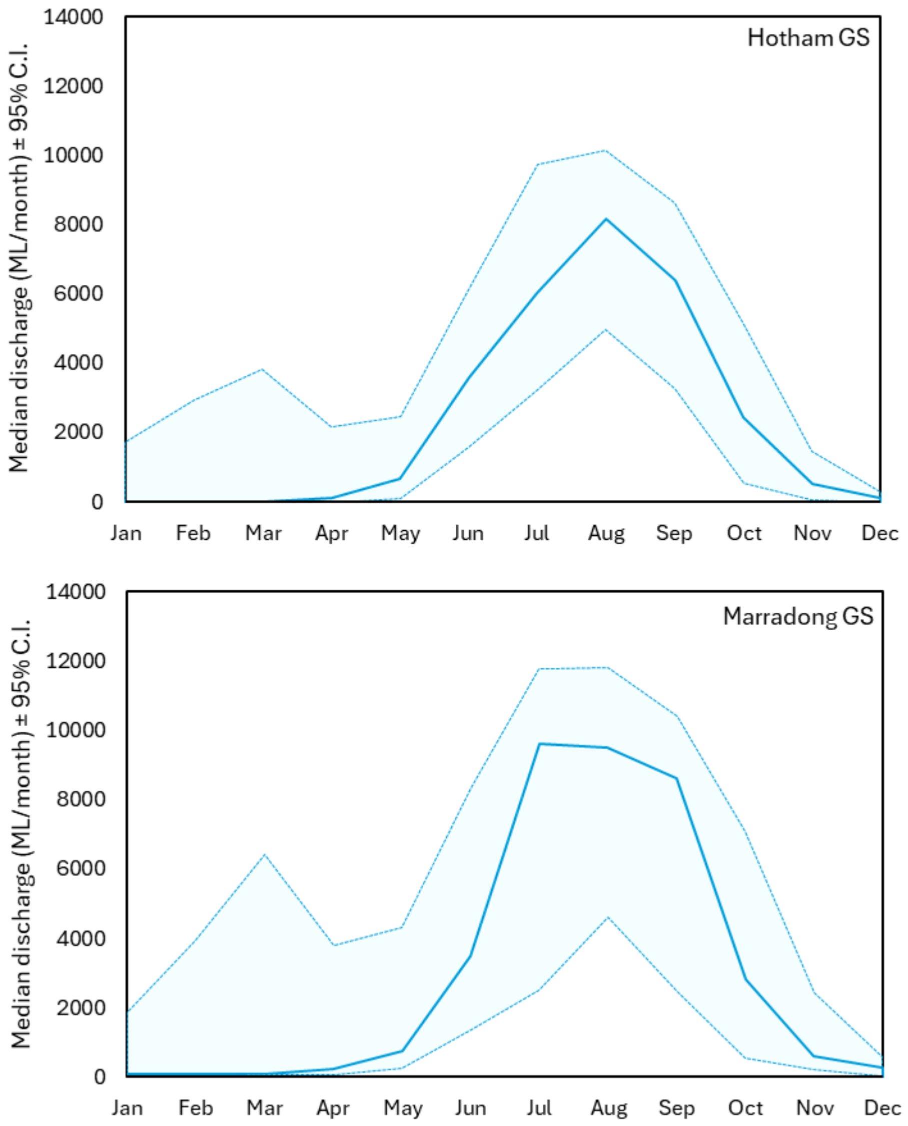

Figure 3. Flow volumes for the Hotham River (median ± 95% C.I.) (ML/month) recorded at Hotham Weir and Marradong Bridge (GS614224) between 2015 and 2023.

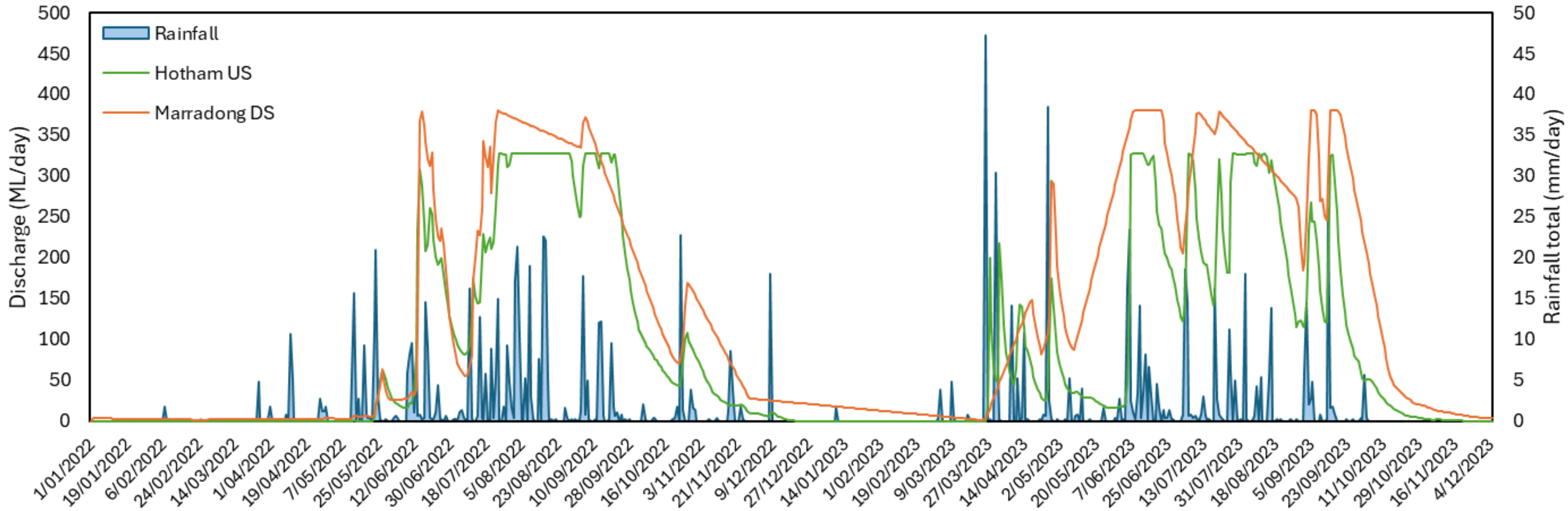

Figure 4. Daily flow hydrographs (ML/day) for the Hotham River at Hotham Weir (upstream) and Marradong (DS) 01/01/2022 to 05/12/2023 inclusive, overlain with daily rainfall observations (mm) for Boddington (BOM station 109516). NB. The 'flat line' of Hotham Weir data reflects the upper limit of the depth gauge/rating curve for this site; actual upper flows would be higher than here reported.

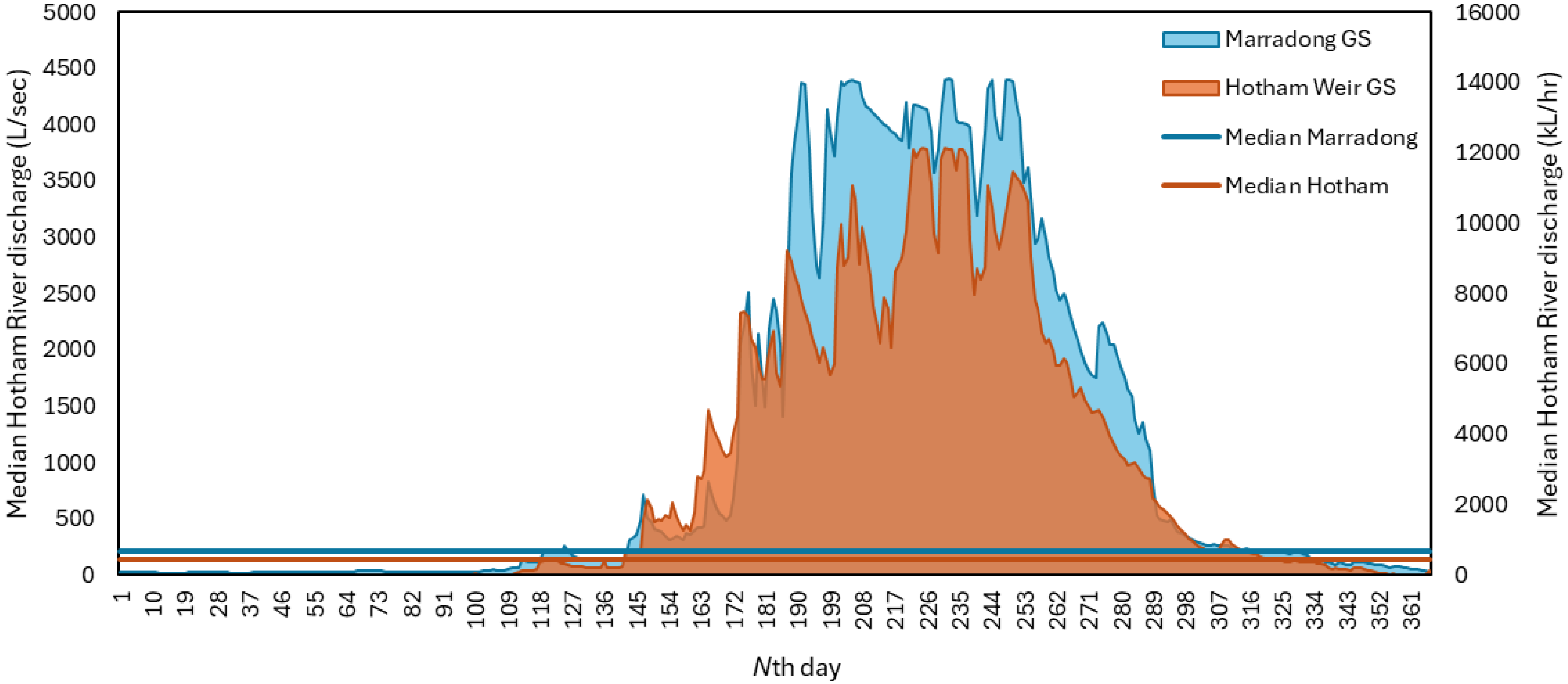

Figure 5. Median flow rate per *N*th calendar day (01/01/2015 to 05/12/2023 inclusive), for the Hotham River above pumping stations (Hotham Weir GS) and below (Marradong Bridge GS614224). Units are instantaneous flows as L/sec, primary axis; kL/hour secondary axis. The median flow rates for both gauges overlain (Hotham Weir 139 L/sec; Marradong Bridge 210 L/sec).

5.0 Desktop review

5.1 Methods and guidance

A review of aquatic ecosystem values of the Hotham River adjacent to NBG is provided below, including faunal records held in publicly available databases (Table 1) and relevant records from published scientific literature, unpublished reports, and other grey literature (Table 2). The majority of available literature focusses in and around the Hotham River near Boddington and surrounds.

The conservation significance of all aquatic fauna recorded was assessed using established lists and databases, outlined below:

- Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act),
- Western Australian *Biodiversity Conservation Act 2016* (BC Act) as Threatened or Priority species, as listed on the DBCA Threatened and Priority Fauna List (DBCA 2023),
- International Union for Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2023),
- Australian Society for Fish Biology Conservation List (ASFB 2018),
- Potential or known short range endemic (SRE) freshwater invertebrate species, that have naturally small distributions of less than 10,000 km^2 (after Harvey 2002), as described by the EPA (2016c) for the purposes of environmental impact assessment, and/or stygofauna (groundwater) species that are also potential or known short range endemic (SRE) species, as described by the EPA (2016b) for the purposes of environmental impact assessment.

Table 1. Summary of database searches for aquatic fauna in Hotham River and surrounding areas.

Database	Description	Authority	Area of Search/ Species
Dandjoo (prev. NatureMap)	Search conducted by SLR on 30th May 2024	DBCA and WAM	Hotham River in vicinity of study area
Freshwater Fish Distribution in Western Australia	Search conducted by SLR on 30th May 2024	DBIRD	All freshwater fish species
The Australian Faunal Directory (AFD)	Utilised in assessing taxonomic status and distribution of aquatic fauna	Australian Biological Resources Study (ABRS; an initiative of DAWE)	All relevant species
Atlas of Living Australia (ALA)	Search conducted by SLR on 30th May 2024 Utilised in assessing taxonomic status and distribution of aquatic fauna	Collaborative project between academic, private and community groups.	Hotham River in vicinity of study area

Table 2. Summary of available literature assessing aquatic fauna in vicinity of Hotham River catchment.

Author	Date	Title	Systems	Ecosystem attributes
Bunn & Davies	1992	Community structure of macroinvertebrates of a saline river system	Hotham River, 34 Mile Brooks	WQ, macroinvertebrates
Morgan & Beatty	2004	Monitoring the Lion's Weir Fishway	Hotham, Bannister & Williams rivers	Fish, salinity
Sharafi et al., (DoA)	2005	Avon-Hotham catchment appraisal	Avon & Hotham catchments	Hydrology, WQ, biota, catchment-scale condition
SLR	2024a	Hotham River Fish Monitoring 2023	Hotham River	Fish populations, hydroregime, long-term analysis
SLR	2024b	Gringer Creek Aquatic Fauna Survey & Interim Site-Specific Guideline Values	Gringer Creek	WQ, macroinvertebrates, fish & crayfish
SLR	2024c	Boggy Brook Baseline Aquatic Fauna Survey and Site-Specific Guideline Values	Boggy Brook	WQ, macroinvertebrates, fish & crayfish
WRM	2012c	Acquired Lands baseline aquatic fauna survey	Boggy Brook, House Brook, Wattle Hollow Brook	WQ, macroinvertebrates, fish & crayfish
WRM	2012d	Hotham River EWRs	Hotham River	Fish populations, hydroregime
WRM	2016	Hotham River Fish Monitoring 2015	Hotham River	Fish populations, hydroregime
WRM	2017	Hotham River Fish Monitoring 2016	Hotham River	Fish populations, hydroregime
WRM	2018	Hotham River Fish Monitoring 2017	Hotham River	Fish populations, hydroregime
WRM	2019	Hotham River aquatic ecosystem health assessment	Hotham River	WQ, sediment quality, fish & crayfish, mussels, other macrofauna
WRM	2020	Hotham River Fish Monitoring 2019	Hotham River	Fish populations, hydroregime
WRM	2020	Hotham-Williams river health assessment	Hotham-Williams rivers (wider sub-catchments)	WQ, macroinvertebrates, fish & crayfish, fringing zone, physical form (SWIRC)
WRM	2022	Hotham River Fish Monitoring 2021	Hotham River	Fish populations, hydroregime

5.2 Summary of results

5.2.1 Water quality records

Several broader scale regional assessments overlapping the project area have been conducted including an assessment and discussion of water quality, for both the Hotham and Williams rivers (WRM 2019, 2020) and the wider Avon-Hotham catchments (Sharafi et al., 2005). Generally, waterways in the vicinity of Hotham River have been characterised by varying degrees of legacy impacts from clearing of native vegetation for agriculture and forestry. In particular, the effects of secondary salinisation and eutrophication are reportedly widespread in the study area, and the effects of these processes are apparent across the

Hotham-Williams sub-catchments (WRM 2020). Prior studies of water quality attributes in creeklines in closer proximity of the study area included Gringer Creek (WRM 2012a; SLR 2024b), 34 Mile Brook and Jungellan Brook (Bunn & Davies 1992; WRM 2012b; WRM 2013) and other minor creeklines in proximity of the NBG mine (WRM 2012c). The connection between catchment clearing and secondary salinisation is demonstrated by baseline surveys conducted at creeklines in proximity to NBG (e.g. WRM 2012c, 2013). For example, Boggy Brook is a minor creekline draining a forested catchment and is fresh in its upper forested reaches (EC 159 – 262 µS/cm), whereas reaches immediately downstream that traverse cleared farmland were brackish to saline (>2,500 µS/cm), to saline in the lower reaches (>10,000 µS/cm) (WRM 2012c; SLR 2024c).

Salinisation has likely affected all waterways in the area that have connectivity to groundwater (i.e. are not solely rainwater fed, perched systems). For example, an assessment of the Avon and Hotham River Catchment by the Department of Agriculture (Sharafi et al., 2005) estimated that since clearing, the Hotham had increased from an average salinity range of 1000 - 5500 µS/cm (fresh to brackish) to between 4000 µS/cm and 25,000 µS/cm, reclassifying it as a brackish to saline system. A widely observed impact of secondary salinisation is alteration of aquatic and riparian flora and fauna composition, through loss of truly freshwater salinity-sensitive species and an increase in saline tolerant species (e.g. Pinder et al., 2005) which has very likely occurred in the Hotham River catchment (Sharafi et al., 2005). This is in acknowledgement that detailed aquatic flora and fauna records do not pre-date impacts of clearing, and therefore we do not have accurate data for pre-European state. As well as direct impacts, loss of native fringing vegetation due to salinisation also has indirect effects including sediment mobilisation, permanently altering channel physical form (van Looj et al., 2009).

Eutrophication is also a frequent impact on waterways associated with agriculture, whereby nitrogen and phosphorus from fertilisers and animal manures accumulate in natural waterways to a point that tips the system from an oligotrophic (nutrient limited) to a eutrophic (nutrient enriched) state (Boulton et al., 2014; Dodds & Smith 2015). Historically, soils of the south-west region were particularly low in nutrients, owing in part to the great age of the landscape. As observed in many southwest catchments, including the upper Hotham catchment, the removal of native vegetation and introduction of broad scale agriculture have drastically changed nutrient cycles (Sharafi et al., 2005). Again noting paucity of historic environmental data for the Hotham River, it is almost certain that the current levels of nitrogen and phosphorus in the upper catchment are a substantial increase from pre-clearing conditions, causing growth of toxic and nuisance algal blooms and altering ecosystem function (Sharafi et al., 2005). Closer to the project area itself, (i.e. the Hotham in vicinity of 34 Mile Brook), Bunn & Davies (1992) found that phosphorus and nitrogen levels were low and did not indicate eutrophication. However, they suggest there is potential for pulses of nutrients from agricultural runoff under first flow events and/or unseasonal rainfall, which may have deleterious ecological impacts. Fish and crayfish kills have been reported after such an event washed large quantities of nutrients and farm-derived manure into the Hotham River (Bunn and Davies, 1992).

Although secondary salinisation is an understood environmental impact occurring in the study area, other water quality attributes are generally reported as in good condition. Studies along the Hotham River and tributaries indicate circum-neutral to slightly alkaline pH, well oxygenated waters, and good clarity in lower reaches (WRM 2012a,b; 2013). In more recent studies (2019), water quality in Hotham River sites, upstream of Boddington, were characterized as saline nearing hypersaline (10,550 - 54,500 µS/cm EC), neutral to moderately alkaline pH (7.78 – 9.10), generally well oxygenated, though with instances of daytime super-saturation and night time oxygen-stress (daily dissolved oxygen (DO) fluctuation 23.8% – 208.5%). The river had good clarity (including low turbidity 3.39 - 26.51 NTU), though occasionally over the DGV, likely due to cessation of flow and receding pools in summer months, which likely caused evapo-concentration of solutes including nutrients. Sites were mainly clear and oligotrophic when flowing during the wetter months.

5.2.2 Aquatic invertebrates

A total of 51 macroinvertebrate taxa[3] were recorded from Hotham River, upstream of Lions Weir in a 2019-2020 study (Hotham River upstream of Lion's Weir; Morgan & Beatty 2004). Macroinvertebrate assemblages were characterised by salt tolerant species. Diptera (two-winged flies) dominated the fauna, in particular chironomids (non-biting midges), which were common and abundant across all sites. Chironomid species *Cladopelma curtivalva*, *Dicrotendipes* sp., *Paratanytarsus* sp., *Tanytarsus* sp. and *Procladius paludicola and* the soldierfly Stratiomyidae sp. were particularly abundant. Other common taxa included the SWWA endemic amphipod *Austrochiltonia subtenuis* and larvae of the diving beetle *Necterosoma* sp. Bunn & Davies (1992) and WRM (2011a) have previously recorded similar dominance of *Tanytarsus* species in Hotham River and also the salinised Thirty Four Mile Brook. Bunn & Davies (1992) concluded the high densities and relatively low species richness and diversity of the fauna in general were a direct consequence of poor water quality, in particular high EC.

No listed species were recorded in Hotham River during the 2019-2020 study (WRM 2020). Of the total fauna, a number of species were considered south-west endemics, or likely south-west endemics, including dragonfly *Procordulia affinis* (south-west endemic) and *Necterosoma darwini* (Western Australian endemic beetle). Other species recorded near the study area include the ancient south-west endemic damselfly *Archiargiolestes pusillus*, the diving beetle *Megaporus solidus*, the caddis-fly *Oecetis* sp. and the W.A. endemic tanypod *Paramerina levidensis*, and the W.A. endemic, salinity-tolerant caddisfly *Symphitoneuria wheeleri* which is unusual among a group of taxa that are typically sensitive to high salinity. Altogether, whilst no listed species were recorded, the presence of several regionally endemic taxa does afford this macroinvertebrate assemblage a degree of conservation significance. Furthermore, few invertebrate taxa actually appear on formal conservation lists, despite well established threats to south-west freshwater fauna due to environmental alteration and climatic drying (Sutcliffe 2003; Pennifold 2018; Carey et al., 2023). Knowledge gaps remain with regard to habitat preferences, life histories and water quality tolerances of south-west endemic macroinvertebrates (e.g. Kay et al., 2001; Stewart et al., 2013; Pennifold 2018; Greenop et al., 2024) therefore it is difficult to predict changes in fauna due to altered water quality or flow regime.

5.2.3 Fish

Of the 11 native freshwater fish species known to occur within south-west Western Australia, four species have been recorded previously from the Hotham River catchment, including the western minnow (*Galaxias occidentalis*), western pygmy perch (*Nannoperca vittata*), nightfish (*Bostockia porosa*) and freshwater cobbler (*Tandanus bostocki*). In addition, two native estuarine species, the Swan River goby (*Pseudogobius olorum*) and the Southwestern goby (*Afurcagobius suppositus*), and two introduced species, Mosquitofish (*Gambusia holbrooki*) and redfin perch (*Perca fluviatilis*) are also known to occur within the catchment (WRM 2016, 2017, 2020, 2021). Limited studies have been undertaken upstream of Lions Weir (e.g. Morgan & Beatty 2004, and WRM 2020), with the majority of historical records from studies downstream of Lions Weir, but upstream of the Marradong Road bridge (WRM 2016, 2017, 2018, 2021; SLR 2024).

[3] Not all specimens could be identified to species, so taxa refers to the lowest level of identification (in most cases species or genus, but also including family and order).

The western minnow and western pygmy perch are common and widespread in southwest Western Australia (SWWA). Both species occupy a wide variety of habitats, have a relatively high salinity tolerance and are typically abundant where present, particularly true of the western minnow (Morgan et al., 1998). Studies indicate that adult western minnow and western pygmy perch can tolerate salinities up to ~14,600 mg/L (~21,000 µS/cm) (Beatty *et al*. 2008), although larvae/juveniles are likely more sensitive to elevated salinity. Potential threats to local populations include habitat alteration, artificial barriers to migration, changes to flow regime, and introduction of exotic species (Morgan et al., 1998, Morgan et al., 2004; Morgan & Beatty 2004). Both species are known to migrate to headwater tributaries to spawn, with western minnow generally breeding over early- to late winter, and western pygmy perch breeding beginning in late-winter through spring, to avoid interspecific competition (Pen & Potter 1990a; Pen & Potter 1991). Particularly for western minnow, this annual migration is obligatory for breeding to take place (pygmy perch are also known to use channel backwaters). Recent surveys on Gringer Creek found gravid females apparently migrating to the upstream reaches (SLR 2024b), this tributary creekline is likely to be an important breeding habitat for local populations of the Bannister and Hotham rivers.

Nightfish are also widespread in SWA. They are solitary, bottom dwelling fish, and, as the name suggests, are more active during the night than during the day. Pen & Potter (1990b) report that nightfish reach approximately 56 mm total length in their first year and live for at least six years. Thorburn (1999) recorded highest densities of nightfish from finer substrate types, especially mud and fine sand. Similarly to the western minnow and western pygmy perch, nightfish migrate from rivers to small tributaries, where they spawn in the upper reaches (Pen & Potter 1990b), and gravid females were also found in the upper reaches of Gringer Creek in recent baseline surveys (SLR 2024b). Migration occurs over early to late winter, following downstream migration of adults and juveniles prior to commencement of drying in streams that are intermittently flowing. Disruption of annual breeding migrations may also be expected to negatively affect local populations of nightfish.

The freshwater cobbler is the largest native freshwater fish endemic to SWWA, reaching a maximum size of approximately 40cm total length. This iconic species has a scattered distribution from Moore River in the north to Frankland River in the south. Distribution of this species is becoming increasingly restricted due to habitat loss (e.g. vegetation clearing, de-snagging, drain construction; Morgan et al., 1998). They are also the only endemic freshwater fish targeted by recreational anglers. Due to the size of individuals, they also require a greater minimum depth compared to other smaller bodied native species to ensure passage over obstacles during migration (i.e. 0.2 m minimum depth rather than 0.1 m for small bodied species) and are thus particularly vulnerable to flow regime change and water abstraction (WRM 2011). This minimum depth criterion (0.2 m) was confirmed by Beatty et al. (2008) for freshwater cobbler in the Blackwood River system. Findings from this study found freshwater cobbler undertake localised migrations (upstream and downstream) between pools during spring and summer (Beatty et al., 2009, Beesley et al.,2019). These movements were highly localised and suggested a high degree of site fidelity, with potentially some ‘home-ranges’ at the scale of individual riverine pools (Beesley et al*.,* 2019). Similar results have been confirmed for the Hotham River, downstream of Lion’s weir, with localised migrations detected during increased flow events (WRM 2011, 2018).

The Swan River goby (*Pseudogobius olorum*) is a typically estuarine species that can occur long distances inland in secondarily salinised rivers (*e.g.* the Avon River and the Blackwood River), and even occurs in some isolated hypersaline lakes. The species only lives for about a year and is thought to be sexually mature once they have attained ~25 mm total length, usually between five and seven months of age (Gill et al., 1996). These small benthic fish are not particularly strong swimmers, and prefer slower flows, and thus may be advantaged in lower rainfall years where flow velocity is lower, as shown anecdotally for the Hotham River (WRM 2017; SLR 2024)

The Southwestern goby (*Afurcagobius suppositus*) exhibits similar characteristics and distribution to the Swan River Goby, being endemic to the region and ranging across much of the south-west coastal drainage systems and further inland where secondary salinization occurs (Gill, 1993; Gill and Potter 1993). Also not strong swimmers, the Southwestern goby will favour silty or muddy creek bottoms with low-flow (Allen *et al.,* 2002).

Two introduced species have been recorded in the Hotham River catchment. The mosquitofish *Gambusia holbrooki* is a small species native to the rivers draining into the Gulf of Mexico, and was introduced into the SWWA in 1934 in a failed attempt to control mosquito populations (Morgan et al., 1998). Mosquitofish have subsequently proliferated across the entire SWWA region, becoming common and abundant in habitats where they have become established (Morgan et al., 2004). Mosquitofish not only compete with native species for food, but also display antagonistic behaviour towards native fish. It has been well documented they aggressively “fin-nip” native fish causing extensive damage to caudal fins, which is demonstrated to cause death in small native species (Gill et al., 1999).

The redfin perch is a large introduced predatory species, and is associated with severe negative impacts on native fish and crayfish populations in rivers where it has become established, including being implicated in local extinctions (Hutchison 1991; Morgan et al., 1998, 2002, 2004). Redfin are known to be a piscivore (fish-eater), and readily predate on native fish and crayfish. Hutchison (1991) examined the introduction of redfin and subsequent disappearance of native pygmy perch in the Murray River system, noting that rarely did they co-exist and any instance of co-existence the numbers of native fish species were very low. Redfin perch are also thought to be a significant predator of native crayfish species including marron (Morgan et al. 2002).

5.2.4 Crayfish

There are 11 native crayfish species in SWWA, six from the genus *Cherax* and five from the genus *Engaewa*, and all are endemic to the region. All species in the genus *Engaewa* are conservation listed crayfish, but are unlikely to occur within the Hotham River catchment given their highly restricted distribution, known habitat preferences, and association with low salinity. The only listed *Cherax* species is the hairy marron, *Cherax tenuimanus*, which is considered critically endangered but is highly restricted to the upper reaches of Margaret River, and would not occur in Hotham River. A twelfth species of freshwater crayfish is the introduced Yabby *Cherax destructor*, which was originally introduced from eastern Australia to farm dams east of the Albany Highway, but has since colonised many systems to the west of the highway, and onto the Swan Coastal Plain, and poses a serious threat to native species and aquatic ecosystems (Lynas et al. 2004, 2006, Beatty et al. 2005).

Of these known 12 species, four freshwater crayfish representing three endemics; the gilgie (*Cherax quinquecarinatus*), the smooth marron (*C. cainii*) and koonac (*C. preissii*) and one introduced species the yabby (*Cherax destructor*) have been recorded from the Hotham River (WRM 2018, 2017, 2016, Morgan & Beatty 2004). None of the three endemic crayfish species known from the Hotham River catchment are considered rare or restricted in distribution. Koonacs and gilgies are also known from other nearby intermittently flowing creeklines, including Gringer Creek, Boggy Brook, Jungelan Brook and 34 Mile Brook (WRM 2012b,c; SLR 2024b; 2024c).

The gilgie is known to exploit almost the full range of freshwater environments, and can be found in habitats that range from semi-permanent swamps to deep rivers (Austin & Knott 1996). These crayfish have a well-developed burrowing ability, digging short burrows under stones on the stream bed or in the banks along the margins (Shipway 1951). In this way, gilgies are able to withstand periods of low water level by retreating into burrows until flows return. Gilgies would appear to be tolerant of salinities up to at least 18,690 µS/cm as evidenced by their presence in Warrin Creek in the upper Helena River catchment (WRM 2011).

The koonac is found in a wide range of permanent and temporary aquatic systems throughout the southwest of Western Australia. They have a range extending from the Moore River in the north to just east of Albany in the south. They exploit a full range of habitats but are most commonly associated with lentic wetlands (Austin & Knott 1996). Koonacs tend to dominate assemblages where water tables fluctuate markedly (Beatty et al. 2006), and can be found in creeklines with highly ephemeral surface water regimes, such as Boggy Brook (SLR 2024c). They burrow to avoid summer drying, and may remain in their burrows for long periods, only leaving when surface waters return in early winter.

5.2.5 Other Aquatic fauna

Rakali (Native Water Rat)

The rakali, or water rat *Hydromys chrysogaster*, is one of Australia's two truly amphibious mammals (the other being the platypus; Australian Museum 2019). Although nationwide the populations appear stable, there are individual populations facing a significant threat. It is listed as a Priority 4 (P4) species on the DBCA threatened and priority fauna list. Despite being a relatively rare and cryptic species, the rakali is found in all Australian states and territories. The water rat was recorded during the Hotham River fish monitoring surveys in 2017, and again in 2023 (WRM 2017; SLR 2024). Records of water rats also occur at 34 Mile Brook, which is 800m upstream of Tullis Bridge and 7kms downstream of the Boddington Town Pool. A study by Williams et. al (2014) found an individual water rat may have a home range of up to 4km. Anecdotal evidence and local sightings suggests there may be rakali present at, or nearby, the Lion's Weir.

Water rats occupy a wide variety of freshwater habitats, from inland waterways to lakes, swamps, and farm dams. Water rats require access to permanent water for feeding and to keep cool over the summer months; they suffer from heat stress if access to permanent pools is lost (Watts & Aslin 1981). Other threats are loss of habitat and loss of aquatic food sources due to altered flow regimes and secondary salinisation (Lee 1995). Water rats are omnivorous, feeding on crayfish, mussels, fish, plants, water beetles, dragonflies and smaller mammals and birds. Water supplementation to maintain these prey items will also provide for their diet. Breeding can occur throughout the year, but more typically in spring. They build nests at the ends of tunnels dug into banks near tree roots or in hollow logs. Therefore, there is a requirement for stable stage heights that inundate banks, tree roots and large woody debris, without erosive flows (WRM 2018). Reduced baseflows or groundwater drawdown leading to drying of pools over summer may result in the loss of local populations.

South-western snake-necked turtle

One species of freshwater turtle is known from the study area; the southwestern snake-necked turtle, or long-necked turtle, *Chelodina oblonga* (Shea et al. 2020). This species is endemic to southwest Western Australia. It is not currently listed at Federal (under the EPBC Act) or State levels (under the BC Act as conservation significant), but is listed under the IUCN Redlist of Threatened Species as Near Threatened (IUCN 2023). It has a relatively widespread distribution throughout the south-west, from Hill River in the north and east to the Sussetta River (Cann 1998), and south to Esperance on the southern coast. Throughout this range, snake-necked turtles are known to occur in both permanent and seasonal habitats, including rivers, lakes, farm dams, swamps, damplands and natural and constructed wetlands (Balla 1994, Guyot & Kuchling 1998). They can migrate relatively long distances overland if local conditions deteriorate (Dr Gerald Kuchling, UWA, pers. comm.) and can aestivate for up to six months to avoid drought in seasonal waterbodies (Kuchling 1988, 1989). Since their diet includes tadpoles, fish, and aquatic invertebrates, south-western snake-necked turtles only eat when open water is present. In permanent waters, this species has two nesting periods (September-October and December-January), but in seasonal systems, nesting will only occur in spring. Females can travel inland as far as 1 km to find appropriate nesting sites at this time (Clay 1981, Kuchling 1998). They generally nest in sandy soils, and eggs take up to two hundred days to hatch. The main threats to these turtles are road deaths during movement in the nesting season and predation by feral animals (Bencini & Turnbull 2012). There are few available data on the spatial distribution of snake-necked turtles in the study area, Peel Harvey Catchment Council (PHCC) report (Pumphreys Bridge, Ranford Pool) Snake-necked turtles inhabit both permanent and seasonal waterbodies throughout their range.

Mussels

The common estuarine mussel *Fluviolanatus subtorta* has been recorded in the Hotham River in vicinity of Boddington, at Town Pool (WRM 2019) and further upstream at Randford Pool (WRM 2020b; Figure 6). This species is not currently listed as conservation significant. It has a relatively widespread distribution in estuarine coastal environments and readily migrates upstream on rivers affected by secondary salinisation (Klunzinger et al., 2011). The range expansion inland to the Hotham River is understood to be a migration upstream from known populations near Mandurah (WRM 2019; ALA 2024).

The freshwater mussel *Westralunio carteri* is endemic to the SWWA region and listed as Vulnerable under the EPBC Act (1999), BC Act (2016) and IUCN Redlist of Threatened Species (2020). Historic investigations into bioaccumulation of metals in mussels (*W. carteri*) were conducted on 34 Mile Brook and its confluence with the Hotham River, using translocated mussels exposed in enclosures (Streamtec 1995). The location of source populations of mussels used in the exposure trials is unknown, however, and not detailed by Streamtec (1995). There have been no records of *W. carteri* in the Hotham River within the vicinity of NBG operations, and Carter's mussel are unlikely to occur due to high salinity concentrations. Carter's mussel is almost never found where salinity is greater than 1.6 g/L^{-1} (EC 2,900 µS/cm), with an acute salinity tolerance LC_{50} of 3.0 g/L^{-1} (EC 4,600 µS/cm), levels below that occurring in the Hotham most of the time. Other bioaccumulation studies at that time (e.g. Storey & Edward, 1989) used mussels from source populations on freshwater creeks in Perth metropolitan area, often the Canning River, and it is suspected those used for the 34 Mile Brook investigation were also from that locality (Andrew Storey, pers. comm.). *W. carteri* has had a range decline of 49% in the last 50 years (Klunzinger *et al*., 2015), while the estuarine species *F. subtorta* is known to colonise waterways that have become too saline for *W. carteri* (Kendrick 1976; Pen 1999), and has even been observed using long-dead *W. carteri* shells as an attachment substrate (Klunzinger et al., 2011). The colonisation of *F. subtorta* in the Hotham lends further evidence that *W. carteri* is likely not present/ no longer present in the Hotham River as a result of salinisation.

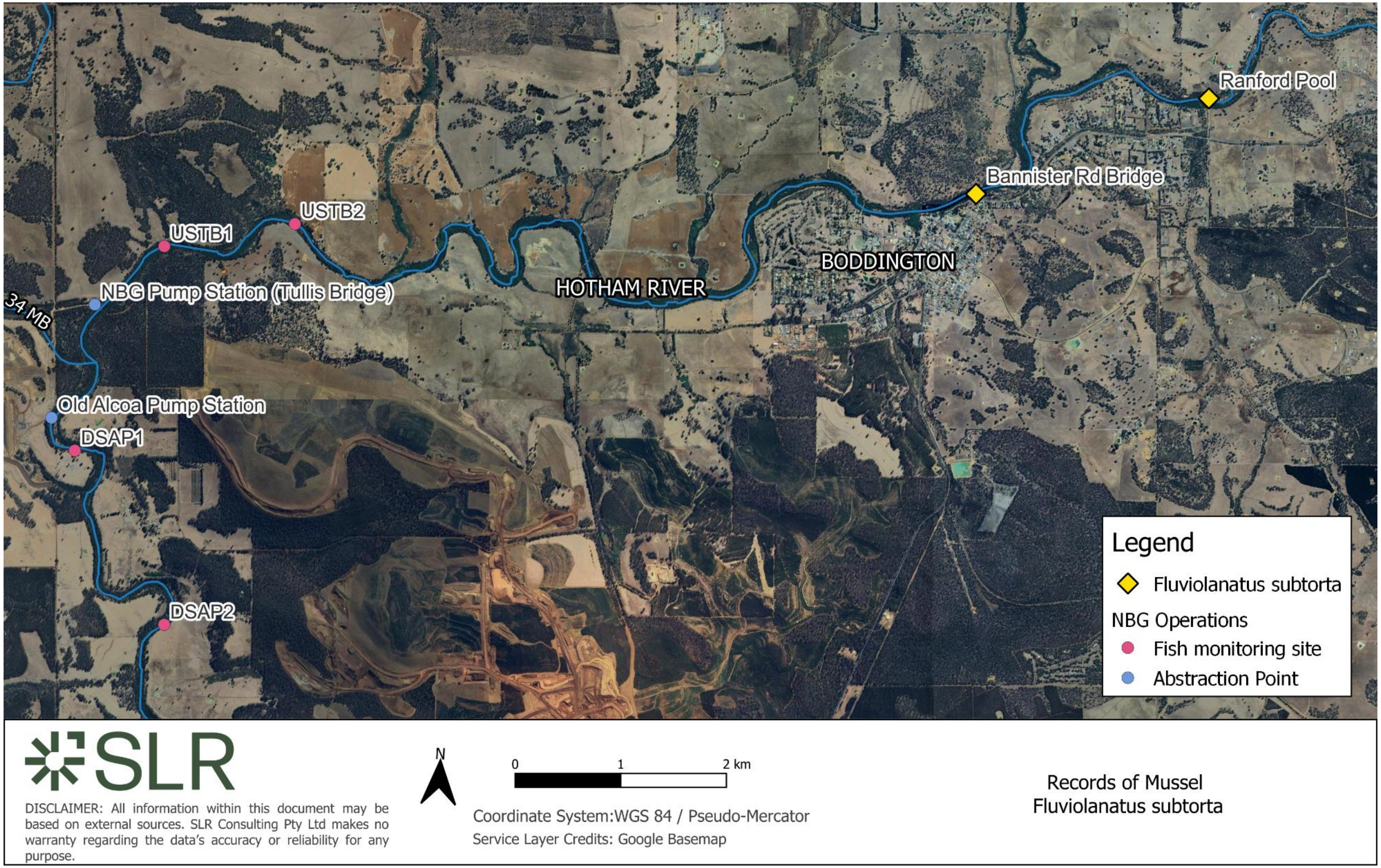

Figure 6. Locations of records of estuarine mussel *Fluviolanatus subtorta* in relation to the NBG abstraction points.

Table 3. Summary of known conservation significant aquatic fauna recorded from the Hotham River upstream and downstream of the NBG abstraction point, including status on state, national and international lists. Macroinvertebrates not listed, however endemism is likely to be high.

Scientific Name	Common Name	Endemic to SWWA region	EPBC Act 1999	BC Act 2016	IUCN 2023
Fish					
Galaxias occidentalis	Western minnow	X			
Nannoperca vittata	Western pygmy perch	X			
Bostockia porosa	Nightfish	X			NT
Tandanus bostocki	Freshwater cobbler	X			NT
Pseudogobius olorum	Swan River goby	X			
Afurcagobius suppositus	Southwestern Goby	X			
Crayfish					
Cherax quinquecarinatus	Gilgie	X			
Cherax cainii	Smooth marron	X			
Cherax preissii	Koonac	X			
Other					
Hydromys chrysogaster	rakali			P4	
Chelodina oblonga	Southwestern snake-necked turtle	X			NT

6.0 Hotham River water quality and interim SSGVs

6.1 Data compilation and methods

Long term water quality datasets from the Hotham River supplied by NBG were compared against ANZG (2018) guidelines, and where appropriate used for derivation of SSGVs as per the NWQMS recommended approach (ANZG 2018; Warne et al., 2018), following data quality screening. Specifics for screening and the removal of outliers is detailed for each analyte in proceeding sections.

The NWQMS recommend the use of a weight of evidence approach when assessing the potential impacts to aquatic ecosystems from analytes of concern, involving multiple lines of evidence across the pressure-stressor-ecosystem receptor pathway (ANZG 2018). This may include assessment using SSGVs/ DGVs against distributions (or changes in distribution) of known sensitive receptors (e.g. macroinvertebrates, fish, zooplankton) as evidence of potential impacts and causal factors. However, in absence of direct toxicity analysis on local fauna, specific tolerances of fauna to levels of concentrations of analytes remain unknown and likelihood of impact can only be inferred. In circumstances where water quality attributes of a system persistently exceed established DGVs for 95% species protection prior to an impact, then it is usually more informative to compare changes over time to site-specific values derived from local data (ANZG 2018). SSGVs provide localised indication of changes in analyte concentrations from background condition. This is usually sufficient to infer likelihood of adverse impact occurring in receiving environments, in absence of direct toxicological assessment on local biota.

In most circumstances, it is recommended that data for toxicants and stressors are compared against DGVs for 99% species protection for minimally disturbed ecosystems, and 95% DGVs for slightly to moderately disturbed ecosystems (ANZG 2018). For the Hotham River, toxicant and stressors with sufficient data available were assessed based on rate of exceedance of the 95% DGV occurring as ‘background’ levels, given the history of clearing and other land uses in the catchment, and associated legacy impacts including secondary salinisation. Lower DGV thresholds are reserved for use in highly disturbed ecosystems, dependent on management goals, and are intended as intermediate targets for water quality improvement (ANZG 2018). Therefore, lower DGV values are not considered appropriate for use as guideline values for the Hotham River.

The widely accepted method for deriving SSGVs recommended by Water Quality Australia for moderately disturbed systems is calculation of 80th percentile values (or 20th percentiles, for analytes for which low values are problematic, e.g. low pH or low oxygen) from a background or ‘pre-impact’ state, and suitable reference sites if available. For this purpose, monitoring data from the Hotham River for physico-chemical stressors and toxicants supplied by NBG were used to calculate summary statistics including:

- Median, 20th and 80th percentiles,
- Minimum and maximum analyte concentrations,
- Proportion of data that exceeds 95% DGVs (as a percentage).

Using these lines of evidence, background concentrations of each analyte were then compared to 95% DGVs to determine whether the default guideline remains appropriate, or if an interim SSGV is justified. Because of strong seasonality in flow regime of the Hotham River, with the majority of flows occurring in June through to October, and low or no flows occurring through the months November to May (see section 4.2; Figure 5, Table 1), with likelihood of winter dilution, and summer evapoconcentration, but also early winter flushing of inland stored salts, water quality data were analysed separately between these “wet” and “dry” seasons. Where necessary, separate SSGVs were derived for wet (June to October) and dry (November to May) seasons.

Prior to analysis, all data were screened for appropriateness for inclusion in interim SSGV calculations, and outliers removed. Data were either gathered in field (e.g. physical attributes DO, pH), supplied from one of two NATA accredited laboratories (ALS or MPL), or were historic archived data prior to 1990. This introduced error in most datasets, most notably due to variation and/or ambiguity in laboratory limits of reports (LORs), and incomplete ‘data transfers’ with erroneous units or orders of magnitude (e.g., potential mis-recordings of data between mg/L and µg/L, giving improbably high or low values). Therefore in some cases, decisions on data exclusion were made on a discretionary basis. Criteria for exclusion included: laboratory limits of detection/ limits of reporting (LORs) too high to be comparable to ANZG DGVs, erroneous values or units, outliers deemed improbable, malfunctioning equipment, or data derived from ‘reliable estimates’, and removal of duplicates. A summary of retained data is provided in Table 4.

Table 4. Summary of water quality data from the Hotham River, suitable for use in calculation of SSGVs. All data are supplied by NBG. Highlighted cells indicate analytes with few data points that are recommended for addition to the regular (monthly) monitoring suite.

Analyte	Date range		*N* samples retained
	Start	End	
Electrical Conductivity	04-01-2012	05-05-2024	405
Dissolved Organic Carbon	08-06-2012	03-12-2019	**101**
pH (H+)	10-05-1988	05-05-2024	711
Temperature (°C)	20-09-1995	05-05-2024	353
Alkalinity	02-09-2015	05-05-2024	258
Hardness	06-04-1989	06-04-2005	**49**
Aluminium	16-12-1998	05-01-2024	375
Ammonia	25-10-1984	05-01-2024	144
Arsenic	23-01-1996	05-01-2024	471
Antimony	19-02-2018	05-05-2024	167
Boron	19-02-2018	08-04-2024	110
Barium	19-02-2018	08-04-2024	103
Bicarbonate (HCO_3)	16-12-1988	05-05-2024	615
Cadmium	02-09-2015	05-01-2024	260
Calcium	25-10-1984	05-05-2024	597
Carbonate (CO_3)	16-12-1988	05-05-2024	345
Cobalt	16-12-1998	05-01-2024	264
Chromium	23-10-2017	17-04-2024	206
Chloride	25-10-1984	05-05-2024	709
Copper	08-06-2012	03-12-2019	275
Cyanide - free	01-04-2015	08-04-2019	**11**
Cyanide - WAD	07-07-2014	08-04-2024	184
Cyanide - total	09-12-1997	08-04-2019	**34**
Iron - dissolved	06-07-2012	08-04-2024	201
Iron - total	30-11-1993	17-04-2024	119
Lead	23-01-1996	17-04-2024	509
Magnesium	30-11-1993	17-04-2024	579
Manganese	23-01-1996	07-04-2024	593
Mercury	07-07-2014	17-04-2024	290
Molybdenum	23-01-1996	02-12-2023	468
Nickel	02-09-2015	17-04-2024	266
Nitrate (N_NO_3)	25-10-1984	05-01-2024	120
Nitrite (N_NO_2)	29-08-1995	05-01-2024	89
Nitrite/nitrate N (N_NOx)	01-06-2017	08-04-2024	**92**
Nitrogen - total	04-09-1989	08-04-2024	128
Phosphorus - total	19-11-2018	08-04-2024	73
Potassium	25-10-1984	05-05-2024	602
Selenium	09-01-2008	06-09-2017	59
Silica (SiO2)	09-12-1997	03-12-2019	162
Sodium	25-10-1984	05-05-2024	630
Strontium	19-02-2018	08-04-2024	103
Sulfate (S_SO4)	23-01-1996	17-04-2024	590
Total dissolved solids (TDS)	06-07-2010	05-05-2024	433
Turbidity	05-10-2010	08-04-2024	96
Total suspended solids (TSS)	30-11-1993	08-04-2024	504
Uranium	19-02-2018	08-04-2024	99
Vanadium	LOR too high		**0**
Zinc	02-02-2015	17-04-2024	266

6.2 Updated ANZG guidance and limitations of the current dataset

Since the ANZG (2018) guidelines were published, there has been further updated guidance regarding the ameliorating effects of some water quality attributes on the bioavailability of some toxicants. Recent draft guidance for toxicants including Cu, ammonia, nitrate and zinc is based on improved understanding of interactions between these toxicants and hardness, pH, temperature, and/or dissolved organic carbon, in determining actual bioavailability to aquatic organisms (ANZG 2023a; ANZG 2023b; ANZG 2024a; ANZG 2024b). These updated guidance documents are still in draft form, but are expected to be published in near future. Update of interim SSGVs to full operational guidelines would require formal publication of these guidance by Water Quality Australia, and any changes in the published versions should be adopted if appropriate.

Ammonia:

Updated guidance is available for ammonia based on new ecotoxicity data for total ammonia-N in freshwaters (ANZG 2023a). The guidelines take into account the effects of both pH and temperature on the relative proportions of un-ionised NH_3 and ionised NH_4^+ in total ammonia-N. Un-ionised NH_3 readily diffuses across cell membranes of aquatic animals and is thus more toxic, and occurs in higher proportions as pH and temperature increase, thereby increasing toxicant uptake by biota. ANZG (2023a) provide DGVs[4] for ammonia at pH 7.0/20ºC and pH 8.0/20ºC, however adjusted DGVs are recommended where local pH and temperature data are available. Adjusted DGVs for total ammonia-N (mg/L) are provided for a range of pH and temperatures (ANZG 2023a). Generally, the laboratory derived total ammonia (expressed as NH_3-N) is directly comparable to the DGVs for total ammonia[5].

The interim SSGV for the Hotham River was derived for the wet and dry season separately, due to seasonal differences in pH and temperature. The 80th percentile values for each season were then applied to the tables provided in ANZG (2023a). Should the distribution of pH and or temperature change substantively following commencement of discharge to the Hotham River, then the toxicity of ammonia present would also change, and an adjusted SSGV would need to be calculated and applied.

Nitrate:

The original ANZECC/ARMCANZ (2000) guidelines for nitrates were erroneous, and as of May 2024 new draft guidance has become available, deriving three sets of N_NO_3 default guidelines for soft, moderately hard, and hard waters (ANZG 2024). Following withdrawal of the ANZECC/ARMCANZ (2000) DGVs, the New Zealand DGVs based on Hickey *et al*., (2013) were recommended for use *in lieu* of specific guidance for Australia, which defined a 95% DGV of 2.4 mg/L as N_NO_3 (ANZG 2018). The new draft guidance is intended to supersede Hickey *et al*., (2013), however is still in the public comment phase. The proposed draft 95% DGV derived for hard waters (defined as >150 mg/L as $CaCO_3$; applicable to the Hotham River) of 29 mg/L, an order of magnitude higher than previously advised by Hickey et al., (2013). However, it is imperative to note that these guidelines refer only to direct toxicity, and do not protect against eutrophication[6] (ANZG 2024), which can have severe negative impacts to both aquatic fauna and human amenity. It is likely that N_NO_3 at *circa*. 29 mg/L would cause displacement of aquatic species and degradation of the environment through eutrophication (i.e. algal blooms etc), well before impacts related to direct toxicity eventuate.

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[4] Bivalves are the most sensitive group to ammonia toxicity, therefore where bivalves are present 99% DGVs should be used.

[5] Confirmed by liason with ChemCentre (a NATA accredited laboratory).

[6] The eutrophication DGV recommended for southwest rivers is much lower at 0.2 mg/L N_NO_x.

Copper:

Water hardness is understood to have an ameliorating effect on toxicity of Cu (particularly hardness >200 mg/L $CaCO_3$; ANZG 2023b). Despite this, hardness modification (as in Warne et al., 2018) is no longer recommended for Cu, as DOC and pH are now understood to have a greater effect on the bioavailability of Cu. The draft guidance recommends that in absence of local DOC data, adoption of the 95% DGV at the standardised DOC of =<0.5 mg/L is the recommended conservative approach, with 0.5 mg/L considered a very low level of DOC, and therefore a very conservative/protective DGV. These values are applicable at pH 6.5 – 8.0 and hardness of 2 – 200 mg/L. Ideally, DOC and Cu data would be collected concurrently and the DGV threshold calculated on a per-sample basis. There is DOC data available for the Hotham River (site HRPB1 08-06-2012 to 03-12-2019; *N* = 101 samples), however there were limited paired datasets for DOC and Cu. However, data were sufficient to calculate an approximation of DOC for the river to produce an interim SSGV for Cu[7]. DOC has been reinstated as part of the regular water quality monitoring suite for the Hotham River, which will allow comparison with Cu data (and other analytes for which DOC is a toxicity modifying factor) on a real-time basis.

Cyanide:

Cyanide (measured as free/uncomplexed HCN + CN^-) has an ANZG (2018) 95% DGV for freshwater ecosystems of 0.007 mg/L. Limited data for free cyanide were provided (*N* = 11). Data were also provided for weak acid dissociable (WAD) cyanide, and total cyanide, which reflect complexed cyanide that is not necessarily directly toxic to aquatic fauna. Although these measures are not directly comparable to ANZG (2018) DGVs, as the data are all under LOR this indicates free and complexed cyanide concentrations are below detection for the Hotham, therefore default guidelines remain appropriate. It is highly recommended that further background data on free cyanide be gathered, at an appropriate LOR, prior to commencement of mining discharge so that a more comprehensive baseline dataset is developed to allow direct comparisons to the DGV, and any potential change over time measured.

Hardness modified trigger values (HMTVs):

Default guideline values for Cd, Cr, Ni and Pb are standardised at hardness of 30 mg/L $CaCO_3$, and again corrections are applied to these guidelines to account for hardness (Warne et al., 2018). To derive an interim SSGV for these analytes, the lowest hardness value recorded (775 mg/L for 49 samples) was used. This was considered a conservative approach aimed at providing accuracy as well as being sufficiently conservative to protect aquatic ecosystem values. However, the data are limited to 49 measurements spanning 12-06-1989 to 06-04-2005 with no recent records available. Given the importance of hardness in ameliorating several toxicants, it is strongly advised that hardness be included in the regular monitoring suite as soon as possible. This would allow comparison of toxicant concentrations against HMTVs on a per sample basis where required. At least monthly data over 24 months would be needed to update interim HMTVs, with replicates advised.

Zinc is also corrected using hardness modification (Warne et al. 2018) however recent draft guidelines also take into account the toxicity modifying factors of pH and DOC, which influence speciation and complexation of Zn, and the rate at which free Zn can cross membranes (ANZG 2024b). The draft guidance provides conservative DGVs at index water quality conditions, where zinc would be highly bioavailable and thus toxic (pH 7.5, hardness 30 mg/L and DOC 0.5 mg/L[8]) for use where sufficient background data are not available. Similarly to analytes

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[7] Equation for the derivation of DOC adjusted copper DGVs (reproduced from ANZG 2023b):

$$\text{Adjusted DGV} = \text{DGV}_{0.5} \times \left(\frac{\text{DOC}}{0.5}\right)^{0.977}$$

[8] Proposed ANZG (2024b) ‘index’ water quality DGV 95% = 0.0041 mg/L

above, to apply the new guidance to the Hotham River the 80th percentile pH, minimum hardness, and 20th percentile DOC were used to derive an appropriate interim SSGV for Zn.

6.3 Summary statistics and selected comparisons against ANZG DGVs

Generally, most analytes were recorded at low concentrations (usually below 95% DGVs), or below laboratory LORs that were sufficiently low as to provide a valid comparison with DGVs. For the vast majority of analytes, the 95% DGV for slightly to moderately disturbed ecosystems remain appropriate for use for the Hotham River, and are included as the interim SSGV for potential future mining discharge (i.e. where analysis of local data showed the derived SSGV to be lower than the 95% species protection DGV, the DGV was adopted). Where required, separate SSGVs are derived to account for differences in analyte concentrations between wet and dry seasons. Year round SSGVs take into consideration differential background concentrations for some analytes in each season, and apply the more conservative value. A summary of statistical analysis for each analyte is presented in Table 5.

Three sets of interim SSGVs are presented in section 6.3.7, including wet and dry seasons, and year-round interim SSGVs. Year-round SSGVs take into account differences in sensitivity between seasons for some analytes, rather than recalculation of percentiles in most cases.

Table 5. Summary statistics of water quality analytes recorded from the Hotham River, analysed between wet and dry seasons (except hardness as $CaCO_3$). Summary statistics are compared to the ANZG (2018) 95% DGVs (highlighted values > DGV), and indication of whether site specific guideline value (SSGV) is recommended. All units are mg/L unless specified. See footnotes.

Analyte	ANZG 95% DGV	WET SEASON (JUNE-OCT)					DRY SEASON (NOV-MAY)					SSGV Recommended	Comment
		No.	Median	80th/ 20th %ile	95th %ile	n exc (%exc)	*No.*	Median	80th/ 20th %ile	95th %ile	n exc (%exc)		
EC (uS/cm)Δ	250	*169*	10,860	13,524 / 7,847	16,510	100%	*236*	10,910	11,678 / 9,130	15,812	100%	YES	Instantaneous (reference condition) Δ
DOC	n.p.	*49*	14	16.0 / 12.0	21	N/A	*52*	14	16.0 / 12.0	24	N/A	NO	Include in monthly monitoring
pH (H+)	6.5 - 7.5	*309*	7.4	7.6 / 6.9	7.9 / 6.4	124 (40%)	*402*	7.6	7.8 / 7.2	8.1 / 6.7	231 (57%)	YES	Year-round SSGV 6.5 to 7.8
Temperature (°C)	n.p.	*151*	13.5	16.3	19.5	N/A	*203*	22.6	25.1 / 19.3	27.1	N/A	YES	Different 80th %ile between seasons
Alkalinity	n.p.	*103*	111	135.6	153.8	N/A	*155*	184	214.2	239.4	N/A	NO	No SSGV recommended
Hardness	n.p.	*49*	1,700	2,100 / 1,360	2,555	N/A	-	-	-	-	-	NO	Min value used for HMTV
Aluminium	0.055	*157*	<LOR	0.03	0.13	24 (15%)	*218*	<LOR	0.02	0.1	16 (7.3%)	NO	Retain 95% DGV
Ammonia #adjusted	0.67 / 0.26	*68*	0.03	0.06	0.11	0%	*76*	0.04	0.07	0.1	2 (2.7%)	YES	SSGV using 80th%ile pH & temp.
Arsenic	0.024	*202*	<LOR	0.005	0.005	4 (2%)	*269*	<LOR	0.005	0.005	2 (0.8%)	NO	Retain 95% DGV
Antimony	0.009	*72*	<LOR	<LOR	<LOR	0%	*95*	<LOR	<LOR	<LOR	0%	NO	Retain 95% DGV
Boron	0.94	*47*	0.08	0.11	0.14	0%	*63*	0.12	0.14	0.16	0%	NO	Retain 95% DGV
Barium	n.p.	*48*	0.011	0.14	0.17	N/A	*55*	0.14	0.17	0.18	N/A	YES	Different 80th %ile between seasons
Bicarbonate (HCO3)	n.p.	*259*	116	143	180	N/A	*356*	192	155	300	N/A	NO	No SSGV recommended
Cadmium	0.002	*151*	<LOR	<LOR	<LOR	0%	*109*	<LOR	<LOR	<LOR	1 (<1%)	NO	Interim HMTV
Calcium	n.p.	*254*	120	158	200	N/A	*343*	141	170	192	N/A	YES	Different 80th %ile between seasons
Carbonate (CO_3)	n.p.	*130*	<LOR	<LOR	<LOR	N/A	*215*	<LOR	<LOR	<LOR	N/A	NO	No SSGV recommended
Cobalt	0.0014	*107*	<LOR	<LOR	0.03	17(16%)	*157*	<LOR	<LOR	0.007	15 (9.5%)	NO	Retain 95% DGV
Chromium	0.001	*84*	<LOR	<LOR	<LOR	0%	*122*	<LOR	<LOR	<LOR	0%	NO	Interim HMTV
Chloride	n.p.	*283*	2600	2,988/2,044	3,100	N/A	*426*	4,100	4,970/3,500	5,770	N/A	NO	No SSGV recommended
Copper *	0.01	*116*	0.002	0.005	0.07	16 (14%)	*159*	0.002	0.005	0.03	18 (11%)	NO	DOC adjusted interim SSGV
Cyanide - free	0.007	*11*	<LOR	<LOR	<LOR	N/A	-	-	-	-	-	NO	Retain 95% DGV
Iron - dissolved	n.p.	*103*	0.1	0.17	0.31	N/A	*98*	0.07	0.11	0.27	N/A	YES	Different 80th %ile between seasons
Iron - total	n.p.	*60*	0.26	0.4	0.8	4 (7%)	*59*	0.17	0.3	0.4	1 (<2%)	NO	DGV likely to be revised
Lead	0.0034	-	-	-	-	-	-	-	-	-	-	NO	Interim HMTV
Magnesium	n.p.	*245*	340	470/250	597	N/A	*334*	350	440/290	535	N/A	YES	Different 80th %ile between seasons
Manganese	1.9	*251*	5	0.1	0.25	2 (<1%)	*342*	0.14	0.25	0.43	0	NO	Retain 95% DGV
Mercury	0.0006	*121*	<LOR	<LOR	<LOR	0	*169*	<LOR	<LOR	<LOR	0	NO	Retain 95% DGV

Analyte	ANZG 95% DGV	WET SEASON (JUNE-OCT)					DRY SEASON (NOV-MAY)					SSGV Recommended	Comment
		No.	Median	80th/ 20th %ile	95th %ile	n exc (%exc)	*No.*	Median	80th/ 20th %ile	95th %ile	n exc (%exc)		
Molybdenum	0.034	*195*	<LOR	0.005	0.005	1 (<1%)	*273*	<LOR	0.005	0.0054	4 (<2%)	NO	Retain 95% DGV
Nickel	0.011	*108*	<LOR	0.001	0.007	4 (4%)	*158*	<LOR	0.001	0.004	3 (<2%)	NO	Interim HMTV
Nitrate (N_NO$_3$) (T)	*29	*52*	0.13	0.64	1.3	0	*66*	0.09	0.19	0.36	0	NO	Retain 95% DGV
Nitrite (N_NO2)	n.p.	*41*	<LOR	0.01	0.02	N/A	*48*	<LOR	<LOR	0.03	N/A	NO	No SSGV recommended
N_NOx (E)	0.2	*40*	0.15	0.7	1.4	19 (47%)	52	0.09	0.3	0.4	16 (31%)	YES	Different 80th %ile between seasons
Nitrogen - total (E)	1.2	*60*	0.9	1.6	2.7	20 (33%)	*68*	0.8	1	1.4	6 (9%)	YES	Different 80th %ile between seasons
Phosphorus total (E)	0.065	*29*	0.02	0.03	0.03	0	*44*	0.02	0.03	0.03	2 (4%)	NO	Retain 95% DGV
Potassium	n.p.	*254*	8.85	10	11	N/A	*348*	15	19	26	N/A	NO	No SSGV recommended
Selenium	0.011	*27*	<LOR	0.011	0.03	5 (18.5%)	*32*	<LOR	0.006	0.03	4 (12%)	NO	Retain 95% DGV
Silica (SiO2)	n.p.	*74*	4	8.6	11.6	N/A	*96*	6.7	9.5	14	N/A	YES	Year-round SSGV
Sodium	n.p.	*265*	1300	1480 / 998	1,548	N/A	*365*	1,960	2,300 / 1,710	2,670	N/A	NO	No SSGV recommended
Strontium	n.p.	69	1.3	1.6	1.9	N/A	34	1.3	1.6	1.8	N/A	YES	Year-round SSGV
Sulfate (S_SO4)	n.p.	*252*	231	176 - 300	387	N/A	*338*	207	160 – 270	330	N/A	YES	Different 80th %ile between seasons
TDS Δ	n.p.	*182*	7,223	9,144 / 5,290	10,833	N/A	*251*	7,110	8,970 / 5,960	10405	N/A	YES	Instantaneous (reference condition) Δ
Turbidity (NTU)	20	*38*	3.8	4.6	14.0	1 (2.6%)	*####*	3.0	5.5	19.7	3 (5%)	NO	Retain 95% DGV
TSS	n.p.	*299*	3	5	13	N/A	*504*	3	7	21.7	N/A	YES	Different 80th %ile between seasons
Uranium	0.0005	*LOR*	-	-	-	-	-	-	-	-	-	NO	LOR unclear, questionable data
Vanadium	0.006	*LOR*	-	-	-	-	-	-	-	-	-	NO	Retain 95% DGV
Zinc*◊	0.028 / 0.025	*108*	<LOR	0.007	0.047	8 (7%)	*158*	<LOR	0.008	0.013	2 (1%)	YES	Seasonal adjusted DGV

Footnotes

Δ – Seasonal fluctuations in EC/ TDS may be ecologically significant, therefore it is recommended that post-dilution concentration be compared to upstream reference sites and kept within 18% of the reference EC/ TDS.

- adjusted DGV for ammonia using 80th percentile temperature and pH for each season (ANZG 2023a)

^ - interim DOC adjusted SSGV for copper using minimum DOC recorded for the Hotham River (775 mg/L)

*- Based on recent draft updates to this DGV, which are expected to supersede current advice. However, as toxicant DGVs are much higher than stressor (N_NO$_x$) DGVs, the SSGV for N_NO$_x$ is recommended as an operational guideline.

◊ - adjusted DGV for Zn using 80th %ile pH (per season), minimum hardness, and 20th %ile DOC (ANZG 2024b).

T – toxicant guideline values

E – eutrophication guideline values

n.p. – DGV not provided

6.3.1 Comparison against ANZG toxicant 95% DGVs

DOC adjusted DGVs for copper

Measured DOC values for the Hotham River ranged between 1 mg/L to 51 mg/L, with mean values well above the draft ANZG (2023b) default concentration of 0.5 mg/L in all months of the year (Figure 7). There was little seasonal variation in DOC, with an increase evident in June corresponding with the usual timing of increased flow (mean = 22.3 mg/L), and mean values between 12-16 mg/L for the rest of the year. Percentile ranges (20th, 50th, and 80th) were equal between seasons (Table 5). Therefore, a single interim SSGV across both wet and dry seasons is applicable.

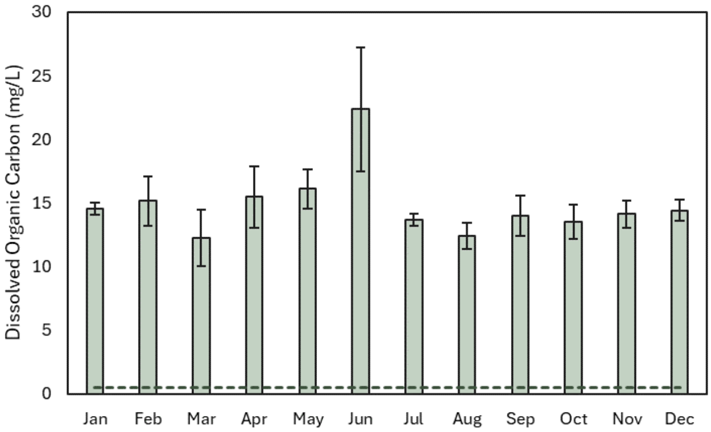

Figure 7. Dissolved organic carbon (mean ± SE) recorded at Hotham River site HRPB1 08-06-2012 to 03-12-2019. The ANZG default DOC threshold in absence of local data shown (0.5 mg/L; dashed line).

Using the lower 20th percentile value of DOC (12 mg/L), the proposed SSGV for Cu is 0.01 mg/L (Table 6). The lower 20th percentile is chosen as a conservative threshold, alternatively the 5th percentile value for DOC (9.7 mg/L) gives an interim Cu SSGV of 0.008 mg/L. It is not recommended to use less conservative values (i.e. the median or 80th percentile DOC), as this condition occurs infrequently. Ideally, DOC and Cu data from the same sample would be used to derive an adjusted DGV for Cu on a per sample basis (ANZG 2023b). At the very least, 24 months of recent DOC data should be collected to assess the validity of the interim SSGV at 20th percentile DOC, and updated if necessary. Then, if exceedances against the SSGV are recorded, paired DOC and Cu data should be collected and analysed to determine actual toxicity level at time of sampling.

Table 6. Indicative DOC adjusted 95% default guideline values for Cu. Dissolved organic carbon data from the Hotham River (HRPB1; 08-06-2012 to 03-12-2019; *N* = 101 samples) used to derive indicative adjusted DGVs.

Statistic	DOC (mg/L)	Cu 95% DGV	Conservatism
ANZG 2023a default	0.05	0.00047	Absence of DOC data
5th %ile	9.7	0.008 mg/L	High
20th %ile	**12**	**0.010 mg/L**	**Recommended**

Copper data for the Hotham River were supplied by NBG consisting of multiple sources, including ALS and MPL laboratories, and historic archived data. Notably, results from different laboratories show completely different orders of magnitude of data points that are above LOR (Figure 8). Because different laboratories were used at different periods of time, it is not possible to compare readings from the same time to determine whether differences in the data were due to actual environmental Cu, or different analytical methods between laboratories. Consequently, only data provided by ALS from 2015 to 2023 were compared to the adjusted interim SSGV (*N* = 262 measurements; Table 5).

A large proportion of data points (approx. 46.5%) for dissolved Cu were at or below the LOR (0.001 mg/L), and the 80th percentile value of 0.005 mg/L was well below the interim SSGV (Table 5). The 95th percentile value (0.05 mg/L) is above SSGV, and sporadic high values have been recorded (approx. 12.4% of data points; Figure 9). This indicates infrequent fluctuations in Cu in the Hotham River. Comparison of the effectiveness of the interim SSGV *in lieu* of per sample Cu and DOC adjustment was possible using a limited subset of paired data from HRPB1 (06-01-2016 to 03-12-2019, *N* = 47). The interim SSGV detected 8 exceedances, whilst per sample adjustment detected 10 (Figure 9). The fact this increase is small demonstrates an interim SSGV using adjusted DGV at 20th percentile DOC is fairly robust against natural variation. However should increases in Cu be detected after commencement of discharge, then analysis using per-sample, paired data is advised.

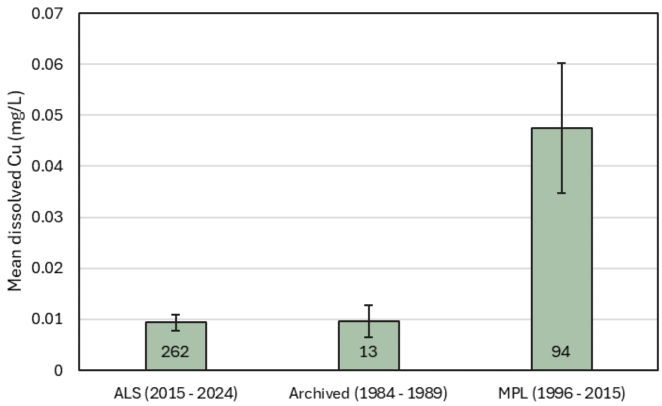

Figure 8. Mean Cu concentrations (± SE) from the Hotham River, data supplied from different sources. Date ranges and number of samples shown.

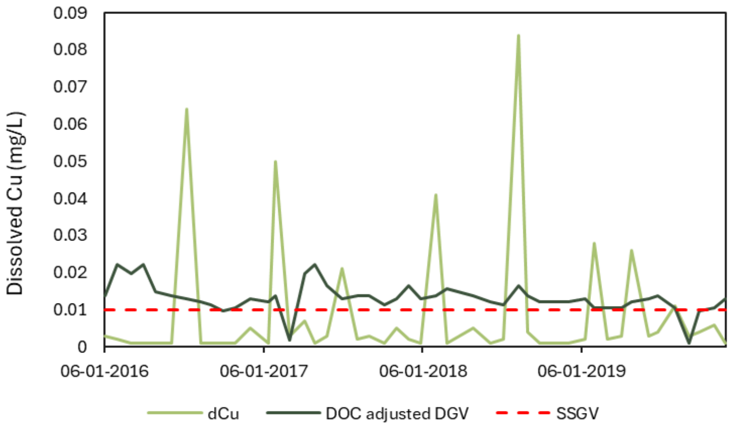

Figure 9. Comparison of per-sample dissolved Cu and DOC adjusted 95% DGV (mg/L), using paired Cu and DOC data recorded monthly between 06-01-2016 to 03-12-2019 from site HRPB1. This is compared against the interim SSGV using 20th percentile DOC for the Hotham River. Only ALS supplied data included.

Cobalt

Records of dissolved Co concentrations at the Hotham River vary spatially as well as over time, and between different laboratories. There were several very high recordings, including at HRPB1 on 04-08-2012 (0.43 mg/L) and 31-10-2012 (0.68 mg/L), and one likely spurious reading of 2.88 mg/L (HRPB1, 06-09-2017). Further assessment of the data revealed that the data supplied from MPL was significantly greater than the other data points (*N* = 10 samples; Figure 10). Data points from MPL were excluded in further analysis.

Spatial and temporal variation was also evident, with the vast majority of high readings recorded from Marradong, in August and September (Figure 11). Marradong is several kilometres downstream of HRPB1 and HRPB2, which are in proximity to Tullis Bridge and the NBG abstraction point. There is potential for a point source of Co downstream of NBG operations area contributing to elevated readings at Marradong, which may not be representative of the river reaches upstream. Nevertheless, the 80th percentile value for the Hotham River remained below LOR (<0.001 mg/L; Table 5), including when wet and dry seasons were analysed separately. The current ANZG DGV is retained as the SSGV for Co (0.0014 mg/L), however noting this guideline is considered a low reliability threshold due to inconsistencies in toxicity data (ANZG 2018). Discharge to the Hotham River should not increase the 80th percentile value for Co in either the wet or dry season.

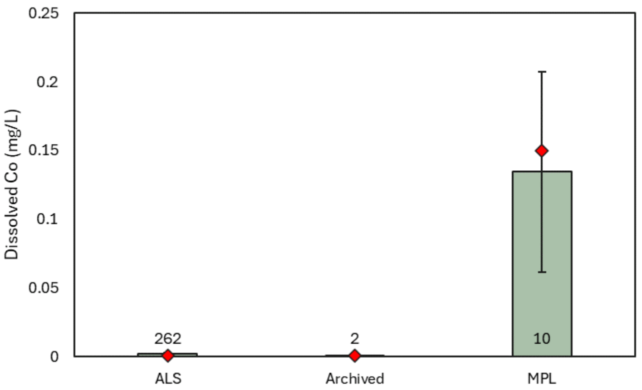

Figure 10. Dissolved Co concentration (mean ± SE) recorded from the Hotham River, between different data sources. Red diamond indicates 80th percentile value for each data source, number of samples shown.

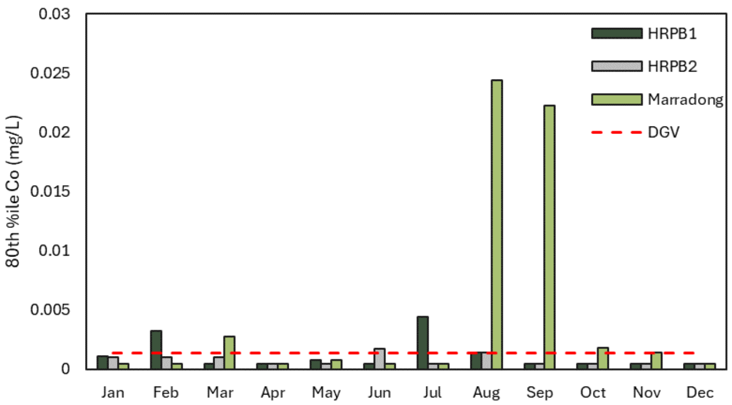

Figure 11. 80th percentile Co concentration (mg/L) per month from monitoring sites on the Hotham River (upstream -> downstream; HRPB1, HRPB2 and Marradong GS614244). Only ALS supplied data included.

Cyanide

After screening, no data for free, WAD or total cyanide were recorded above laboratory LORs, therefore further analysis was not possible. Under normal circumstances, cyanide is not expected to be detected in rivers at levels above laboratory LORs, or at levels which may warrant consideration of a site-specific guideline; the presence of detectable cyanide indicates an anthropogenic source. The current ANZG 2018 guideline for 95% species protection is retained as SSGV for free cyanide (0.007 mg/L), there are no default guidelines for WAD or total cyanide.

Aluminium

Background concentrations of Al in the Hotham River are generally low, with the 80th percentile in both seasons below the ANZG DGV of 0.055 mg/L (wet season = 0.03 mg/L; dry season = 0.02). There were sporadic high values recorded (e.g. wet season 95th percentile = 0.13 mg/L), with two records over 1.5 mg/L (Figure 12). Smaller creeks in the catchment (Gringer Creek and Boggy Brook) show strong seasonal pulses of dissolved Al, as they drain upland catchments rich in bauxite (SLR 2024a; SLR 2024b). Interestingly, a strong seasonal pulse was not detected for the Hotham River, the inference being that Al derived from creeklines is precipitated before reaching the Hotham, or that catchment runoff is sufficient to dilute seasonal Al pulses to background levels. Therefore, because concentrations are well below DGV at the 80th percentile, the default guideline value for Al is retained for the Hotham River.

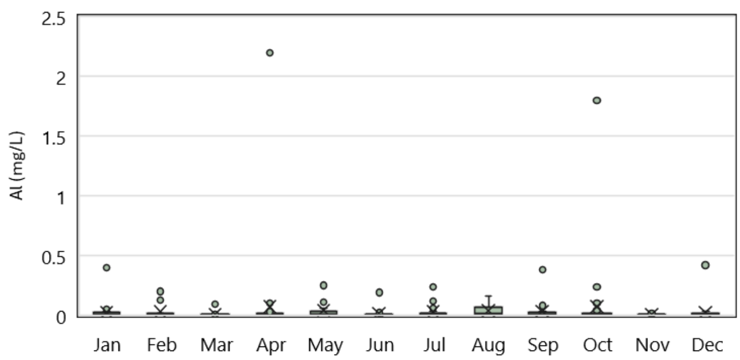

Figure 12. Distribution of dissolved Al (mg/L) recorded in each month from the Hotham River (16/12/1998 to 05/01/2024). The majority of records are below the ANZG 95% DGV (0.055 mg/L).

Ammonia

Ammonia fluctuates in the Hotham River seasonally, generally being highest in April and lowest in August (Figure 13). Fluctuation in ammonia and other analytes related to biological processes is typical in rivers that undergo seasonal low or no flow periods, as pools disconnect and channels dry. Ammonia is a product of decomposition, and can accumulate in disconnected pools during the dry season, and may also indicate livestock access during summer when other pools are not available.

Applying draft ANZG (2023b) guidance on the role of temperature and pH in determining the toxicity of ammonia, wet and dry season interim SSGVs were derived using 80th percentile values for pH and temperature from the Hotham River. Generally, temperature and pH are lower during the wet season months, due to increased flow and lower ambient temperatures (Figure 14 and Figure 15). Using pH 7.6 and temperature 16ºC (wet season) and 7.8 and 25ºC (dry season), interim SSGVs are 0.67 mg/L and 0.29 mg/L respectively. These values are an order of magnitude above the 80th percentile values for the Hotham River (0.07mg/L and

0.06mg/L respectively; Table 5). Ideally, paired pH and temperature data should be compared to ammonia data on a per-sample basis, and should be implemented if increases are detected.

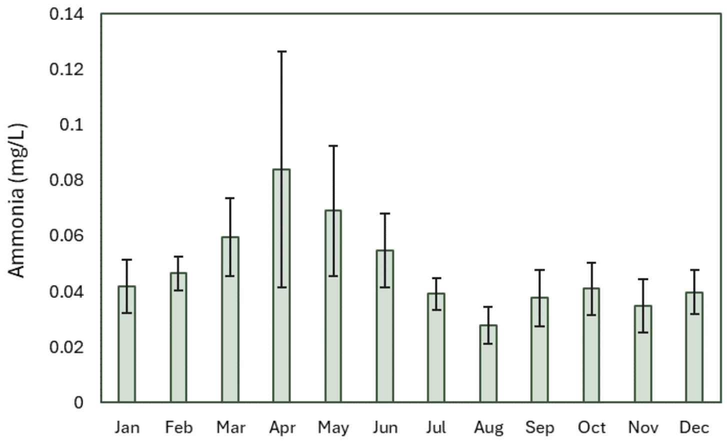

Figure 13. Mean ammonia concentration (mg/L) per month, recorded from the Hotham River between 24-10-1984 and 05-01-2024.

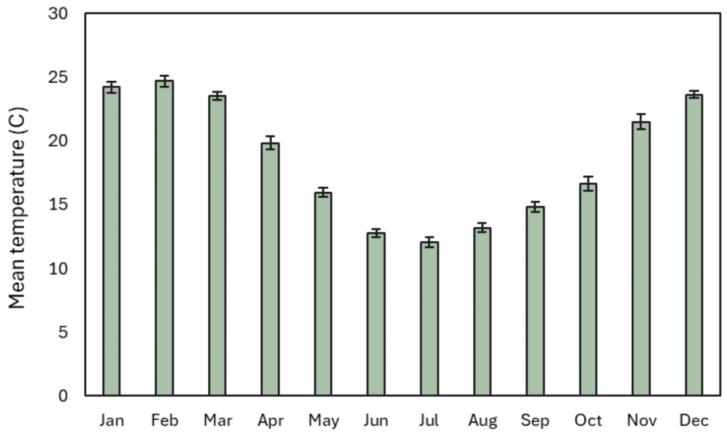

Figure 14. Mean temperature (ºC) per month recorded from the Hotham River between 20-09-1995 to 05-05-2024.

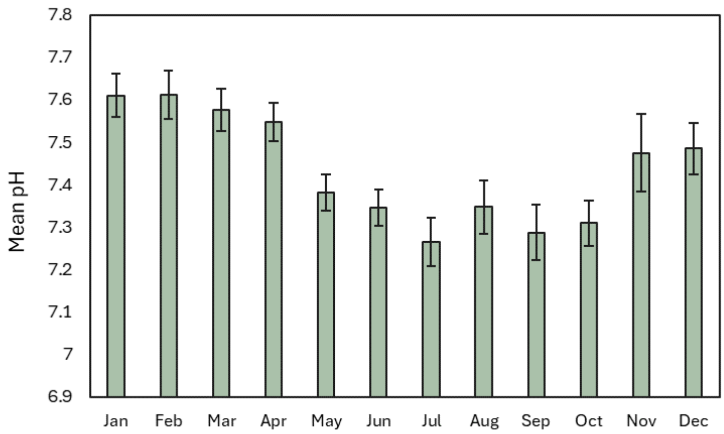

Figure 15. Mean pH per month recorded from the Hotham River between 10-05-1988 to 05-05-2024.

6.3.2 Hardness modified trigger values (HMTV)

Recommended algorithms for the modification of guideline values are provided for Cd, Cr, Ni, Pb and Zn. Data for the Hotham River indicate hardness is very high, ranging from 775 mg/L to 3,000 mg/L. The ANZG (2024a) considers surface waters to have high hardness at or above 150 mg/L $CaCO_3$ (ANZG 2024a), and the background concentrations of the Hotham River are well in excess of these levels. Therefore, as a conservative threshold for calculation of HMTVs, the minimum value recorded of 775 mg/L was used. Both the unmodified DGVs and interim HMTVs at 95% and 99% species protection are presented in Table 7.

Table 7. Summary of proposed interim hardness-modified trigger values (HMTVs) for Cd, Cr, Ni and Pb, compared to unmodified DGVs for 95% and 99% species protection (%SP). HMTVs derived using minimum recorded hardness for the Hotham River (775 mg/L; 12-06-1989 to 08-04-2005; *N* = 49). Recent hardness data paired with metals listed below would be required to update interim SSGVs to full operational SSGVs.

Analyte	Unmodified DGV 30 mg/L CaCO3		Interim HMTV 775 mg/L CaCO3	
SP%	***95%***	***99%***	***95%***	***99%***
Cd	0.002	0.00002	0.0036	0.0011
Cr	0.001	0.00001	0.014	0.00014
Ni	0.011	0.008	0.17	0.12
Pb	0.0034	0.001	0.21	0.016

Cadmium

Data for Cd from different laboratories likely had different LORs, noting the LOR for MPL and historic values are unknown. However, comparing the mean (± SE) between ALS and archived values suggests that the LOR for the latter was much greater than that reported by ALS (Figure 16). Similarly, the 95th percentile for ALS data was equal to the limit of reporting (0.0001 mg/L), whereas 0.005 mg/L for historic archived values. Of the historic values, 63.6% = 0.001 mg/L, and 24.6% = 0.005 mg/L, indicating these were LOR rather than true Cd concentrations. Among the ALS dataset, 97% = 0.0001 mg/L, which is the known LOR. Given ambiguity in LOR between data sources, only the ALS dataset (02-09-2015 to 05-01-2024; *N* = 260) were retained for calculation of interim HMTV.

As established, the vast majority of Cd data were below LOR, including the 80th percentile value (Table 5). The maximum value (0.05 mg/L, HRPB1 06-09-2017) was the only exceedance of the 95% HMTV (0.0036 mg/L) in this dataset, and only two exceedances compared to the ANZG DGV (Figure 17). Clearly, background levels of Cd are very low for the Hotham River.

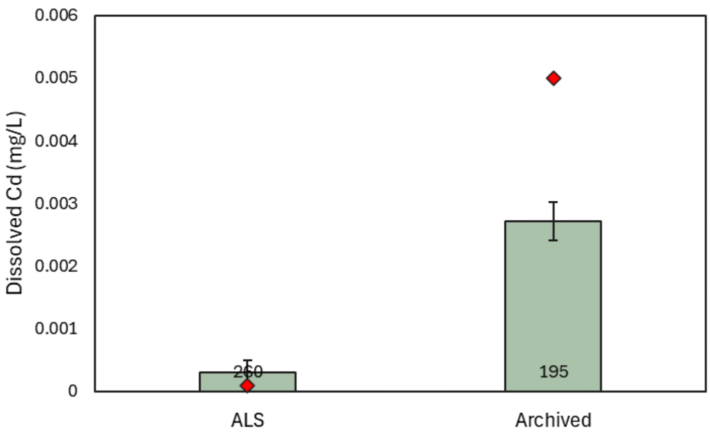

Figure 16. Cadmium (mean ± SE) recorded from the Hotham River, from ALS (02-09-2015 to 05-01-2024) and archived values (23-01-1996 to 09-01-2008). Red point shows the 95th percentile value, which is 0.0001 mg/L (ALS, equal to LOR) and 0.005 (archived value, LOR unknown). Number of samples shown.

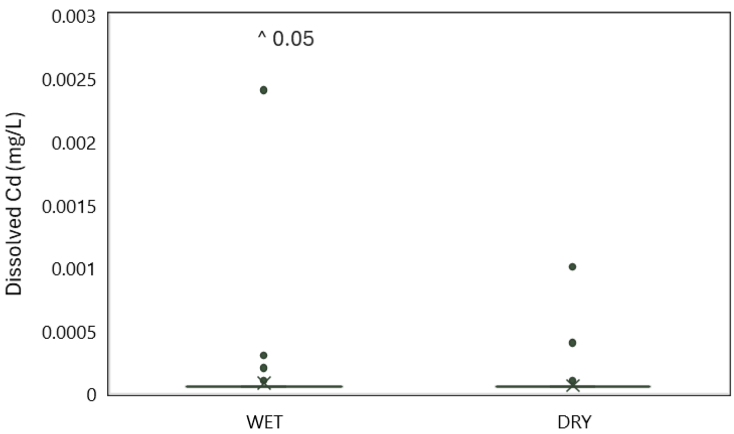

Figure 17. Distribution of Cd data (mg/L) recorded from the Hotham River between 02-09-2015 to 05-01-2024. High outlier indicated.

Chromium

All records of Cr for the Hotham River were below LOR (0.001 mg/L), precluding the further analysis. The 95% HMTV (0.014 mg/L) is retained as interim SSGV.

Nickel

The majority of data for Ni were below or equal to LOR, such that the 95th percentiles for both the wet and dry season were much below both the HMTV and unmodified 95% DGV

(0.007mg/L and 0.004 mg/L, respectively). There was a single exceedance of the HMTV (0.17 mg/L), recorded from Marradong on 10-07-2017 (0.23 mg/L).

Lead

The toxicity of Pb is strongly ameliorated by hardness, such that the interim HMTV (0.21) is two orders of magnitude greater than the default DGV (0.0034 mg/L at 30 mg/L as $CaCO_3$; Table 7). Of the available data for the Hotham River, it was unclear whether reported values provided by NBG represented actual concentrations, or reflected data reported as less than laboratory LORs, with the less than detection symbol removed. Approximately 63% of records were 0.001 mg/L, and 23% were 0.005 mg/L (Figure 18). As a result, the 80th percentile value (0.005 mg/L) was above the ANZG DGV, however this potentially indicates data at laboratory LORs, and LORs are too high to meaningfully compare against DGVs. Nevertheless, background Pb concentrations for the Hotham River were low, and there were no exceedances of the HMTV.

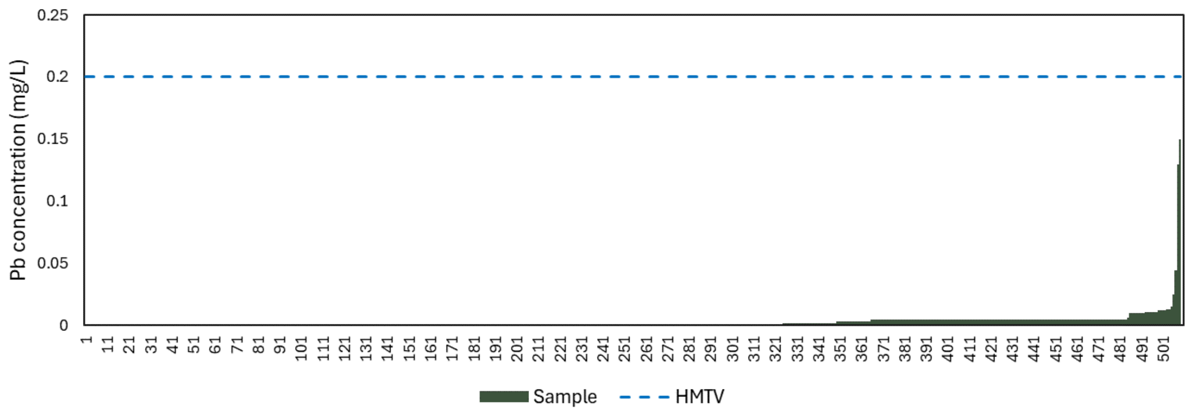

Figure 18. Dissolved Pb (mg/L) records in ascending order. It is suspected that the majority of the data reflects differences in LOR, rather than actual Pb concentrations. 95% HMTV shown (dashed line).

Zinc

Recent draft guidance for Zn toxicity takes into account modifying factors including DOC, pH and hardness, and once published will supersede current protocol for hardness modification presented in Warne et al., (2018). Hotham River background hardness exceeds the maximum threshold considered by ANZG (2024b), which presents DGVs under the caveat that DOC, pH or hardness values outside the ranges given may be less reliable. However, as DOC and pH are well within the ranges given, therefore it is likely that if the new guidance be adopted as ANZG DGVs, these would be appropriate for use in the Hotham River.

There were differences in mean zinc concentrations between ALS and MPL laboratories, whilst outliers occurred in both datasets (Figure 19) data from ALS was predominantly below LOR, whereas MPL reported higher values with a significantly higher mean ($t = 4.6$, $df = 459$, $P < 0.0001$). Because data from ALS and MPL were collected at different times, it is not possible to determine whether differences occurred in the environment, or were due to different analytical methods. Therefore, only ALS data (02-02-2015 to 17-04-2024) were compared to adjusted DGVs.

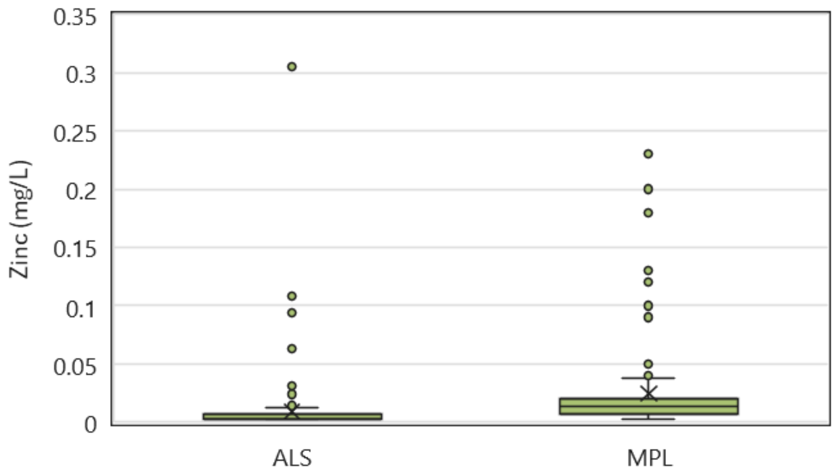

Figure 19. Distribution of dissolved Zn data (mg/L) recorded from the Hotham River, data from ALS or MPL laboratories.

Interim SSGVs were derived using the 80th percentile pH for the wet and dry season (7.6 and 7.8 respectively), with minimum hardness (775 mg/L) and 20th percentile DOC (12 mg/L) to derive conservative adjusted guidelines. The adjusted 95% DGV for the wet season is 0.028 mg/L and for the dry season 0.025 mg/L (Table 5). Approximately 7% of records in the wet season were above the SSGV, and the 95th percentile (0.047 mg/L) was equal to the adjusted 90% DGV (ANZG 2024b). Whereas, there were few records above SSGV in the dry season (~ 1%; Table 5). This indicates very occasional elevations of Zn occur during wet season flows.

6.3.3 Nutrients

A range of elements can act both as stressors and as toxicants. Nutrients for example, can be a stressor at low levels by promoting growth of algae and associated adverse effects on aquatic ecosystems (i.e. eutrophication), but can also be direct toxicants in aquatic species at higher concentrations. Therefore, two sets of guideline values may be required for such analytes.

Concentration of nitrogen-based analytes and phosphorus indicate some pre-existing nutrient enrichment of the Hotham River, most likely attributable to agricultural land uses. Several analytes of nitrogen are used to determine quantities present against stressor DGVs, as indicators of eutrophication (including nitrite/nitrate N_NO_X and total N), and much higher thresholds for nitrate (N_NO_3) as a direct toxicant (Hickey et al., 2013; ANZG 2024a). Although eutrophication thresholds do not imply risk of direct toxicity to aquatic fauna, negative consequences of eutrophication include nuisance and toxic algal growth, which can have significant ecosystem impacts including anoxia and fish kills, as well as aesthetic and amenity impacts.

<u>Nitrate (N_NO_3) direct toxicant guidelines</u>

Data for N_NO_3 included ALS, MPL laboratories and archived data, which had differences in the means and ranges of values (Figure 20). Readings from MPL laboratories (1994-2004) were markedly higher than ALS and archived data, and with unknown LORs. Therefore MPL supplied data were excluded from further analysis. In addition, two datapoints in the archived datasets exceeded SSGV, both on 12/06/1989 (HRRV 22 4 mg/L; HRRV 23 5 mg/L). Given these were recorded on the same day, and a data transfer error cannot be ruled out, these datapoints were also excluded.

A seasonal pulse in N_NO$_3$ is evident with the onset of winter (June and July), suggesting a flush of nutrients entering the Hotham from the catchment, which is not unexpected (Figure 21). Mean N_NO$_3$ is then low through the remainder of the flow season (September and October) and remains low throughout the dry season.

As above, guidelines designed to be protective of direct toxicity affects from N_NO$_3$ are much higher than those protective of eutrophication. The ANZG are likely to recommend guideline values that supersede previous advice (Hickey et al., 2013), which are expected to be higher under high hardness scenarios (i.e. in the order of ~29 mg/L; ANZG 2024a[9]). There are no records of N_NO$_3$ in the Hotham River that exceed this level, with a maximum value in the retained dataset of 1.64 mg/L. Should discharge cause increases in the 80th percentile N_NO$_3$ concentration to levels approaching the toxicant DGV, it is very likely that the negative impacts of eutrophication would have adverse impacts on ecosystem health, prior to the toxicant DGV being exceeded and direct toxicity effects being manifest. Ecosystem health impacts could include toxic cyanobacterial blooms and loss of aquatic habitat due to excessive primary production (algal mats etc). Particularly following algal/cyanobacterial blooms, loss of dissolved oxygen can ensue, particularly in summer when flows are low and pools are acting as critical refuges. Therefore, it is not recommended that the N_NO$_3$ DGV be used as an operational guideline for discharge to the Hotham River. Rather, the eutrophication (stressor) guidelines applied to N_NO$_x$ (see below) are recommended for use, as they are designed to be protective of ecosystem values, not solely against the mechanism of direct toxicity.

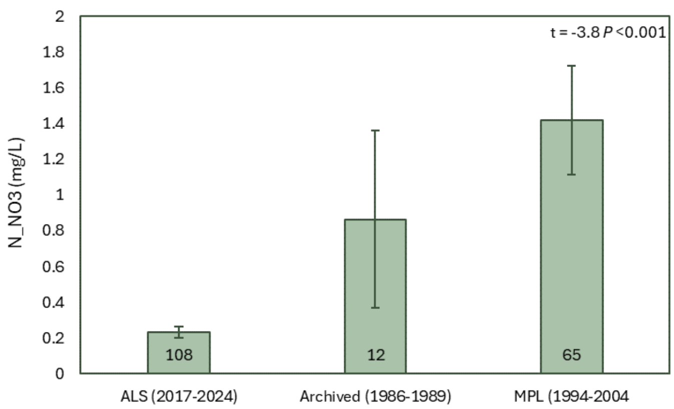

Figure 20. Nitrate concentrations (N_NO$_3$ mg/L) reported from various laboratories (mean ± SE). Number of samples and date ranges shown. A t-test between ALS and MPL data found differences in the mean were significant (df = 170, *P* <0.001).

[9] High hardness as defined as >150 mg/L as $CaCO_3$

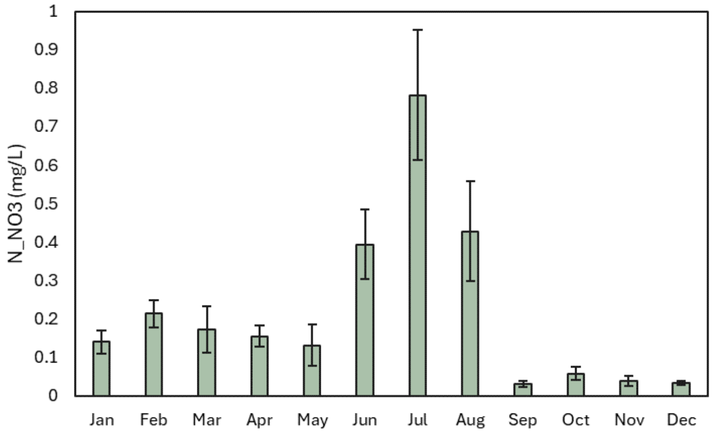

Figure 21. Mean N_NO3 concentrations (mg/L) per month collected from the Hotham River, between 25-10-1984 to 05-01-2024. Data only includes ALS and archived data.

Nitrite/nitrate (N_NOx) eutrophication guidelines

Similarly to nitrate, a seasonal pulse of N_NOx is evident in the first three months of the wet season (June to August), followed by lower concentrations over the remaining duration of the wet season into the early dry (Figure 22). Differences between wet and dry seasons were significant (t = 3.38, df = 90, *P* 0.001; Figure 23).

The stressor DGV for N_NO_X is intended to be protective against habitat loss due to eutrophication. The DGV (0.2 mg/L) was frequently exceeded in the wet season, with an 80th percentile value of 0.7 mg/L (Table 5), the 80th percentile for the dry season was also above the DGV at 0.3 mg/L. Therefore, it is recommended that seasonal SSGVs for N_NO_X (0.7 mg/L wet; 0.3 mg/L dry) be applied to the Hotham River to maintain the system in its current condition regarding N_NOx. It is also recommended that the N_NO_X SSGV inform discharge operations (i.e. water treatment planning) as a wholistic protection GV, rather than the toxicant DGV for N_NO_3.

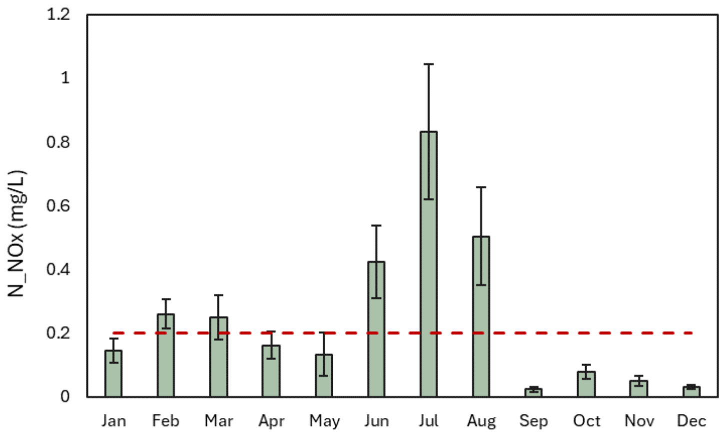

Figure 22. Mean N_NOx concentrations (mg/L) per month recorded from the Hotham River between 16-07-2017 to 08-04-2024.The ANZG DGV shown (0.2 mg/L, red dashed line).

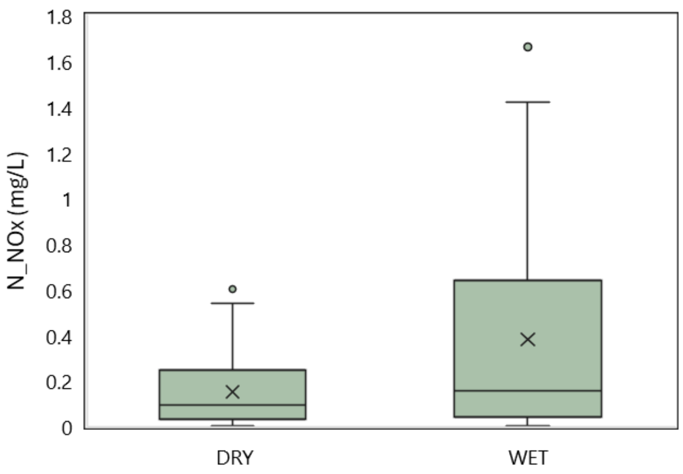

Figure 23. Distribution of N_NO_X (mg/L) between dry season and wet season records.

Total N and total P eutrophication guidelines

Unlike nitrate and nitrite/nitrate N, mean concentration for total N were not different between laboratories (*P* >0.95), and thus MPL data were retained. Seasonal differences in total N concentrations were evident, though not as pronounced as the former analytes, with a peak occurring during June and July, which exceeded the DGV, but with the mean remaining below DGV (1.2 mg/L) for the other months of the year (Figure 24 and Figure 25). The 80th percentile value among wet season samples was greater than the DGV (1.6 mg/L), whereas the dry season was below SSGV (1.0 mg/L). Therefore, a seasonal SSGV is proposed for the wet season (1.6 mg/L), whereas in the dry season the DGV (1.2 mg/L) is recommended.

There were no records were above the ANZG DGV for total P (0.065 mg/L) in the wet season, and only two in the dry, the 95th percentile for both seasons was 0.03 mg/L (Table 5). The ANZG DGV is retained for total P.

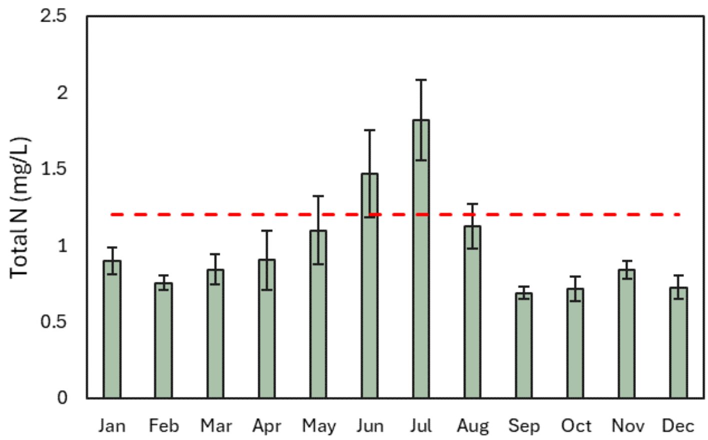

Figure 24. Mean total N concentrations (mg/L) per month recorded from the Hotham River between 04/09/1989 to 08/04/2024, compared against the ANZG DGV (1.2 mg/L; red dashed line).

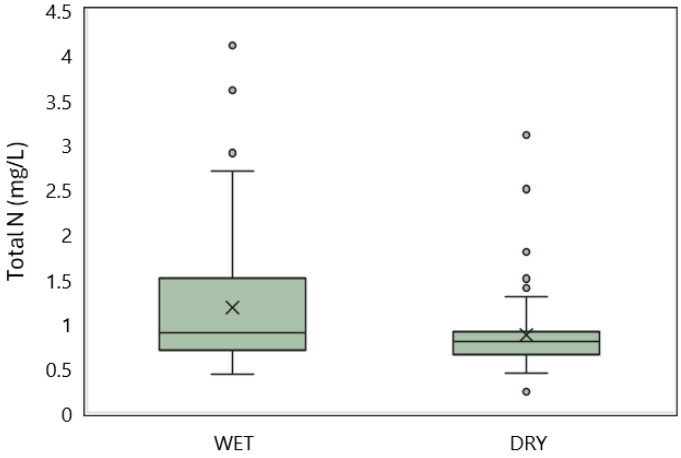

Figure 25. Distribution of total N (mg/L) between dry season and wet season records

6.3.4 Analytes without DGVs associated

Barium

Barium is a naturally occurring element considered to have low toxicity, however industrial activities can increase Ba concentrations in surface waters (Donald 2016; Verbruggen et al., 2020). The solubility of Ba is affected by concentrations of salts, particularly S_SO_4, affecting uptake by aquatic organisms (Golding et al., 2018). Ecotoxicological studies indicate toxic affects of dissolved and precipitated Ba in aquatic species (e.g. Golding et al., 2018), however there is currently no accepted guideline value for Ba for inland waters, including for Australia. The NWQMS (2011) recommend a drinking water guideline of 2 mg/L, although this is likely to be significantly higher than ecosystem protection limits. Verbruggen et al., (2020) indicate 0.093 mg/L threshold for expected harm to aquatic fauna in the Netherlands, however the level of species protection was not assessed, and cannot be inferred.

Background concentration of Ba were generally above 0.1 mg/L, and well below the 2 mg/L NWQMS (2011) drinking water guideline (Table 5). There were differences in wet and dry season Ba concentrations in the Hotham River, with slightly higher values recorded in the dry season (Figure 26). Therefore, seasonal interim SSGVs are proposed using the 80th percentile values from the monitoring dataset for the wet season (0.14 mg/L) and dry season (0.17 mg/L).

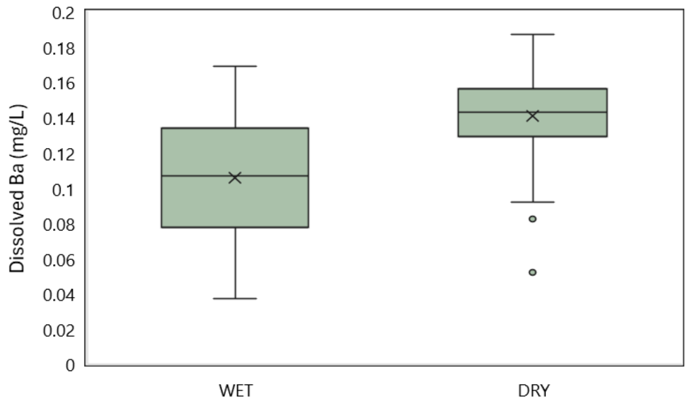

Figure 26. Distribution of Ba (mg/L) between dry season and wet season records.

Iron

Iron is the fourth most common element in the Earth's crust, and is often a major constituent of soils (especially clays) and is found in waterways as a result of natural runoff, erosion of clay-based soils, and other geologic sources (ECCC, 2019). While Fe is an essential trace element for both plants and animals, acute toxicity to aquatic insects has been reported (Warnick & Bell 1969, ANZG 2018). Background concentrations of the Hotham River show seasonal differences in Fe concentrations, with higher concentrations during the wet season than the dry (Figure 27). There was a significant difference in the mean dissolved Fe between seasons ($t = 2.4$, $df = 197$, $P = 0.02$) which was more pronounced for total Fe ($t = 4.0$, $df = 115$, $P < 0.001$). Currently, the freshwater DGV for Fe is under review, and until updated guidance becomes available, it is recommended that seasonal SSGVs based on the 80th percentile for dissolved Fe be applied to the Hotham River (wet = 0.17 mg/L, dry = 0.11 mg/L; Table 5).

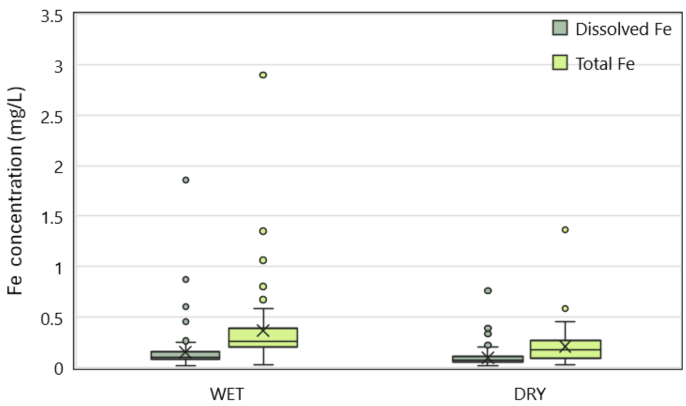

Figure 27. Distribution of dissolved Fe and total Fe (mg/L) recorded from the Hotham River between 06-07-2012 to 08-04-2024, between wet (June to October) and dry seasons (November to May).

6.3.5 Turbidity and TSS

Turbidity

Turbidity tended to be slightly higher in the dry season (Figure 28), which is typical in rivers that undergo seasonal low flows and disconnection of pools (Boulton 2003; Gómez et al., 2017). In particular, the late autumn months had the greatest mean turbidity, and the lowest occurred in the peak of the wet season (July; Figure 29). However, turbidity was highly variable year around, such that the difference between wet and dry season mean turbidity was not significant (t-test $P > 0.36$). Furthermore, the DGV for southwest rivers of 20 NTU was rarely exceeded in either season (Table 5), therefore it is recommended that the DGV be retained for the Hotham River.

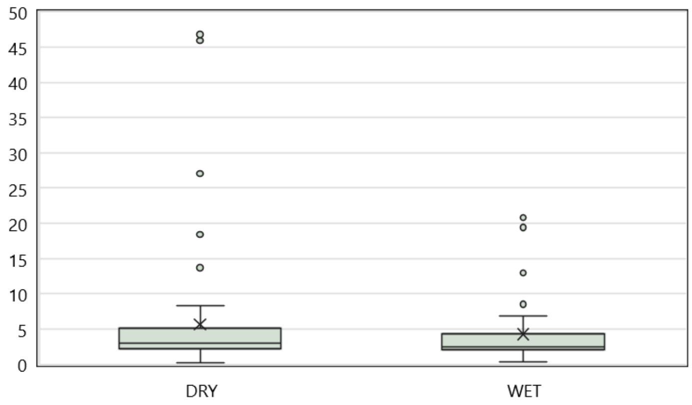

Figure 28. Distribution of turbidity (NTU) recorded from the Hotham River between 05-10-2010 to 08-04-2024, between wet (June to October) and dry seasons (November to May).

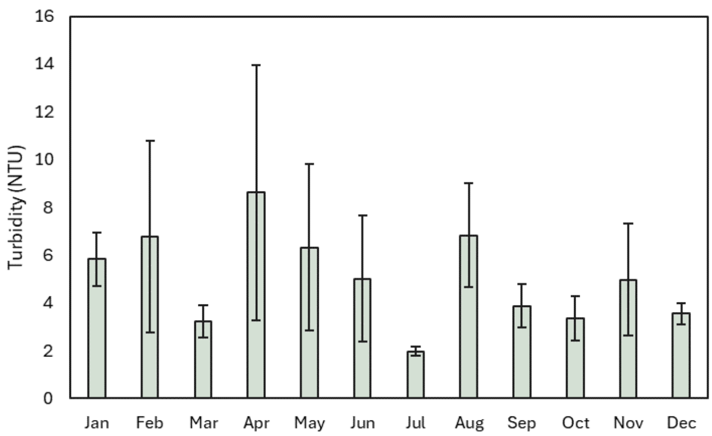

Figure 29. Mean turbidity (NTU) per month recorded from the Hotham River between 05-05-2010 to 08-04-2024.

TSS

There was some seasonal variation in TSS, with lower values tending to occur in August and September, and higher values in the summer and autumn months (Figure 30). However, overall seasonal differences were marginally non-significant ($t = -1.6$, $df = 502$, $P = 0.06$; Figure 31). This reflects that the percentile ranges between seasons was similar, and there were more frequent high values recorded in the dry season (95^{th} percentile wet = 19 mg/L, dry = 25 mg/L; Table 5). Therefore, the recommended SSGV for the wet season is 7.0 mg/L TSS, and for the dry season 8.0 mg/L TSS.

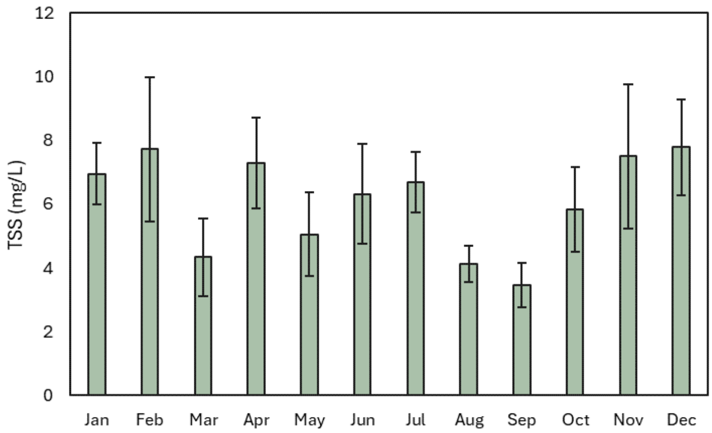

Figure 30. Mean total suspended solids (TSS; mg/L) per month recorded from the Hotham River between 31-11-1993 to 08-04-2024.

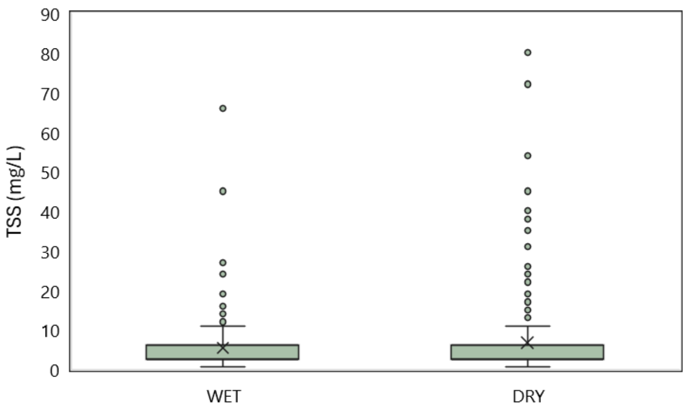

Figure 31. Distribution of total suspended solids (TSS) recorded from the Hotham River between 30-11-1993 to 08-04-2024, between wet (June to October) and dry seasons (November to May).

6.3.6 Electrical conductivity, TDS and ions

Electrical conductivity and TDS

Electrical conductivity (EC; µS/cm) and total dissolved solids (TDS, mg/L) are inter-related measures of salinity, with TDS representing the sum of dissolved solids (including ions and dissolved organics), and EC reflecting the conductivity of the water in microsiemens per centimetre, with an approximate conversion possible between the two (TDS = (EC x 1000)/2). The Hotham River is saline, with median EC over 10,000 µS/cm in both seasons, and 80th percentiles over 13,000 and 11,000 µS/cm in the wet and dry season respectively (Table 5). Some seasonal complexity in EC and TDS is evident, with a peak in June/July followed by a minimum in August (Figure 32 and Figure 33). Low EC and TDS is then observed throughout the latter months of the wet season, and increases again in the early dry season, followed by decreases in the later dry season months (February to April). Comparison of monthly percentiles shows that variation is generally greater in July and August, whereas variation was lower during October and January to April (Figure 34). Altogether, complexities in seasonal variation in TDS and EC reflects seasonally predictable rainfall patterns. This includes a first flush of salts accumulated from the catchment with the onset of winter flows in May/June, which is subsequently diluted by winter rainfall by August/September. Reduction in winter flows, evapo-concentration and return to baseflows during the early dry season (November to January) shows an increase in salts, followed by a slight decrease as catchment baseflows decline, and potentially an increased dominance of local groundwater in the late dry season. There is potential that seasonal reductions in salinity assist life history events, such as reduced salinity in spring coinciding with fish migration and spawning, or reduction in osmotic stress during late dry season periods when refuge pools are critical habitat.

Although there is no defined DGV for TDS per se, there is an ANZG DGV for EC applicable to southwest rivers of 250 µS/cm. This DGV was designed to be protective of the negative impacts of salinisation on fresh water systems, and is now superfluous for the Hotham River. Although freshwater adapted species may have been lost from the system, additional species will be lost if salinity continues to increase. Therefore, there is merit in maintaining current levels of salinity to protect current diversity, including observed month by month differences in accordance with flow regime. Furthermore, discharge of waters that are significantly fresher than catchment flows, especially in summer, with low flows, may have limited mixing and result in density stratification, with less dense freshwater overlying more dense saline water in pools, and should also be avoided.

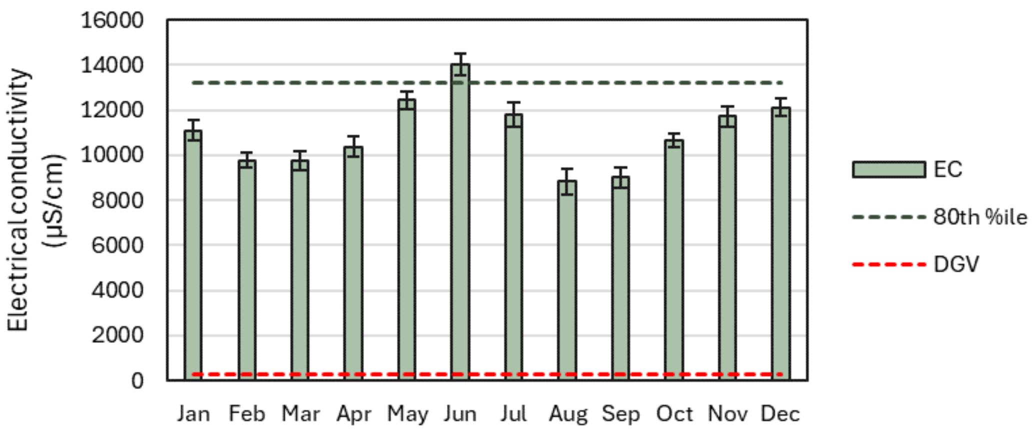

Figure 32. Electrical conductivity µS/cm (mean ± SE) per month recorded from the Hotham River between 04-01-2012 to 05-05-2024, overlain is the ANZG DGVs for southwest rivers (250 µS/cm) and the 80th %ile value.

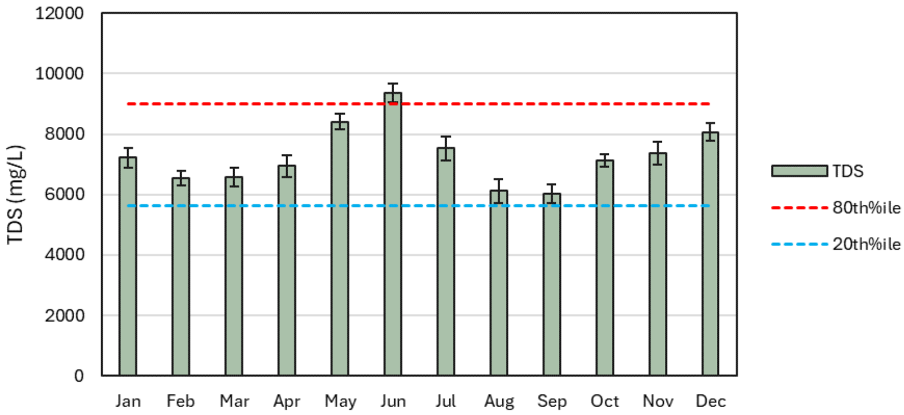

Figure 33. TDS (mg/L) recorded from the Hotham River, analysed by season. Wet = June to October; Dry = November to May. Data collected between 04-01-2012 to 05-05-2024.

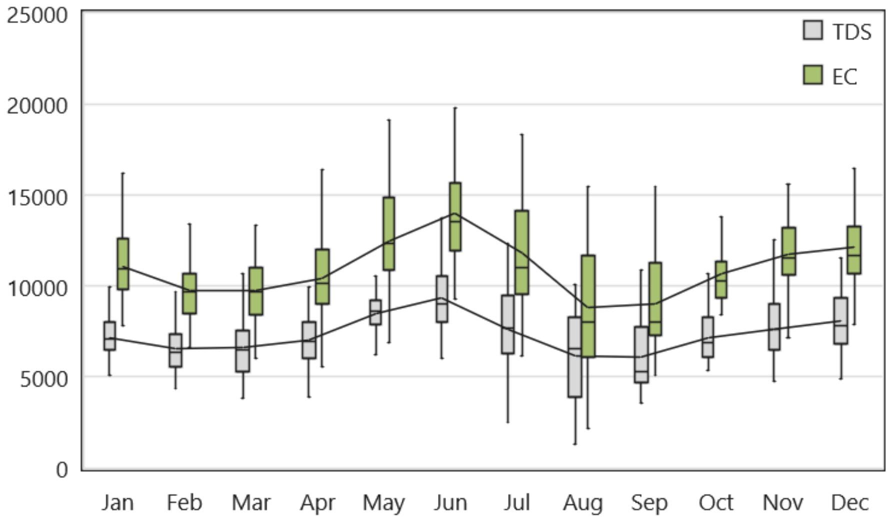

Figure 34. Distribution of total dissolved solids (TDS, mg/L) and electrical conductivity (EC, µS/cm) per month recorded at from the Hotham River between 06-07-2010 to 05-05-2024. Line links mean values.

Adhering to the ANZG (2018) protocol, discharge should not cause an increase in the 80th percentile value EC (or TDS). However as discussed, there are challenges presented in setting "seasonal" SSGVs for salinity due to variations over the course of the year. As such, direct application of a threshold EC or TDS value as an SSGV may not be a pragmatic approach to post-discharge monitoring. Rather, it is recommended that post-discharge concentrations be compared instantaneously to upstream reference condition above the discharge point (i.e. at Hotham Weir). Between months, the average difference between the 50th and 80th percentile value was approximately 18% ± 2% (for both EC and TDS; Figure 35). Therefore, it is proposed that at the point of discharge, the maximum post-dilution EC/TDS levels should not exceed background condition by more than 18% (measured at the Hotham Weir), which would indicate a corresponding increase in the 80th percentile value, and thus an exceedance.

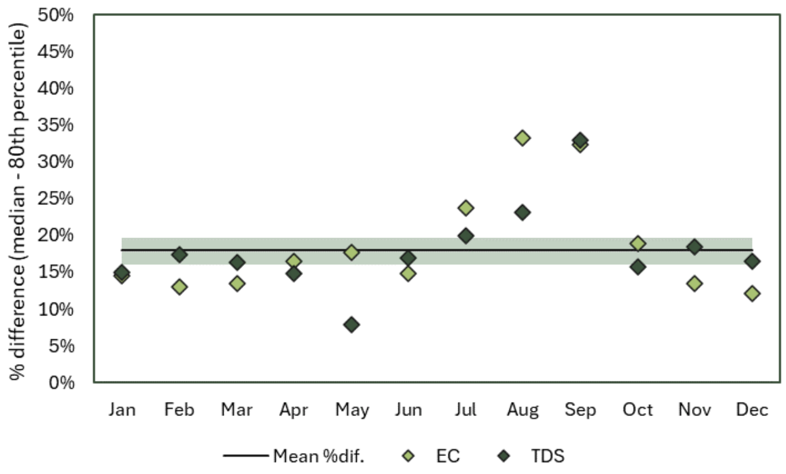

Figure 35. Percent difference between EC and TDS 50th and 80th percentiles per month. The mean difference (± SE) shaded.

Ionic composition

Ionic composition of waters of the Hotham River were dominated by Na anions and Cl cations, followed by Mg and S_SO_4, in the order Na > Mg > Ca > K and Cl > S_SO_4 > HCO_3 > CO_3 (Table 8). Concentrations of Na and Cl were stronger in the dry season, consistent with evapo-concentration in summer pools, as were concentrations of Ca and HCO_3.

Comparison of the dominant and sub-dominant cations (Na and Mg) reveals the relationship is linear, that is, increases is both analytes are proportional to increases in TDS (Figure 36a). This is similar for anions, where the concentrations of dominant and sub-dominant anions (Cl and S_SO4) increase linearly, rather than increased concentrations of one ion altering the relative proportion of the other. The concentrations of both Na+Mg and Cl+S_SO4 are directly linearly related to TDS (Figure 36b), showing that under natural circumstances, ionic dominance is likely to be highly conserved with fluctuations in TDS. This is to be expected when the source of TDS is generated from groundwater baseflows, however introduction of a new source of ions (e.g. mine-derived discharge) may alter proportions of dominant ions relative to background catchment conditions. Therefore, discharge should not substantively alter ionic composition, both in terms of total concentrations of sub-dominant ions or relative proportions of ionic composition. Application of instantaneous guidelines for EC/TDS, based on upstream reference concentrations (as described above) are sufficient for use as a surrogate for GVs for individual ions, because an increase in ionic concentrations would cause a corresponding increase in EC/TDS. If exceedances are detected (i.e. EC/TDS monitoring data exceeds reference data by 18%, equivalent to an increase in the median to the 80th percentile reference condition) then further investigation into changes to ionic dominance should also be conducted.

Table 8. Summary of composition of major cations and anions of the Hotham River, showing 20th to 80th percentile ranges between wet and dry seasons.

Cations			Anions		
	wet	***dry***		***wet***	***dry***
Na	999 – 1,480	1,710 – 2,300	Cl	2,044 - 2,988	3,500 - 4,970
Mg	250 - 470	290 - 440	S_SO_4	176 - 300	160 - 270
Ca	81 - 158	120 - 170	HCO_3	82 - 142	155 - 235
K	7 - 10	12 - 19	CO_3	<1	<1

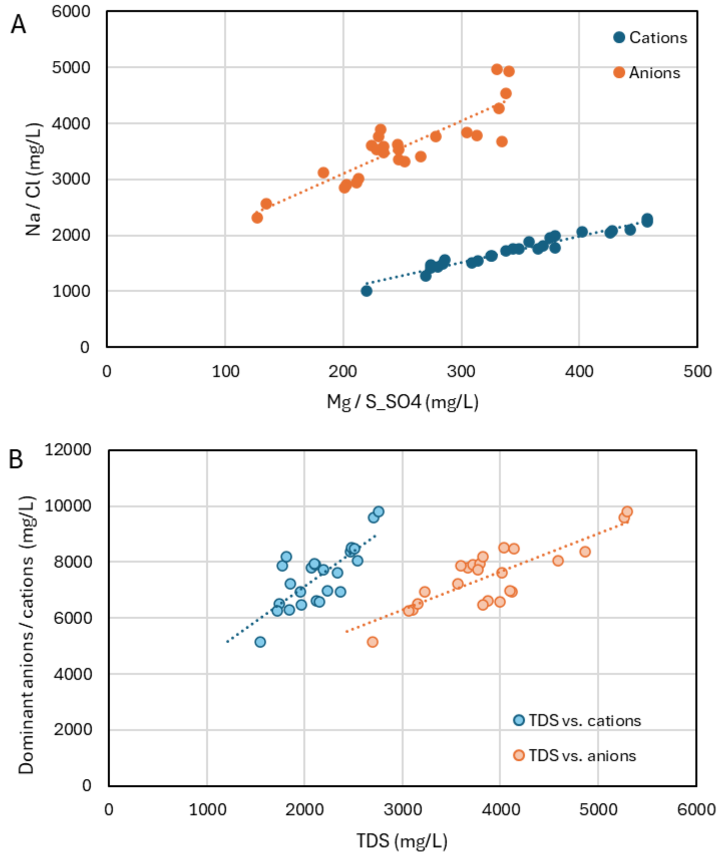

Figure 36. Concentrations (mg/L) of dominant and sub-dominant cations (Na and Mg) and anions (Cl and S_SO4) measured in 2023 (top – A). Combined dominant and sub-dominant cations/anions against TDS (bottom – B). All data pairs were recorded from the same samples and thus are representative.

6.3.7 Interim SSGVs

Table 9. Proposed interim site-specific guideline values (SSGV) for the Hotham River applicable to the wet season defined as June to October, or flow rate > median (2015-2023). Units are mg/L unless specified. (T) – DGV/SSGV for direct toxicity; (S) – DGV/SSGV for stressors. #Further risk assessment based on modelled

Analyte	Toxicant - T Stressor - S	ANZG 95% DGV	80th %ile value	WET SEASON Interim SSGV
EC (uS/cm)#	S	250	7,847 - 13,524	**#Reference +18%[A]**
pH (H+)	S	6.5 - 7.5	6.9 - 7.6	**6.5 - 7.8[B]**
Temperature (°C)	S	-	16.3	**16.3**
Alkalinity	-	-	135.6	**-**
Hardness	-	-	-	**-**
Aluminium	T	0.055	0.03	**0.055**
Ammonia*	T	0.67	0.06	**0.67[C]**
Arsenic	T	0.024	0.005	**0.024**
Antimony	T	0.009	<LOR	**0.009**
Boron	T	0.94	0.11	**0.94**
Barium	T	-	0.14	**0.14**
Bicarbonate (HCO3)	-	-	143	**-[A]**
Cadmium	T	0.002	<LOR	**0.004[D]**
Calcium	-	-	158	**-[A]**
Carbonate (CO3)	-	-	<LOR	**-[A]**
Cobalt	T	0.0014	<LOR	**0.0014**
Chromium	T	0.001	<LOR	**0.014[D]**
Chloride	-	-	3,110	**-[A]**
Copper *	T	0.01	0.005	**0.01[E]**
Cyanide - free	T	0.007	<LOR	**0.007**
Iron - dissolved	T	-	0.17	**0.17**
Iron - total	T	-	0.4	**0.4**
Lead	T	0.0034	-	**0.2[D]**
Magnesium	-	-	470	**470**
Manganese	T	1.9	0.1	**1.9**
Mercury	T	0.0006	<LOR	**0.0006**
Molybdenum	T	0.034	0.005	**0.034**
Nickel	T	0.011	0.001	**0.17[D]**
Nitrate (N_NO^{3})*	T	29	0.64	**29[F]**
Nitrite/nitrate (N_NOx)	S	0.2	0.7	**0.7**
Nitrogen total	S	1.2	1.6	**1.6**
Phosphorus total	S	0.065	0.03	**0.065**
Potassium	-	-	10	**-[A]**
Selenium	T	0.011	0.011	**0.011**
Silica (SiO2)	-	-	8.6	**9.3**
Sodium	-	-	998 – 1480	**-[A]**
Strontium	S	-	1.6	**1.6**
Sulfate (S_SO4)	T, S	-	300	**300**
TDS#	S	-	5290 - 9144	**Reference +18%[A]**
Turbidity	S	20	4.6	**20**
TSS	S	-	7	**7**
Uranium	T	0.0005	-	**0.0005**
Vanadium	T	0.006	-	**0.006**
Zinc*	T	0.028	0.02	**0.028[G]**

A – Increase in EC/TDS by 18% of instantaneous upstream reference condition, equivalent of an increase in the median to the 80th percentile background concentration. Ionic composition examined if exceedances in EC/TDS detected.

B - Includes upper 80th %ile year

C – Adjusted using 80th %ile pH and temperature (ANZG 2023a)

D - Hardness-modified trigger value at 775 mg/L as CaCO3

E - DOC adjusted DGV for copper, using 20th %ile DOC (12 mg/L)

F - Based on draft guidance (ANZG 2024a), which incorporates hardness modification. Draft guidance may be subject to change.

G - Based on draft guidance (ANZG 2024b) using pH = 7.6, hardness = 775 mg/L, and DOC = 12 mg/L.

Table 10. Proposed interim site-specific guideline values (SSGV) for the Hotham River applicable to the dry season defined as November, or flow rate < median (2015-2023). Units are mg/L unless specified. (T) – DGV/SSGV for direct toxicity; (S) – DGV/SSGV for stressors. #Further risk assessment based on modelled discharge quality recommended * Interim SSGV based on draft updates to ANZG guidance, therefore changes to this guidance should be incorporated into full operational SSGVs. Refer to footnotes.

Analyte	Toxicant - T Stressor - S	ANZG 95% DGV	80th %ile value	DRY SEASON Interim SSGV
EC (uS/cm)#	S	250	11,678 - 9,130	**Reference +18%[A]**
pH (H+)	S	6.5 - 7.5	7.2 - 7.8	**6.5 - 7.8[B]**
Temperature (°C)	S	-	19.3 - 25.1	**25.1**
Alkalinity		-	214.2	**-**
Hardness		-	-	**-**
Aluminium	T	0.055	0.02	**0.055**
Ammonia*	T	0.26	0.07	**0.26[C]**
Arsenic	T	0.024	0.005	**0.024**
Antimony	T	0.009	<LOR	**0.009**
Boron	T	0.94	0.14	**0.94**
Barium	T	-	0.17	**0.17**
Bicarbonate (HCO3)		-	155	**-[A]**
Cadmium	T	0.002	<LOR	**0.004[D]**
Calcium		-	170	**-[A]**
Carbonate (CO3)		-	<LOR	**N.R**
Cobalt	T	0.0014	<LOR	**0.0014**
Chromium	T	0.001	<LOR	**0.014[D]**
Chloride		-	4,970	**-[A]**
Copper *	T	0.01	0.005	**0.01[E]**
Cyanide - free	T	0.007	-	**0.007**
Iron - dissolved	T	-	0.11	**0.11**
Iron - total	T	-	0.3	**0.3**
Lead	T	0.0034	-	**0.2[D]**
Magnesium		-	440	**440**
Manganese	T	1.9	0.25	**1.9**
Mercury	T	0.0006	<LOR	**0.0006**
Molybdenum	T	0.034	0.005	**0.034**
Nickel	T	0.011	0.001	**0.17[D]**
Nitrate (N_NO_3)*	T	29	0.19	**29[F]**
Nitrite/nitrate (N_NOx)	S	0.2	0.3	**0.3**
Nitrogen total	S	1.2	1	**1.2**
Phosphorus total	S	0.065	0.03	**0.065**
Potassium		-	19	**-[A]**
Selenium	T	0.011	0.006	**0.011**
Silica (SiO2)	S	-	9.5	**9.5**
Sodium		-	2,300	**-[A]**
Strontium	S	-	1.6	**1.6**
Sulfate (S_SO4)	T, S	-	270	**270**
TDS#	S	-	5,960 - 8,970	**Reference +18%**
Turbidity	S	20	5.5	**20**
TSS	S	-	8	**8**
Uranium	T	0.0005	-	**0.0005**
Vanadium	T	0.006	-	**0.006**
Zinc*	T	0.025	0.017	**0.025[G]**

A – Increase in EC/TDS by 18% of instantaneous upstream reference condition, equivalent of an increase in the median to the 80th percentile background concentration. Ionic composition examined if exceedances in EC/TDS detected.

B - Includes upper 80th %ile year

C – Adjusted using 80th %ile pH and temperature (ANZG 2023a)

D - Hardness-modified trigger value at 775 mg/L as CaCO3

E - DOC adjusted DGV for copper, using 20th %ile DOC (12 mg/L)

F - Based on draft guidance (ANZG 2024a), which incorporates hardness modification. Draft guidance may be subject to change.

G - Based on draft guidance (ANZG 2024b) using pH = 7.8, hardness = 775 mg/L, and DOC = 12 mg/L.

Table 11. Proposed interim site-specific guideline values (SSGV) for the Hotham River applicable to year round discharge. Units are mg/L unless specified. (T) – DGV/SSGV for direct toxicity; (S) – DGV/SSGV for stressors. #Further risk assessment based on modelled discharge quality recommended * Interim SSGV based on draft updates to ANZG guidance, therefore changes to this guidance should be incorporated into full operational SSGVs. Refer to footnotes.

Analyte	Toxicant - T Stressor - S	ANZG 95% DGV	YEAR ROUND Interim SSGV	Source
EC (uS/cm)#	S	250	**Reference +18%[A]**	Instantaneous
pH (H+)	S	6.5 - 7.5	**6.5 - 7.8[B]**	Year-SSGV
Temperature (°C)	S	-	**19.3 / 25.1**	Retain seasonal SSGV
Alkalinity		-	**198**	-
Hardness		-	**1,360 - 2,100**	-
Aluminium	T	0.055	**0.055**	DGV
Ammonia*	T	0.26	**0.26[C]**	DRY SSGV
Arsenic	T	0.024	**0.024**	DGV
Antimony	T	0.009	**0.009**	DGV
Boron	T	0.94	**0.94**	DGV
Barium	T	-	**0.14**	WET SSGV
Bicarbonate (HCO3)		-	**-[A]**	-
Cadmium	T	0.002	**0.004[D]**	Interim HMTV
Calcium		-	**-[A]**	-
Carbonate (CO3)		-	**-[A]**	-
Cobalt	T	0.0014	**0.0014**	DGV
Chromium	T	0.001	**0.014[D]**	Interim HMTV
Chloride		-	**-[A]**	-
Copper *	T	0.01	**0.01[E]**	DOC adjusted DGV
Cyanide - free	T	0.007	**0.007**	DGV
Iron - dissolved	T	-	**0.11**	DRY SSGV
Iron - total	T	-	**0.3**	DRY SSGV
Lead	T	0.0034	**0.2[D]**	Interim HMTV
Magnesium		-	**459**	-
Manganese	T	1.9	**1.9**	DGV
Mercury	T	0.0006	**0.006**	DGV
Molybdenum	T	0.034	**0.034**	DGV
Nickel	T	0.011	**0.17[D]**	Interim HMTV
Nitrate (N_NO$_3$)*	T	29	**29[F]**	Hardness adj. DGV
Nitrite/nitrate (N_NOx)	S	0.2	**0.3**	DRY SSGV
Nitrogen total	S	1.2	**1.2**	DGV
Phosphorus total	S	0.065	**0.065**	DGV
Potassium		-	**-[A]**	-
Selenium	T	0.011	**0.011**	DGV
Silica (SiO2)	S	-	**9.5**	Year-SSGV
Sodium		-	**-[A]**	-
Strontium	S	-	**1.6**	Year-SSGV
Sulfate (S_SO4)	T, S	-	**270**	DRY SSGV
TDS#	S	-	**Reference +18%**	Instantaneous
Turbidity	S	20	**20**	DGV
TSS	S	-	**7**	Year-SSGV
Uranium	T	0.0005	**0.0005**	DGV
Vanadium	T	0.006	**0.006**	DGV
Zinc*	T	0.025	**0.025[G]**	Adjusted DGV

A – Increase in EC/TDS by 18% of instantaneous upstream reference condition, equivalent of an increase in the median to the 80^{th} percentile background concentration. Ionic composition examined if exceedances in EC/TDS detected.

B - Includes upper 80^{th} %ile

C – Adjusted using 80^{th} %ile pH and temperature (ANZG 2023a)

D - Hardness-modified trigger value at 775 mg/L as CaCO3

E - DOC adjusted DGV for copper, using 20^{th} %ile DOC (12 mg/L)

F – Based on draft guidance (ANZG 2024a), which incorporates hardness modification. Draft guidance may be subject to change.

G -Based on draft guidance (ANZG 2024b) using pH = 7.8, hardness = 775 mg/L, and DOC = 12 mg/L.

7.0 Hazard analysis

7.1 Post-discharge water quality

7.1.1 Assumptions

As part of the current scope, SLR were asked to undertake a hazard analysis (HA) to assess the potential adverse effects of mine discharge to the Hotham River. This is performed by comparing concentrations of analytes in discharge to the proposed SSGVs. Because the discharge water will be mixed and diluted when discharged, and ANZG (2018) allows for a mixing zone in such situations, the hazard analysis allows for this mixing and dilution of concentrations before assessing against SSGVs. As understood, the planned excess water management strategy is underpinned on discharge of excess water to the Hotham River, at a discharge rate that "sustains a 5:1 dilution factor in the receiving drainage" (Piteau 2024). Nominally, 2,000 kL/hr is given as either an average or a maximum discharge rate (Piteau 2024), based on flows in the Hotham River of 10,000 kL/hr. In order to provide a clear indication of ecological hazard, the following assumptions were made as to the actual volumes of discharge intended, as this affects ultimate dilution:

- The nominal 2,000 kL/hr referenced in Piteau (2024) is a maximum (not an average), and discharge rates will not exceed this level,
- A "5:1 dilution factor" refers to a 20% increase in flow volume above natural catchment discharge, not dilution comprised 1 part mining discharge in 5 parts total (i.e. 2,000 kL discharged into 10,000 kL results in a flow of 12,000 kL and associated dilution)
- Natural catchment discharge is measured from the Hotham Weir gauging station, not Marradong gauging station,
- The dilution factor of 5:1 is nominal, and the final dilution rate will optimise environmental protection alongside operational need,
- That in practice, discharge rates will be adjusted in real time in accordance with natural flow conditions of the Hotham River, to maintain dilution at rates at or above the optimal dilution factor (which may not be 5:1).

7.1.2 Data analysis

Assessment of risk was based on estimated exposure of aquatic fauna to mine discharge under different modelled discharge concentrations and different Hotham flows. This was achieved by combining modelled discharge water quality with Hotham River background levels, and calculation of post-discharge analyte concentrations allowing for relative dilution (i.e. combining volume of discharge with instantaneous flows; thus actual exposure) to identify potential analytes of concern (PAoC).

Hazard assessment included three components of data analysis:

- An initial screening level risk assessment of modelled, undiluted analyte concentrations (using Piteau 2024) against Hotham River proposed interim SSGVs (or DGV) to identify PAoC,
- Calculation of instantaneous discharge rates for application in dilution sensitivity analyses, using the nominal 5:1 dilution rate (Piteau 2024), against two further discharge scenarios (6.8:1 dilution rate, and a 10:1 dilution rate),
- Post-dilution concentrations of PAoC at different discharge rates, to determine actual exposure of aquatic fauna and thereby risk of ecological harm from toxicant or stressor exceedances.

Discharge water quality (modelled 50th and 95th percentile concentrations for each analyte as provided by NBG) for 21 selected analytes were provided by NBG for use in this hazard analysis (Piteau 2024; section 5.2.4). Modelled concentrations were screened against Hotham River SSGVs and compared against 50th and 95th percentile background concentrations for the Hotham River, for the wet and dry season, to identify PAoC.

Discharge volumes were determined by applying nominal 5:1 dilution factor presented in the water management strategy (Piteau 2024) to instantaneous hourly flow rates from the Hotham Weir for use in dilution calculations. To test the sensitivity of total analyte concentration to differences in discharge rate, two further discharge scenarios were considered in calculations of actual exposure, not exceeding 2,000 kL/hr mine water discharge (section 7.1.4). The relative load in mining discharge (i.e. concentration x volume) and Hotham River background loads (i.e. concentration x discharge) were then calculated, from which final estimated concentrations were derived.

To determine actual risk of ecological harm, final analyte concentrations under each discharge scenario were compared to Hotham River SSGVs (section 7.1.5). Four levels of comparison included the 50th and 95th percentile modelled discharge concentrations, against background concentrations for the Hotham River at the 50th and 80th percentile (hereafter HR-50%ile and HR-80%ile). Figure 37 summarises the hazard assessment applied under constraints of the data available.

Analytes were determined as PAoC if, in either season:

- Calculated discharge at the 50th percentile concentration increased final concentrations at HR-80%ile to above SSGV.
- PAoC were considered particularly high risk if discharge at the 50th percentile increased total concentrations to above SSGV at HR-50%ile, indicating sustained exceedances of SSGVs are likely to occur.
- PAoC are also flagged if discharge concentrations at the 95th percentile increased concentrations to above SSGV at HR-50%ile, as there is high likelihood of exceedances under these conditions.
- PAoC were deemed as marginal risk if the discharge concentration of the 95th percentile resulted in exceedances at HR-80%ile. Particularly for toxicants, single records are usually sufficient to infer an exceedance (section 8.0; ANZG 2018), however this indicates exceedances would only occur during periods where background concentrations were also elevated, confirmed in practice with monitoring suitable reference sites.
- Further analysis of identified PAoC considered different discharge scenarios, and where appropriate compared against different levels of species protection (i.e. ANZG 2018 90% and 80% DGVs).

Several analytes were not able to be formally assessed using the methods outlined above, due to lack of information regarding expected discharge concentration. These analytes include those requested by NBG for risk assessment, including cyanide, barium, TDS, and TSS. With exception of cyanide, dilution studies were performed in the reverse, to determine maximum discharge concentrations to maintain HR80%ile below SSGV, acknowledging the limitations of the data at hand.

Regarding cyanide, summary statistics for discharge quality were provided for weak-acid dissociable (WAD) and total cyanide, whereas the ANZG (2018) DGVs relate to free cyanide ($HCN + CN^-$), which is the most readily bioavailable (thus toxic) form. Whilst this precluded direct hazard assessment on free cyanide, the concentration of free cyanide would be equal to or less than the predicted concentrations for WAD or total cyanide. For order of magnitude dilution tests, dilution rates were applied assuming background free cyanide concentrations of

0.004 mg/L (LOR, equivalent to 99% DGV), and the reverse applied to WAD and total cyanide concentrations. Further information regarding the expected concentration of free cyanide in discharge water, as well as background levels for the Hotham River would be required to formally assess risk.

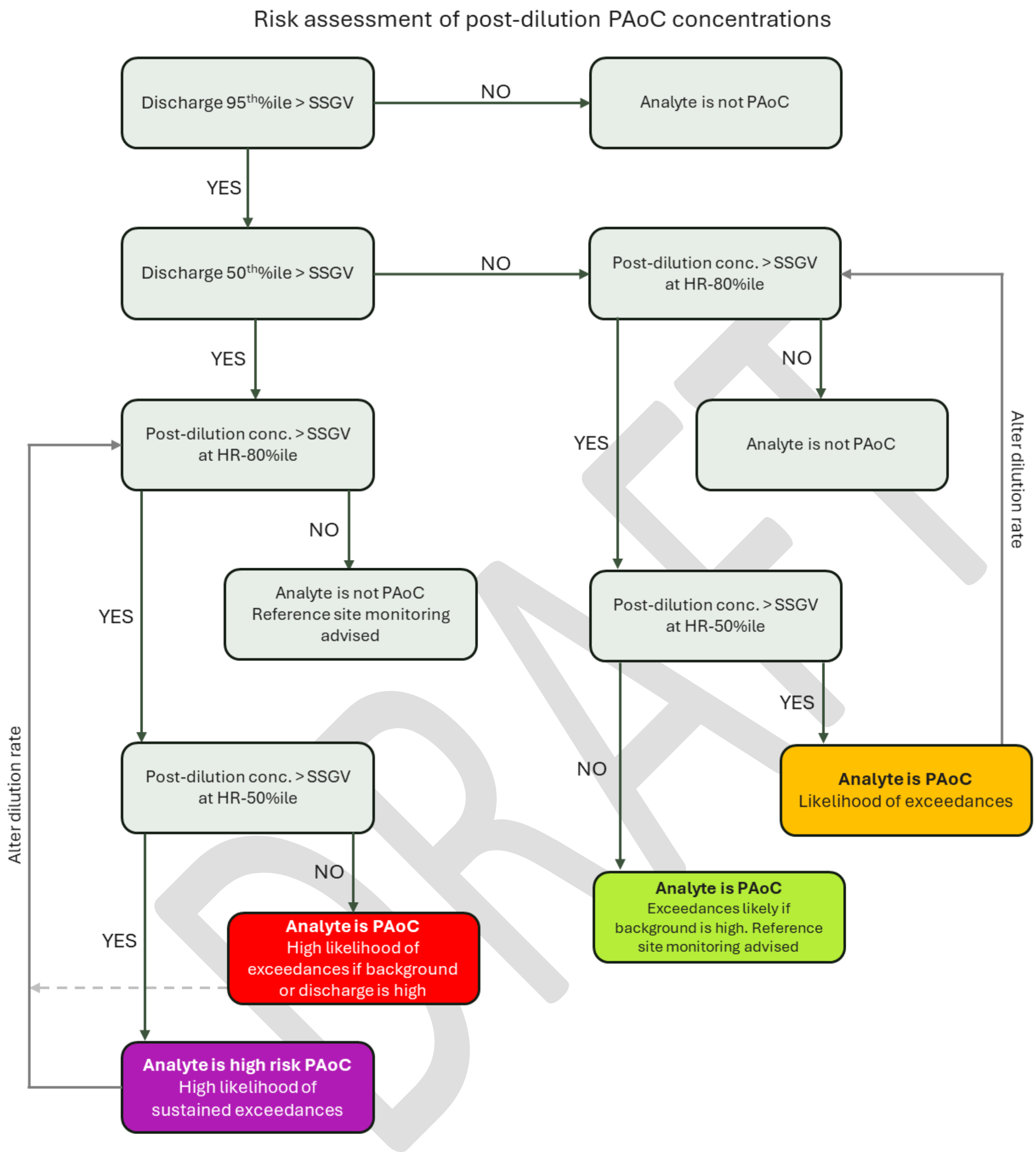

Figure 37. Risk assessment framework applied to post-dilution concentration of potential analytes of concern (PAoC), using modelled discharge concentrations at the 50th and 95th percentile, compared against background concentrations at the Hotham River at the 50th and 80th percentiles (HR-50%ile and HR-80%ile).

7.1.3 Modelled discharge water quality

The majority of the 21 analytes for which modelled discharge concentrations were provided are likely to pose low risk to aquatic ecosystem values of the Hotham River. The majority are expected to be at concentrations below SSGVs, and in some cases below background concentrations (Table 12). Eight analytes exceeded proposed SSGVs. Several analytes were below SSGV at the median, with exceedances at the 95th percentile, for example Mo, N_NO_3 (direct toxicant GV) and S_SO_4. However, several analytes exceeded SSGV at the median modelled discharge concentration, where median background values were well below SSGV, for example Al, Co, marginally Cu, and N_NO_X (eutrophication GV), and were thus considered PAoC. It was not possible to directly assess the potential risk from cyanide, due to provision of modelled total and WAD cyanide concentrations, whereas the ANZG (2018) guidelines refer to free cyanide.

Table 12. Comparison of interim SSGVs against modelled discharge water quality (50th and 95th percentiles, Piteau 2024, Table 5-3), and 'background' wet and dry season concentrations for the Hotham River. Units are in mg/L unless specified. Modelled concentrations above SSGV highlighted green, red highlight indicates concentration > 5x SSGV. Background data above SSGV shaded grey. Dash indicates no data, strikethrough applied to seasonal SSGVs.

Analyte	Interim SSGV	Modelled discharge quality		WET SEASON		DRY SEASON	
		50th %ile	95th %ile	50th %ile	95th %ile	50th %ile	95th %ile
Al	0.055	0.071	0.15	0.008	0.13	<LOR	0.1
As	0.024	0.0027	0.00515	<LOR	0.005	<LOR	0.005
Ca	-	232	321	120	200	141	191.9
Cd	0.0036	0.00033	0.00051	<LOR	<LOR	<LOR	<LOR
Cl	-	2,007	4,746	3450	5400	3515	5497.5
CN_Total	-	0.019	0.078	-	-	-	-
CN_WAD	-	0.0023	0.0089	-	-	-	-
CN_Free	0.007	-	-	<LOR	<LOR	<LOR	<LOR
Co	0.0014	0.014	0.02	<LOR	0.03	<LOR	0.007
Cu	0.01	0.011	0.015	0.002	0.07	0.002	0.03
Fe - dissolved	0.17	0.087	0.12	0.1	0.31	0.07	0.27
K	-	18	52	11	22	12	23
Mg	470	215	454	340	597	350	535
Mo	0.034	0.017	0.035	<LOR	0.005	<LOR	0.0054
Na	-	880	2246.5	1700	2494	1690	2500
NH_3_N (wet)	0.67	0.038	0.52	0.03	0.11		
NH_3_N (dry)	0.26	0.038	0.52			0.04	0.1
NO2_N	-	0.014	0.026	<LOR	0.04	<LOR	0.02
NO3_N	29	1.4	4.1	0.13	1.3	0.09	0.36
NO_X_N (wet)	0.7	1.414	4.126	0.15	1.4		
NO_X_N (dry)	0.3	1.414	4.126			0.09	0.4
pH (H^+)	6.5 - 7.8	6.7	7.1	7.4	6.4 - 7.9	7.6	6.7 - 8.1
Se	0.011	0.0055	0.0072	<LOR	0.03	<LOR	0.03
S_SO4	300	291	437	231	387	207	330
Zn (wet)	0.028	0.029	0.041	<LOR	0.047		
Zn (dry)	0.025	0.029	0.041			<LOR	0.013

7.1.4 Discharge flow rates for sensitivity testing

The relative proportions of analyte loads in both discharge water and natural flow were used to determine actual exposure to aquatic fauna under the following nominal discharge scenarios (Figure 38; Table 13). Flow data from the Hotham Weir (hourly, 01-01-2015 to 06-12-2023) was used to determine discharge volumes at different dilution rates, maintaining 555 L/sec (2,000 kL/hr) as the maximum instantaneous discharge rate. Flow for the Hotham River at Hotham Weir reaches a maximum of 2,777 L/sec (13,644 kL/hr), at which point the data asymptotes, as a limitation of the flow gauging equipment, rather than a natural 'maximum' flow level. However, this does not materially affect the outcomes of this analysis, as this upper flow is used as a threshold, but also, higher flows would result in greater dilution (i.e. lower risk).

Three discharge scenarios were considered for the sensitivity analysis, each assuming discharge between 01 June and 31 October, if flow rate is above the median rate for the Hotham Weir (139 L/sec).

- SC1: Discharge maintains a dilution rate of 5:1 (equivalent to 20% natural catchment flow), to a maximum 2,000 kL/hr (555 L/sec);
- SC2: Maximum instantaneous discharge rate of 2,000 kL/hr at peak flows in the Hotham River (13,644 kL/hr), but maintain an equal discharge ratio throughout the wet season (6.8:1, or 14.6% of natural catchment flow).
- SC3: Maintain a 10:1 dilution rate, to a maximum of 1000 kL/hr, as a conservative scenario.

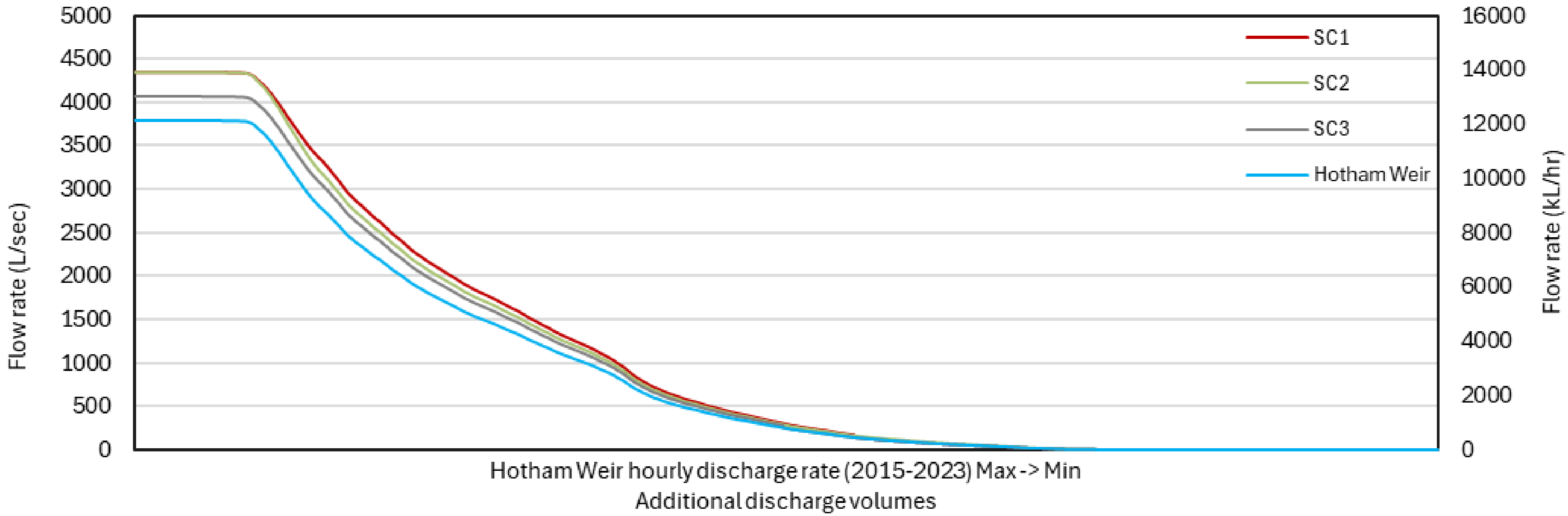

Figure 38. Hourly flow rate (L/sec) recorded from Hotham Weir gauging station (01-01-2015 to 06-12-2023), arranged in descending order (blue), with estimated additional flow volumes from discharge under three scenarios considered for this hazard analysis.

Table 13. Comparison of flow rates used in hazard analysis under discharge additional to Hotham River volumes. Values given in kL/hr and L/sec. *Assumed maximum discharge in practice is 2,000 kL/hr (555 L/sec).

Flow rates		Median flow	Nominal flow	Maximum flow
Hotham Weir	kL/hr	500	10000	13644
	L/sec	139	2778	3790
Discharge (additional volumes)				
5:1 flow	kL/hr	100	2000	2729*
	L/sec	27.6	556	758
6.8:1 flow	kL/hr	73	1460	2000
	L/sec	20.3	406	556
10:1 flow	kL/hr	50	1000	1364
	L/sec	13.9	278	379

7.1.5 Post-dilution concentrations of PAoC

Comparing post-dilution concentrations at a discharge dilution rate of 5:1 at both HR50% and HR80%, aluminium, ammonia, copper, molybdenum, zinc and nitrate (as N_NO_3, direct toxicant, not a stressor) were below SSGV, inferring low risk to the receiving environment under a 5:1 dilution rate or lower (Table 14).

Addition of discharge water was estimated to increase background concentrations to above SSGV for three PAoCs. The 95th percentile discharge concentrations of sulfate (S_SO_4) was estimated to cause modest increases in total concentration at HR80%. Discharges of water with cobalt at the 50th percentile increased total concentration to above the DGV at both HR50% and HR80%, by more than a factor of two, indicating high likelihood of exceedances under a 5:1 discharge dilution rate. As a stressor, 50th percentile nitrite/nitrate (N_NO_X) concentrations in discharge is expected to increase background concentrations at HR80% in the wet season, and both HR50% and HR80% in the dry season. This indicates high likelihood of adverse effects of nutrient enrichment as a stressor on the aquatic environment, especially if discharge were to occur in the dry season.

Further sensitivity analysis is presented below for copper, cobalt, N_NO_X and sulfate. Several analytes of interest were precluded from this direct risk assessment, due to lack of background data for the Hotham, unknown predicted discharge concentrations (or modelled discharge not provided), or both. These included cyanide, TSS, EC and TDS.

Table 14. Estimated post-dilution concentrations of selected analytes for the Hotham River, at median and 80th%ile background concentrations, and median and 95th%ile modelled discharge concentrations. Assumed discharge at 5:1 additional volume (scenario 1). Green highlighted values exceed interim SSGVs. Risk is determined as low if 95th%ile discharge does not increase 80th%ile to SSGV concentration; further examination of identified PAoC is advised. Note: both HR50% and HR80% for Co are below LOR. W = wet and D = dry season.

Analyte (Wet, Dry)	DGV/	Hotham %ile	50th %ile Hotham		80th %ile Hotham		Risk
	Interim SSGV	Discharge %ile	50th %ile	95th %ile	50th %ile	95th %ile	
Al	0.005		0.018	0.032	0.036	0.05	Low risk
Cu	0.01		0.002	0.003	0.006	0.007	Low risk
Co	0.0014				0.003	0.004	PAoC
Mo	0.034		0.0029	0.0039	0.0032	0.0042	Low risk
N_NH_3 D	0.26		0.03	0.11	0.06	0.14	Low risk
N-NO_3 W	29		0.34	0.79	0.71	0.95	Low risk*
N_NO_3 D	29		0.31	0.76	0.39	0.84	Low risk*
N_NO_x W	0.7		0.36	0.82	0.81	1.27	PAoC
N_NO_x D	0.3		0.31	0.76	0.48	0.94	PAoC
S_SO_4	300		241	265	298.5	323	Marginal
Zn W	0.028				0.011	0.013	Low risk
Zn D	0.025				0.011	0.013	Low risk

*The SSGV for N_NO_3 is intended to be protective from impacts of direct toxicity, it does not indicate protection against detrimental impacts of eutrophication.

Copper

Residual concentrations of Cu do not exceed the interim SSGVs, using the adjusted draft DVGs at 12 mg/L DOC providing an adjusted SSGV for Cu of 0.01 mg/L. Therefore, it is likely that predicted Cu concentrations present a low risk of direct toxicity to aquatic fauna, however, the draft DGVs are yet to be formalised and may be subject to change. Post-dilution concentrations remained well below the proposed interim SSGV (based on ANZG 2023b; Figure 38).

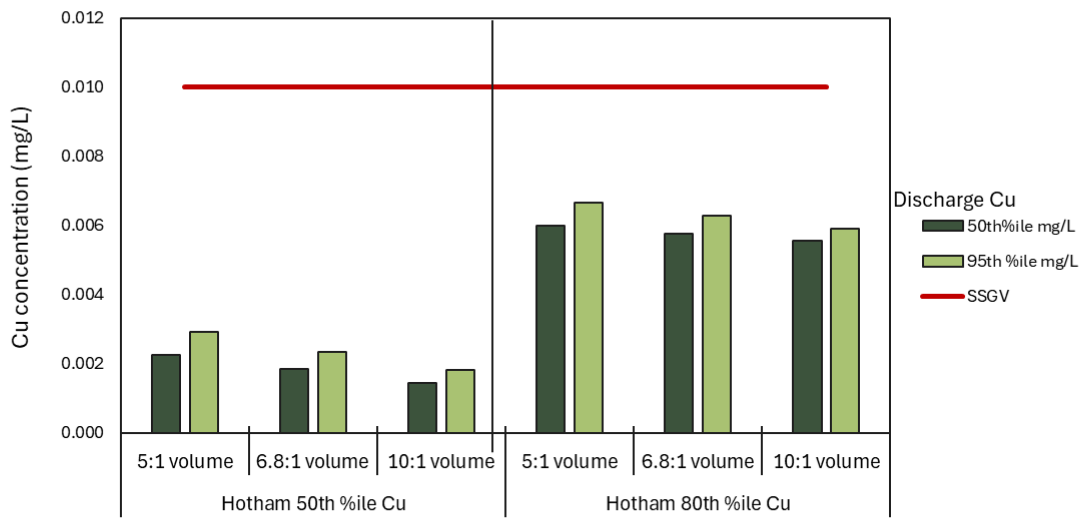

Figure 39. Estimated post-dilution concentration of Cu (mg/L) under different discharge scenarios (5:1, 6.8:1 and 10:1 additional flow volume) at background 50th%ile and 80th%ile concentrations for the Hotham River, and modelled 50th percentile and 95th percentile discharge concentrations. The Hotham River DOC adjusted SSGV (0.01 mg/L) shown as red line.

Cobalt

There is high likelihood that discharge of Co will increase total concentrations significantly against background concentrations of the Hotham River. Naturally, background concentrations are low with HR80% below the laboratory LOR (0.001 mg/L), noting exceedances in the dataset are rare (and potentially spurious).

- Discharge 50th percentile Co concentration are modelled to be 10x SSGV, and at a 5:1 dilution rate remains above 2x the SSGV. Post-dilution concentrations were estimated to be well above SSGV under all three dilution scenarios (Figure 40).
- To maintain a post-dilution 80th percentile Co concentration below SSGV (0.0014 mg/L), a maximum dilution rate of 13.5:1 at modelled 50th percentile discharge Co would be required (or 714.3 kL/hr discharge to 10,000 kL/hr natural catchment flow). At the modelled discharge 95th percentile (0.02 mg//L) a maximum of 21:1 would be required (or 483.9 kL/hr discharge in 10,000 kL/hr catchment flow) (Figure 41).

Although occasional high values of Co were recorded for the Hotham River, particularly from the monitoring site at Marradong Bridge, the 80th percentile was below LOR, indicating background concentrations are low. Whilst it is worth noting the ANZG (2018) DGV is considered a low reliability guideline, due to inconsistencies in the available toxicity data, it is based on direct toxicity assessment and thus Co at this level would be expected to have toxic effects. Published literature using the SSD method for Co are sparse, however Stubblefield et al., (2020) found a median hazardous concentration for 5% of organisms of 0.0018 mg/L, approximating with the current Australian guideline of 0.0014 mg/L. Without specific toxicity testing on local fauna of the Hotham River, it should be assumed that exceedances of the guideline value would result in the loss of species from the river.

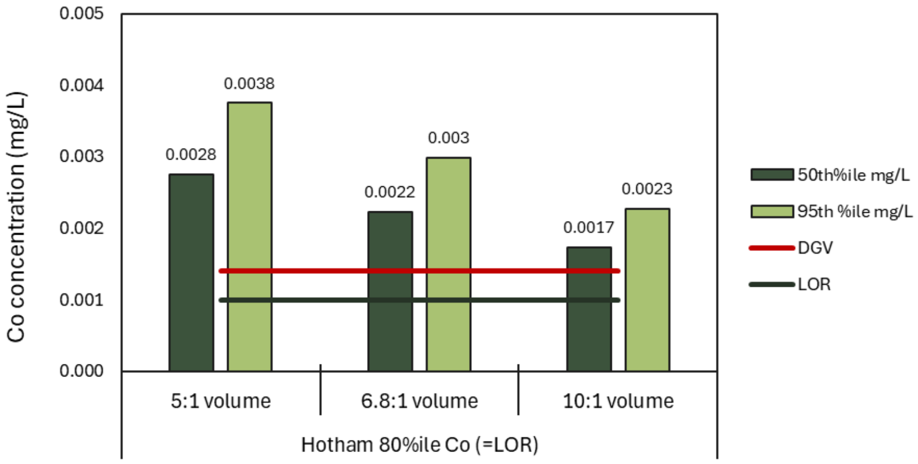

Figure 40. Estimated post-dilution concentration of Co (mg/L) under different discharge scenarios (5:1, 6.8:1 and 10:1 additional flow volume) at background 80th%ile concentration for the Hotham River (less than LOR, 0.001 mg/L), against modelled 50th percentile and 95th percentile discharge concentrations. The ANZG DGV (0.0014 mg/L) shown as red line; LOR shown as black line.

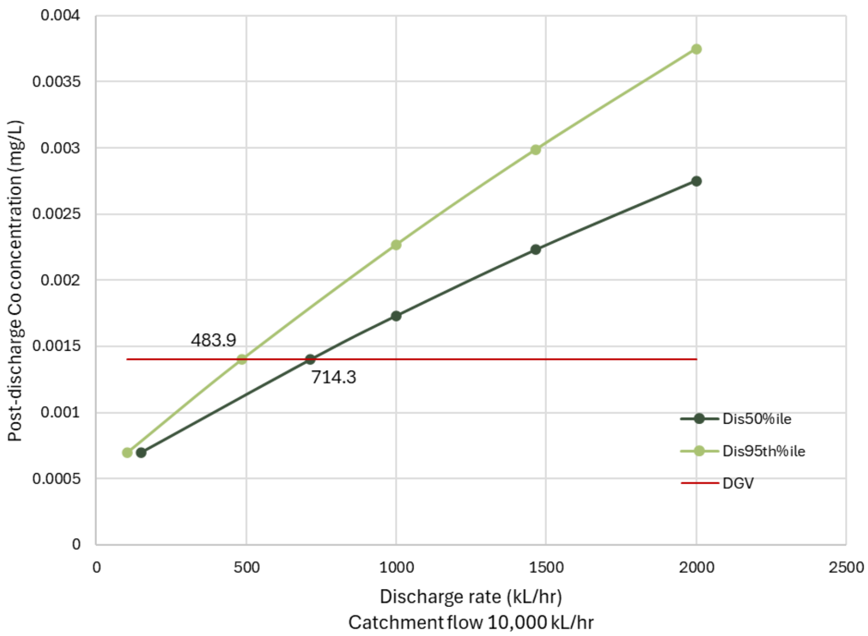

Figure 41. Post-dilution cobalt concentration (mg/L) under increasing discharge rates (kL/hr) at 50th and 95th percentile discharge Co concentrations. Discharge volumes calculated on 10,000 kL/hr catchment flow in the Hotham River. The ANZG (2018) DGV is shown in red, data callouts indicate discharge rate (kL/hr) that results in a final concentration equal to the SSGV/DGV.

Nitrate/ nitrite N_NO_X

Although nitrogen nutrients are already elevated in the Hotham River, there is high likelihood of further nitrogen enrichment in downstream environments of the Hotham River due to elevated N_NO_X is discharge water, relative to background concentrations. As N_NO_X is a stressor, not a direct toxicant, exceedances are deemed as likely to occur if discharge at any concentration increases post-dilution concentrations at HR-50% to equal or above SSGV (equivalent to HR-80%).

- In the wet season, median and 95th percentile discharge N_NO_X concentrations are expected to exceed SSGV at the 5:1 discharge scenario at HR80% (Figure 42). If actual discharge concentrations are closer to the modelled 95th percentile, increases at HR-50%ile (and thus stressor exceedances) may be likely to occur at a 5:1 or 6.8:1 dilution rate.
- In the dry season, median and 95th percentile discharge N_NO_X concentrations are expected to exceed the SSGV at HR50% and HR80% (Figure 43). This suggests that discharge over the dry season would result in near constant exceedances of the SSGV, at each dilution scenario considered.
- There is some benefit of reducing nominal discharge rates to 6.8:1 or 10:1 in the wet season, as there is reduced likelihood of exceedances at HR50%, if actual discharge concentrations are near the modelled 95th percentile. Exceedances would likely remain frequent in the dry season under all scenarios.
- Because the SSGV = HR-80%, any discharge at the modelled 50th percentile (1.414 mg/L) or 95th percentile (4.126 mg/L) would result in a total concentration > SSGV in either season at 80th percentile background concentrations. This highlights the importance of regular monitoring at suitable upstream reference sites throughout the duration of discharge operations.
- Even if discharge is limited to the wet season, there is still risk of eutrophication during summer, as the capacity of downstream environments (e.g. pools) to assimilate N_NO_X is unknown. However, if wet season 80th percentile values remain at or below 0.7 mg/L post-dilution then no increase in risk over the dry season is implied.

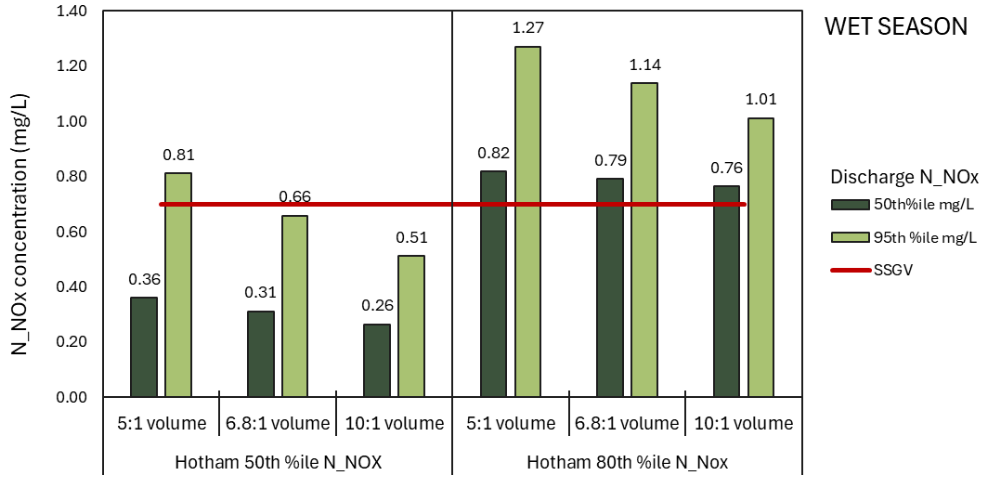

Figure 42. Estimated post-dilution concentration of N_NO_X (mg/L) in the wet season, under different discharge scenarios (5:1, 6.8:1 and 10:1 additional flow volume) at background 50th%ile and 80th%ile concentrations for the Hotham River, and modelled 50th percentile and 95th percentile discharge concentrations. The Hotham River wet season SSGV (0.7 mg/L) shown as red line.

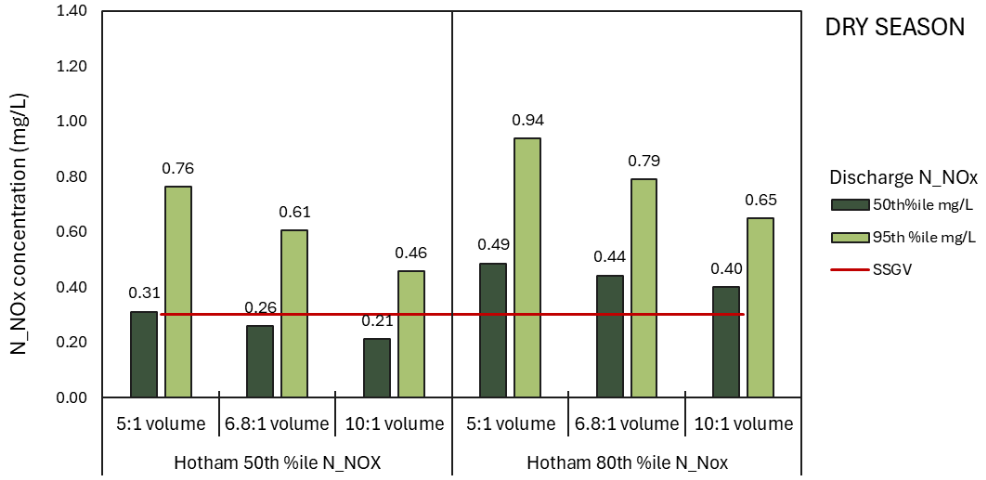

Figure 43. Estimated post-dilution concentration of N_NO_X (mg/L) in the dry season, under different discharge scenarios (5:1, 6.8:1 and 10:1 additional flow volume) at background 50th%ile and 80th%ile concentrations for the Hotham River, and modelled 50th percentile and 95th percentile discharge concentrations. The Hotham River dry season SSGV (0.3 mg/L) shown as red line.

<u>Sulfate</u>

The modelled discharge concentrations of sulfate (S_SO_4) are generally below SSGV, with potential for exceedances at the 95th percentile (437 mg/L; Table 11). Post-dilution, discharge is not anticipated to increase S_SO4 above wet seasons SSGV at HR-50%ile, however discharge with concentrations above median would result in post-dilution concentrations above SSGV at HR-80%ile (Figure 44). The magnitude of exceedance is predicted to be slightly higher in the dry season, due to background concentrations being lower (Figure 45).

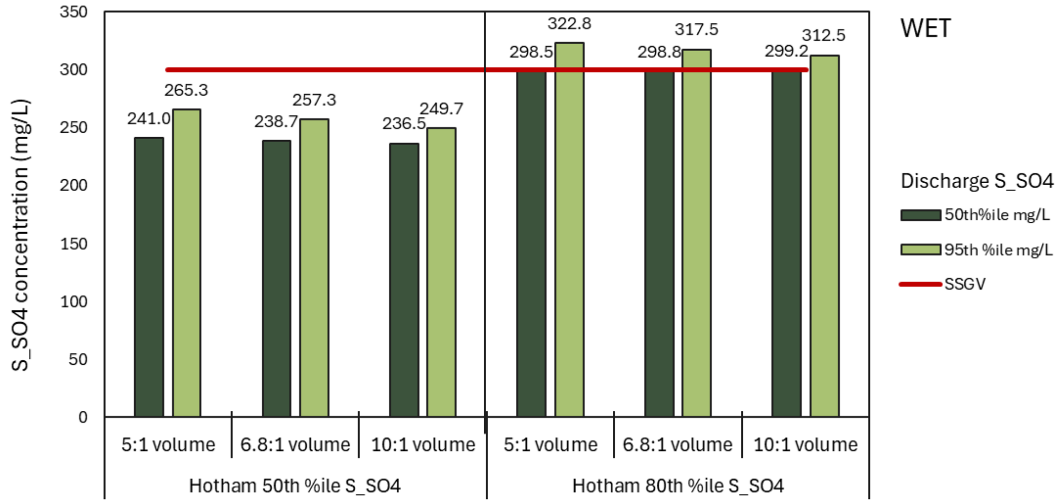

Figure 44. Estimated post-dilution concentration of S_SO4 (mg/L), under different discharge scenarios (5:1, 6.8:1 and 10:1 additional flow volume) at background 50th%ile and 80th%ile concentrations for the Hotham River, and modelled 50th percentile and 95th percentile discharge concentrations. The Hotham River SSGV (300 mg/L) for the wet season shown as red line.

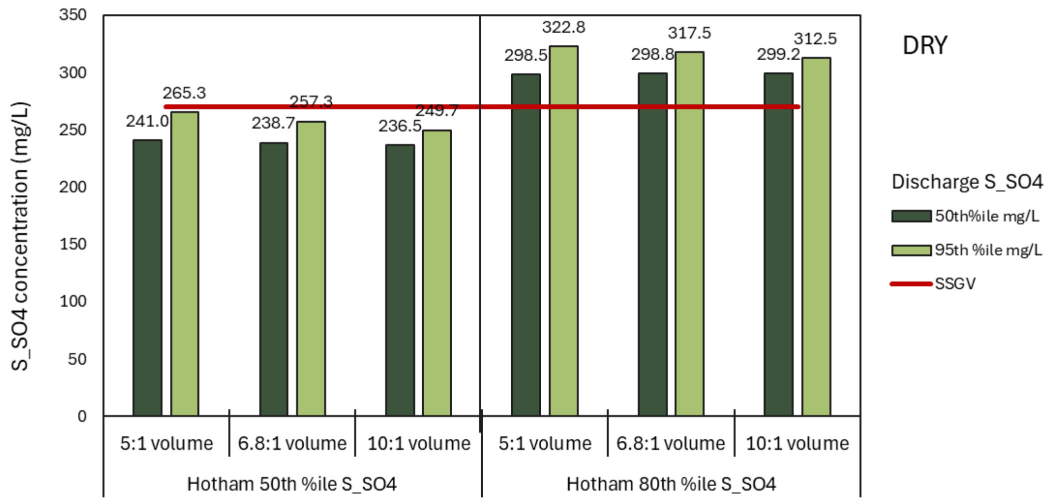

Figure 45. Estimated post-dilution concentration of S_SO4 (mg/L), under different discharge scenarios (5:1, 6.8:1 and 10:1 additional flow volume) at background 50th%ile and 80th%ile concentrations for the Hotham River, and modelled 50th percentile and 95th percentile discharge concentrations. The Hotham River SSGV for the dry season (270 mg/L) shown as red line.

7.1.6 Analytes without comparable discharge data

7.1.6.1 Free cyanide and cyanide complexes

Cyanide is measured as either free, weak-acid dissociable (WAD) or total cyanide. Water Quality Australia provide guidelines for free cyanide concentrations, as this is the form that is most readily bioavailable, and thus most toxic to aquatic fauna (ANZG 2018). Modelled discharge concentrations of total cyanide and WAD cyanide were provided (50th and 95th percentiles; Piteau 2024), however these are not directly comparable to the ANZG DGVs for free cyanide, as these measures also include complexed cyanide. Nevertheless, available measurements of free, WAD and total cyanide for the Hotham River were <LOR, implying very low naturally occurring concentrations, noting that some data points had LORs much higher than ANZG DGVs (0.01 mg/L).

In lieu of direct comparison to DGVs, the below calculations consider discharge concentrations that would present a high risk of exceeding post-dilution concentrations above species protection limits. This does not infer 'safe' operating limits to free cyanide concentrations, which would require further investigation using discharge quality data, and updated data for the Hotham River.

- The LOR for free cyanide is 0.004 mg/L, which is equal to the 99% species protection DGV. Thus, 0.004 mg/L is conservatively assumed as background concentrations for the Hotham River.
- Assuming a background concentration of 0.004 mg/L free cyanide, a final concentration of 0.007 mg/L at a 5:1 dilution rate is estimated to result from discharge at concentrations at or below 0.022 mg/L (Figure 46; Table 15).
- Modelled discharge concentrations of WAD cyanide at the 95th percentile (0.0089 mg/L) indicate free cyanide is unlikely to exceed the 95% species protection DGV at a discharge rate of 5:1 additional flow (Figure 46). Modelled permeate chemistry is much higher (WAD cyanide 0.032 mg/L), thus this comparison assumes treatment to be as effective as indicated by water balance models (Piteau 2024). At a maximum of 0.032 mg/L, exceedances would be expected under 5:1 and 6.8:1 dilution scenarios (Table 15).
- Because the Hotham River appears to be little exposed to cyanide, an increase in free cyanide concentrations to equal the 95% species protection DGV, or above, would be assumed to have negative impacts, acknowledging local species sensitivity data are not available. The capacity of the Hotham River and downstream systems (e.g. the Murray River) to assimilate cyanide is not known, nor is the future spatial extent of detectible cyanide originating from NBG discharge. This analysis does not take into consideration factors such as local community responses to perceived impacts of discharges containing cyanide.

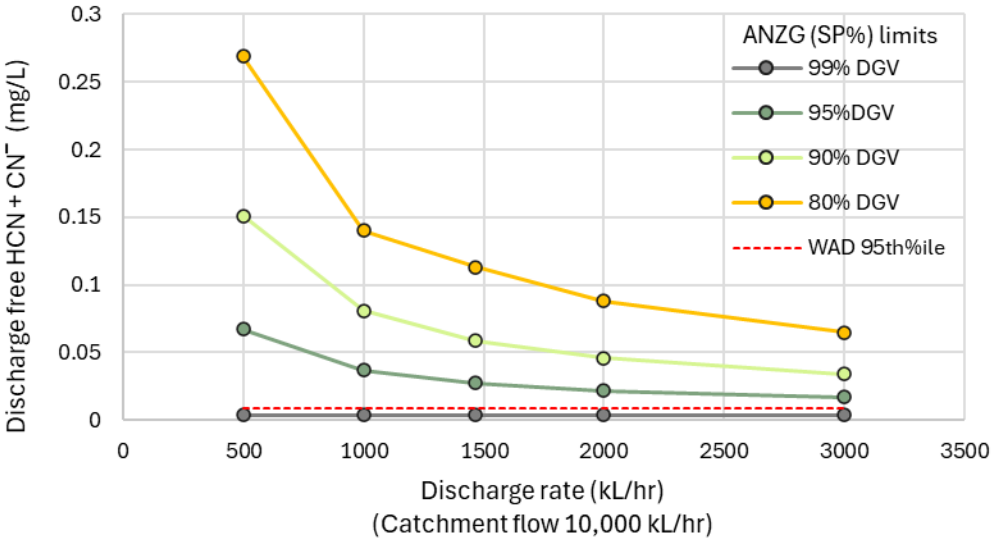

Figure 46. Calculated discharge concentrations of free cyanide (CN^- + HCN) that would present elevated risk of exceedances of ANZG (2018) species protection (SP%) limits 99%, 95%, 90% and 80%. Background concentrations of the Hotham River are assumed to be 0.004 mg/L (<LOR, equal to 99% LOSP). Dashed line presents modelled weak-acid dissociable (WAD) cyanide at the 95th percentile concentration (0.0089 mg/L). Calculated values presented in Table 15.

Table 15. Calculated discharge concentrations of free cyanide (CN^- + HCN) presenting elevated risk of exceedances of ANZG (2018) DGVs, under different discharge dilution scenarios (dis. rate as kL/hr and L/sec). Background concentrations of the Hotham River are assumed to be 0.004 mg/L (<LOR, equal to 99% SP). SP% limits = 0.004 mg/L (99%); 0.007 mg/L (95%); 0.011 mg/L (90%); 0.018 mg/L (80%).

Catchment flow 10,000 kL/hr (*2,778 L/sec*)			Discharge concentration (mg/L) (Post-dilution => ANZG DGV)			
Dilution	Discharge rate					
	kL/hr	*L/sec*	*99% SP*	*95% SP*	*90% SP*	*80% SP*
20:1	500	*138.9*	0.004	0.067	0.151	0.269
10:1	1,000	*277.8*	0.004	0.037	0.081	0.14
6.8:1	1,466	*407.2*	0.004	0.027	0.059	0.113
5:1	2,000	*555.6*	0.004	0.022	0.046	0.088
3.33:1	3,000	*833.4*	0.004	0.017	0.034	0.065

7.1.6.2 EC, TDS and ionic composition

Modelled discharge concentration of EC and TDS were not provided, therefore direct comparison to background for the Hotham River were not possible. However, modelled 50th and 95th percentiles for the anions Ca, Mg, Na and K, and for the cations Cl, S_SO4 were provided.

Na and Cl are the dominant cation/anion for both the Hotham River and modelled discharge, suggesting discharge is unlikely to substantively alter the ionic composition of waters below the discharge point. Discharge concentrations are predicted to have a lower median value for both ions, and 95th percentiles slightly above (Na) or below (Cl) that of background concentrations (Figure 47). Among less dominant ions, Mg is predicted to be within the background ranges of the Hotham River, however, K, Ca, and S_SO4 are all predicted to be higher, with median discharge values exceeding the 95th percentile background values (Figure 47).

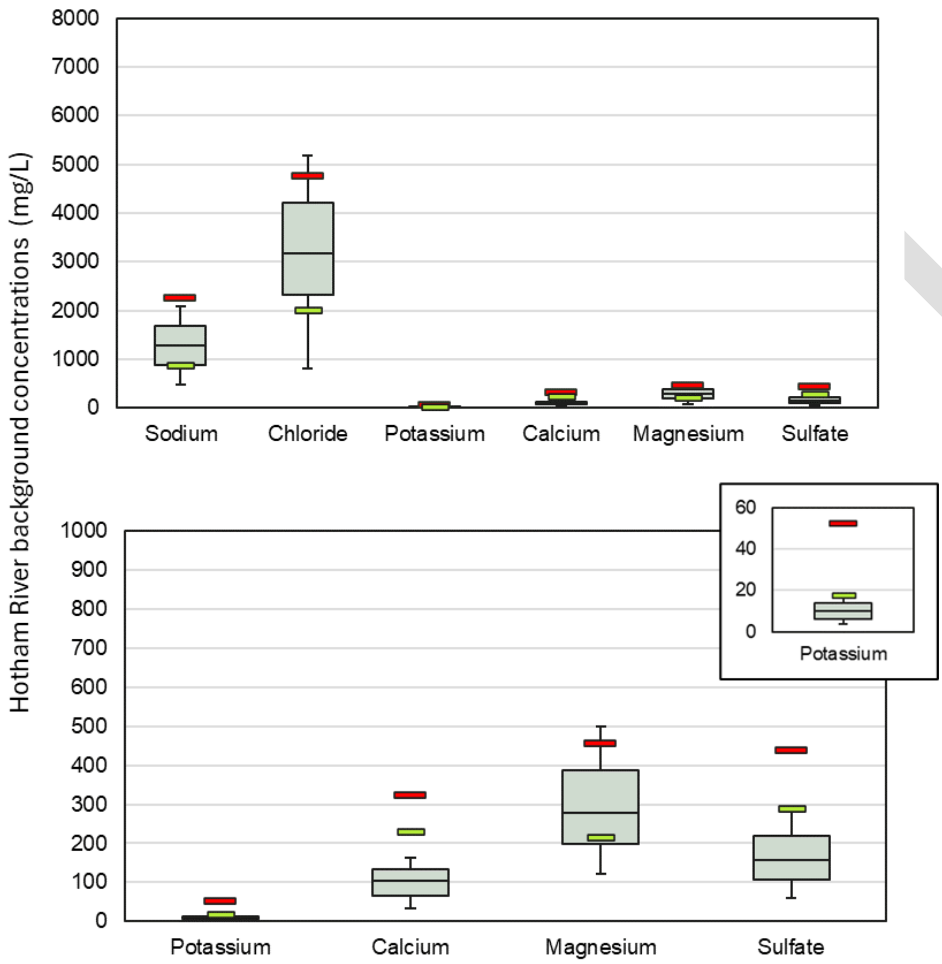

Figure 47. Percentile ranges of dominant ions for the Hotham River (mg/L), with modelled discharge concentrations shown: 50th %ile (green dash) and 95th %ile (red dash).

The Hotham River is widely affected by secondary salinisation, as a legacy effect of agricultural clearing in the catchment. Nevertheless, discharge of mine water with significantly higher TDS/EC to background conditions will affect the receiving environment. If TDS is markedly higher, then aquatic biota will experience increased osmotic stress, which may exceed species tolerances. Conversely, if discharge water is markedly fresher than riverine condition, then potential for incomplete mixing and density stratification could occur, particularly during low flow periods/dry season.

As an indicator, the sum of median concentrations of anions and cations (Ca, Mg, Na, K, Cl, S_SO4) were compared between the Hotham River and modelled discharge, noting this is strictly not a substitute for direct comparison of TDS and salinity. However, the median concentrations of these ions suggests TDS may be lower in discharge than background concentrations (discharge = 3,643 mg/L; Hotham = 5,866 mg/L; ~37% difference). The difference during the dry season is further pronounced (Hotham = 6,773 mg/L; ~46% difference). There may be potential for a density differential between dry season Hotham flows and discharge, that may directly cause ecological harm due to density stratification and deoxygenation of refuge pools if there is discharge of fresher water into pools in summer. During the wet season, it is probable that the turbulent action of winter flows would assist in mixing, likely precluding risk of density stratification. It is highly recommended that modelled discharge TDS and EC data be provided for further risk assessment against background concentrations.

7.2 EWRs

Under the current abstraction licence, NBG abstract water from the Hotham River for use in mine processes. Ecological monitoring is undertaken biennially to determine whether changes to the duration of time flow thresholds are above established ecological water requirements (EWRs), with a particular focus on the low flow thresholds relating to fish passage (WRM 2015; SLR 2024). Increased flow to the Hotham River from discharge would present the reverse scenario, where amount of time flow thresholds are above EWRs, particularly those related to channel morphology and erosive potential (i.e. active channel and top of bank flows) are increased (Figure 48).

Using 2023 flow data to illustrate, under a 5:1 dilution scenario, with a maximum discharge of 2,000kL/hr, results in modest increases to active channel and top of bank flows (5.5% and 3.9%, respectively; Table 16). Should discharge be allowed to increase to 3,000kL/hr during peak flows, top of bank flows would be expected to occur 16% of the year, and active channel over 30% of the year (6% and 10% increases, respectively). This, notwithstanding greater flow totals generally, may be expected to increase erosion in the downstream reaches of the Hotham River, altering channel morphology and sedimentation dynamics. Risk may be assumed to be low under conservative discharge scenarios (5:1 dilution or less, to a maximum of 2,000 kL/hr).

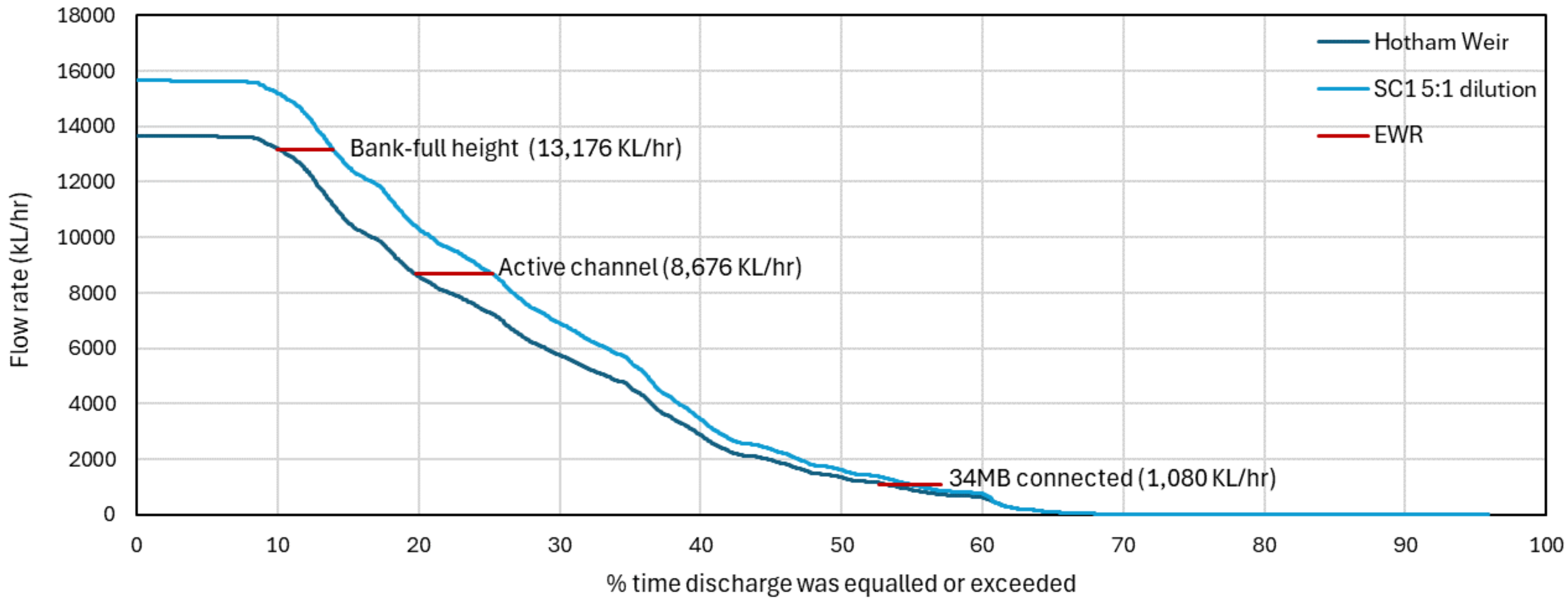

Figure 48. Amount of time EWR was equalled or exceeded in 2023 (kL/hr vs. %time), with indicative change in amount of time under a 5:1 dilution scenario (SC1).

Table 16. Amount of time five EWR flow criteria were met in 2023, and indicative increases (%time) under a 5:1 dilution scenario (SC1).

		% time flow equals or exceeds		
EWR flow criteria	**KL/hr**	**Natural flow regime**	**SC1**	**%Difference**
Small fish passage	*288*	62.0	62.0	0
Large fish passage	*900*	55.0	55.0	0
34MB connectivity	*1080*	53.0	55.0	+2
Active channel	*8676*	19.8	25.3	+5.5
Top of bank	*13176*	10.0	13.9	+3.9

8.0 Testing against SSGVs under discharge operations

The interim SSGVs proposed are intended to detect exceedances of water quality toxicant and stressors in future monitoring data for the Hotham River. The definition of "exceedance" adopted is in line with guidance from the ANZG[10], which recommend data analysis approaches for post-impact monitoring of toxicants and stressors. The hazard assessment considers whether modelled discharge is likely to increase post-dilution concentrations of the Hotham River above background conditions, to determine if the below conditions for toxicant or stressor exceedances are likely to be met.

- For toxicants, the 95th percentile of monitoring data is compared with the DGV/SSGV, that is, an exceedance is deemed to have occurred if the 95th percentile of the monitoring data exceeds the guideline value. Because the toxicant DGVs are based on actual biological effects data, and the proportion of the values required to be less that the guideline is very high (95%), in most situations a single observation greater than the guideline would be legitimate grounds for determining that an exceedance has occurred.
- For stressors, by default, the median values of the monitoring data should be below the 80th percentile value of suitable reference sites (or above 20th percentile where low values are a problem; e.g. dissolved oxygen or pH). Exceedances are determined to have occurred if the median monitoring data is greater than the 80th percentile reference data (or less than the 20th percentile, where applicable) (ANZG 2018). For most stressors, occasional spot measurements above the DGV are not considered an exceedance, rather a consistent change in monitoring data compared to baseline or reference condition needs to be demonstrated to have occurred.

9.0 Summary

With the exception of high TDS and EC, waters of the Hotham River were generally of good quality, and in most cases background analyte concentrations were below the ANZG (2018) DGVs for 95% species protection, applicable to moderately disturbed systems. Salinisation of the Hotham River is a long standing legacy impact of agricultural clearing; nevertheless, the river supports a diverse assemblage of tolerant fauna. Key findings of this desktop assessment of fauna and water quality are summarised below, and key PAoC are discussed in section 9.2.

9.1 Aquatic fauna

Invertebrates

The known invertebrate fauna of the Hotham River is dominated by salt-tolerant species, including a diversity of Diptera, Coleoptera and crustaceans. Whilst no formally listed species appear in taxa records, the assemblage includes a suite of SWWA endemic invertebrates, including dragonflies and caddisflies. Altogether, no listed species were recorded, the presence of several regionally endemic taxa does afford this macroinvertebrate assemblage conservation significance. Furthermore, few invertebrate taxa actually appear on formal conservation lists, despite known regional declines in many SWWA endemic species (Sutcliff 2003; Pennifold 2018). Because little is known about the habitat preferences, life histories and water quality tolerances of south-west endemic macroinvertebrates (e.g. Davis et al., 2014; Pennifold 2018), predicting direct effects of altered water quality on Hotham River aquatic biota is not possible, however, in general terms, exceedances of accepted DGVs for toxicants

[10] https://www.waterquality.gov.au/anz-guidelines/monitoring/data-analysis/derivation-assessment

will result in loss of species, and exceedances of thresholds for stressors, will result in reduced ecosystem health, with also a likelihood of reduced diversity. Although the fauna is displaying modification and adaption to increased salinity, further increases would cause additional reductions in diversity.

Fish and crayfish

All the native freshwater fish and crayfish in SWWA are endemic to the region, including the four native fish species frequently recorded from the Hotham: the western minnow (*Galaxias occidentalis*), western pygmy perch (*Nannoperca vittata*), nightfish (*Bostockia porosa*), freshwater cobbler (*Tandanus bostocki*), and three crayfish species gilgie (*Cherax quinquecarinatus*), koonac (*Cherax pressii*) and smooth marron (*Cherax cainii*). In addition, the Swan River goby (*Pseudogobius olorum*) and the Southwestern goby (*Afurcagobius suppositus*) are typically estuarine, but are halotolerant and can be found considerable distances inland where reivers are affected by secondary salinisation. Anecdotally, increases in their abundances have been detected in the Hotham in years where rainfall is below average, suggesting reduced flushing and flow velocities may favour these species locally (SLR 2024).

Distributions of freshwater fish in the Hotham River have been monitored biennially by WRM (now SLR) in accordance with conditions for NBGs abstraction licence. Generally, population demographics of western minnow, nightfish and Swan River goby are shaped in accordance with inter-annual variability in rainfall, which controls variation in flow regime and connectivity of tributary creeks (SLR 2024). Very little is known about the population size or demography of cobbler in the Hotham, as catches have been low in number and temporally sporadic. It is likely that cobbler reside in deeper pools, and undertake movements under specific migratory cues (Beatty et al., 2010; Beesley et al., 2019). There is a significant knowledge gap regarding cobbler populations in the Hotham River, therefore the potential for deleterious impacts on populations are difficult to predict. Should there be stable populations in the Hotham, it can be inferred with confidence that this species (and others) are residing in deeper permanent refuge pools over summer, when flows are low and reaches undergo disconnection. During these times populations would be vulnerable to deterioration in water quality of refuge pools. Should a decline in water quality occur and result in local extirpation, then reduction in supply of colonists to upstream and downstream reaches upon the resumption of flows would also occur.

Rakali

The rakali (or water rat; *Hydromys chrysogaster*; P4) has been confirmed as present at the Hotham River (SLR 2024) and it was also recorded at 34 Mile Brook. Anecdotal evidence and local sightings suggests there may be rakali present at, or nearby, the Lion's Weir. Rakali occupy a wide variety of freshwater habitats, from inland waterways to lakes, swamps, and farm dams, and are thought to have home ranges of up to 4km (Williams et al., 2014). Key threats to their populations include altered flow regimes and secondary salinisation (Lee 1995), which has related effects on prey animals, and provision of stable habitat for the construction of burrows. Therefore, there is a requirement for stable stage heights that inundate banks, tree roots and large woody debris, without erosive flows (WRM 2018). Using Hotham Weir flow data from 2023 data as an example, modest increases in erosive flows (top of bank and active channel flows) were predicted under a maximum discharge scenario of 2,000 kL/hr, however should discharge exceed this level, then increased erosion may become a risk to rakali populations. Although secondary salinisation is listed as a threat, the threshold levels for adverse effects on rakali are not quantified, however, their presence in the system currently suggest the threshold is higher than current salinity in the Hotham. An increase in salinity due to discharge could adversely affect this listed species.

South-western snake-necked turtle

One species of freshwater turtle is known from the study area; the southwestern snake-necked turtle, or long-necked turtle*, Chelodina oblonga* (Shea et al. 2020). This species is endemic to southwest Western Australia. It is not currently listed under the EPBC Act or BC Act, but appears on the IUCN Redlist of Threatened Species as Near Threatened (IUCN 2024). They are frequently encountered at the Hotham River (SLR 2024), including sub-adult individuals, suggesting successful recruitment in the local population.

Mussels

Mussel records in the Hotham River include the estuarine species, *Fluviolanatus subtorta*, which was been found during studies in 2019 and 2020 (WRM 2019; WRM 2020). This species is not currently listed as conservation significant. It has a relatively widespread distribution in estuarine coastal environments, and the range expansion inland to the Hotham River is understood to be an extensive migration upstream from known populations near Mandurah (ALA 2024).

The freshwater mussel *Westralunio carteri* is endemic to SWWA and is listed as Vulnerable under the EPBC Act (1999), BC Act (2016) and IUCN Redlist of Threatened Species (2024). There is anecdotal evidence of local *W. carteri* occurring in the Hotham River (Streamtec 1995), but there have been no recent records most likely due to high salinity. Carter's mussel is almost never found where salinity is greater than 1.6 g/L^{-1} (EC 2,900 µS/cm), with an acute salinity tolerance LC_{50} of 3.0 g/L^{-1} (approximately 4,600 µS/cm; Klunzinger et al., 2015). The estuarine species *Fluviolanatus subtorta* is known to colonise waterways that have become too saline for *W. carteri* (Kendrick 1976; Pen 1999) and its presence suggests the freshwater species no longer resides there. Moreover, the background EC of the Hotham River is well in excess of the known LC_{50} therefore extant populations are improbable.

9.2 Background water quality and post-dilution PAoC

With the exception of high TDS and EC, waters of the Hotham River were generally of good quality, and in most cases background analyte concentrations were below the ANZG (2018) DGVs for 95% species protection, applicable to moderately disturbed systems.

Concerning toxicants, for all analytes with sufficient background data available, the 80th percentile values were well below the DGVs (or LORs) for each with exception of zinc. However, there were very few exceedances once the HMTV was applied, which is a more accurate indication of actual toxicity to fauna (Warne et al., 2018). Most stressors were also below DGV, with the exception of EC, N_NOx and total N. Elevated background salinity and nitrogen (N_NOx and TN) are both indicative of agricultural land uses, and in acknowledgement of these legacy impacts the seasonal 80th percentile values are recommended as interim SSGVs (using TDS as a measure of salinity). Seasonal SSGVs were also applied to analytes without guidelines available, where seasonal differences were detected.

Estimates of concentrations of analytes in discharge water were provided for 21 analytes for use in hazard analyses, and of these, nine are modelled to occur in concentrations above the Hotham River SSGVs at the point of discharge. Order of dilution analyses found that the majority were unlikely to pose actual risk to the receiving environment, under the nominal dilution rate of 5:1[11] and maximum of 2,000kL/hr (555 L/sec). Post-dilution concentrations of aluminium, copper[12], molybdenum, ammonia, and nitrate N_NO_3 as a direct toxicant (not as a

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[11] A 5:1 dilution rate refers to discharge equivalent to 20% of natural catchment flows, additional to catchment flows.

[12] Based on adjusted DGVs provided in draft updates to ANZG guidance (ANZG 2023b).

stressor) were well below SSGV in each instance, including under elevated (80th percentile) background concentrations. Therefore, these analytes are not considered PAoC, however regular monitoring is strongly advised.

Cobalt and nitrite/nitrate ($N\_NO_{x;}$ as a stressor) remained as PAoC at post-dilution concentrations, and would be likely to cause sustained exceedances of interim SSGVs (or DGVs) under the nominal discharge scenario (5:1 dilution rate), and more conservative scenarios, in both wet and dry seasons.

9.2.1 Cobalt

Accumulation of cobalt from anthropogenic sources is becoming a major concern in agricultural fields and water bodies in many regions globally (Maher et al., 2020). Sources of cobalt in the environment are both natural and anthropogenic, with excesses of cobalt associated with industrial processes including production of hard metals, cement, and its use as an additive in products such as paint (ANZG 2018; Maher et al., 2020). Cobalt exists naturally as a trace element in surface waters, however levels in the Hotham were typically below laboratory LOR. Discharge water with cobalt concentration at the 50th percentile was estimated to increase the post-dilution median to more than twice the DGV. This represents a significant increase from background condition, and indicates a high risk of effectively constant exceedances. To maintain post-dilution cobalt concentration equal to SSGVs, at the modelled 50th percentile discharge concentration, an estimated maximum discharge rate of 13.5:1 would be applied, equivalent to 714 kL/hr in 10,000 kL/hr natural catchment flow. If actual discharge concentrations were closer to the modelled 95th percentile, then an estimated maximum of 21:1, or 483 kL/hr in 10,000 kL/hr catchment flow would be applied. These dilution rates may not be feasible in context of managing the forecast water balance at NBG.

The environmental toxicology of cobalt is much less studied in comparison to other heavy metals associated with contamination of surface waters. Cobalt is known to bioaccumulate in benthic organisms, plants, zooplankton and phytoplankton (CCREM 1987; ANZG 2018; Banaee et al., 2020; Mahey et al., 2020) and is shown to cause an array of biological effects including oxidative stress, impaired osmotic, reproductive and immune systems, and reduced growth (demonstrated for crustaceans, Banaee et al., 2020). Potential toxicity modifying factors are poorly understood. Acute toxicity of cobalt (96hr LC_{50})[13] was found to be significantly reduced under high hardness for the fish *Capoeta fusca* (Pourkhabbaz et al., 2011), however it is not known how this applies to the open environment. Several studies examine the lethal effects (96hr LC_{50}) of cobalt on various aquatic species (e.g. *Hydra,* Zeeshan et al., 2017; fish including *Tilapia nilotica* Rai et al., 2015; *Danio rerio*, Singh & Ansari 2017) however there are no studies on Australian fauna that the authors are aware of, and laboratory lethality assessments do not directly translate to ecological risk in the environment.

Without ecotoxicological information on local species, it is not possible to infer the exact outcome of discharges containing 0.014 mg/L (or higher) cobalt concentrations to the Hotham River. However, it is reasonable to assume the fauna residing there have not been exposed previously to high concentrations of cobalt. Therefore, a significant increase caused by discharge would be anticipated to have some impact, which will not be known until after the impacts have occurred. The displacement of species including fish, frogs and turtles are all possible, unless water treatment options that significantly reduce cobalt concentrations in discharge are considered achievable. Laboratory based ecotoxicity testing on local species covering a range of trophic levels would provide valuable insights into likely toxicity thresholds of Co in the Hotham River.

[13] The concentration the results in death of 50% of subjects at 96 hours exposure

9.2.2 Nitrite/nitrate N

The enrichment of nitrogen in freshwaters is a ubiquitous impact of human activities worldwide, and is well understood to directly accelerate primary production and alter trophic state (Dodds & Smith 2015; Wurtsbaugh et al., 2019; Campbell 2019). Surface waters in a eutrophic[14] state are predictably prone to excessive algal and cyanobacterial growth, which can cause related effects including production of toxic compounds, oxygen depletion, as well as development of surface scum, odour and loss of clarity (i.e. the formation of phytoplankton "pea soup"; Smith 2003; Boulton et al., 2014). Blooms result in a number of changes to water quality, including elevated pH during the day, and extreme diel fluctuations in dissolved oxygen. Eventually, the algal bloom dies, triggering intense microbial decomposition, severely depleting dissolved oxygen and creating anoxic "dead zones" (Chislock et al., 2013; Wurtsbaugh et al., 2019). Altogether, the gamut of ecosystem changes that can accompany eutrophication result in losses of biodiversity, through direct fish kills and habitat degradation, and severely impacts human use and aesthetic values of freshwater systems (Jakowyna et al., 2000; Smith 2003; Wurtsbaugh et al., 2019).

The Hotham River is not in an oligotrophic[15] state, and background levels of nitrogen (both N_NO_X and total N) are elevated by comparison to ANZG default guidelines for southwest rivers (ANZECC/ARMCANZ 2000). The interim SSGVs applied to the Hotham River therefore take the 80th percentiles for the wet and dry seasons respectively, acknowledging the already modified state of the system. Discharge at the modelled N_NO_x concentration would be expected to cause further enrichment. This would be especially pronounced during the dry season, when background levels are lower and the system is also more susceptible to the detrimental impacts of eutrophication, due to lack of flushing and warmer temperatures. This is a relatively high risk of adverse effects to aquatic ecosystem health. Discharge of water with elevated nutrients in winter is lower risk due to mixing, dilution, cooler temperatures and reduced sunlight.

Whilst exceedances of the N_NO_x stressor SSGV would be broadly expected under current anticipated discharge concentrations and volumes, exceedance of the toxicant SSGV for nitrate (N_NO_3) is not expected. If draft updates to the default guidance are accepted, then the new toxicant DGV may be as high as 29 mg/L (ANZG 2024a). It must be reiterated that the DGV for direct toxicity is not at all protective against eutrophication, and biodiversity loss would be expected well prior to that concentration being reached. Therefore it is strongly advised that the stressor SSGV should be used as an operational guideline for discharge, not the new toxicant DGVs.

9.2.3 Analytes requiring further investigation

The below analytes cannot be ruled out as PAoC without further data collection/provision, and further information regarding modelled discharge quality.

9.2.3.1 Cyanide

Cyanide can be found naturally in the environment in different forms, including organocyanides produced by some plants. Cyanide ion, hydrogen cyanide (collectively "free cyanide") and metal-cyanide complexes typically enter surface waters from anthropogenic sources (Jaszczak et al., 2017). Background concentrations of cyanide (inclusive of free, WAD, and total measures) were very low in the Hotham River, and it is unlikely that the river has been exposed to cyanide previously. Modelled discharge concentrations of WAD and total cyanide were provided for use in order of dilution risk assessment, whereas water quality guidelines

[14] Nutrient enriched, usually from anthropogenic pollution

[15] Nutrient limited, typical of pristine systems

refer to free cyanide, which is the most bioavailable (toxic) form. At best estimate, comparison of WAD cyanide concentrations indicates the 95% DGV for free cyanide is unlikely to be exceeded post-dilution, at the nominal discharge rate, and assuming the reduction calculated between permeate water and final discharge concentrations are realised (*cf*. Piteau 2024). However, given the profound toxicity of cyanide, and probable public sensitivity to discharge water containing cyanide, further investigation and hazard assessment is advised. This should include at a minimum estimation of free cyanide concentrations of discharge water, including level of confidence in those predictions.

9.2.3.2 TDS and EC

Modelled TDS and EC levels in discharge water were not provided for this risk assessment, and the approximations used in section 7.1.6 are not adequate to properly assess risk from significantly different TDS/EC, or change in ionic composition (i.e. partial components of TDS). Rudimentary analysis applied, based on comparisons of total ionic load, suggests discharge water could be fresher than background conditions of the Hotham River. Reduction of TDS from discharge of fresher water may provide a benefit to downstream environments, as reduction in osmotic stressors may allow temporary establishment of more salt-sensitive species and improve fitness of the inhabitant fauna. Reduction in TDS may also displace salinity tolerant species including the Swan River goby and estuarine mussel *Fluviolanatus subtorta* over the duration of discharge. However, addition of water that is markedly fresher than existing catchment condition may limit the amount of mixing that occurs, due to density differential between saltier (heavier) and fresher waters, especially in summer when flows are negligible and there is minimal mixing. There is potential for density stratification of refuge pools to occur if mixing between the layers is incomplete (e.g. Western et al., 1996; Turunen et al., 2020). Stratification can cause deoxygenation in lower layer/s of the water column, resulting in anoxia and suffocation of aquatic fauna. This risk would be greatest during summer low/no flow periods, and the effects most acute in deeper pools (i.e. summer refuges). These pools are critical refuge habitats for a range of fauna, including Rakali and likely also cobbler.

Provision of estimated discharge salinity (TDS and EC) would be required to assess the actual likely differences between discharge and catchment water, and conduct order of dilution assessment. However, estimations relating to post-discharge mixing of discharge water, including scenarios where a density differential exists, would need to be conducted by a suitably qualified professional.

9.3 Considerations for discharge timing and location

The hazard analysis presented assumes discharge would only occur when catchment flows to the Hotham River are sufficient to dilute mine derived water to a maximum of one part in six (or a 5:1 river:mine water ratio), which in practical terms would exclude discharge occurring over much of the dry season. SSGVs are presented for the dry season should investigations into year round-discharge progress, as there was some seasonality identified in background water quality of the Hotham River, with differences in nutrients and some metals identified, as well as seasonal differences in salinity. However, high consideration should be placed on the reduced capacity for dilution under low or no flow conditions. Reduced mixing of discharge and riverine water may mean that any potential water quality risks to the receiving environment are amplified during the dry season, for example, addition of nitrogen-nutrients as an identified PAoC. Furthermore, as discussed in sections above, potential for an EC/TDS differential between discharge water and residual pools, or a temperature differential (not assessed) may mean that discharge water is incompletely mixed in deeper pools. This may contribute to stratification of pools over summer which would have direct consequences to aquatic fauna through oxygen depletion and loss of habitat quality, particularly if eutrophication was also occurring. Therefore, it is recommended discharge occur during the wet season only, defined as months where flows are consistently above the median (i.e. June through to October).

Previous investigations include potential for discharge to a tributary creek, rather than directly to the Hotham River, with target creeks including Gringer Creek (which flows to the Hotham via the Bannister River; SLR 2024b) and Boggy Brook (SLR 2024c). Provided any identified impacts to the aquatic values of those tributaries are deemed acceptable, benefits of discharging into a tributary are presumed mixing of discharge with local surface water before it reaches the main river, which may include potential for biogeochemical processes to reduce risk from PAoC, for example adsorption of metal ions onto inorganic and/or organic substrate particles, or assimilation of nutrients. The water qualities of Gringer Creek and the downstream reaches of Boggy Brook share some similarities to the receiving Hotham River, notably high salinity (SLR 2024b, SLR 2024c). However, the magnitude of catchment flows are very different, for example Gringer Creek wet season flows are generally less than the proposed maximum discharge volumes of 2,000 kL/hr[16], and Boggy Brook is highly ephemeral, therefore addition of discharge water would represent a marked change in flow regime. The volumes of discharge water proposed would likely alter the geomorphology of the creeks, increasing erosion and downstream sediment transport. Even if this is deemed to be acceptable from an aquatic ecological standpoint, erosion would be especially prevalent through areas of cleared farmland (which both of these creeks intersect), which may also present complications with landholders. Discharge to the upper reaches of Boggy Brook (within relatively undisturbed native jarrah forest) was not recommended (SLR 2024c); discharge locations might also consider an already impacted tributary, such as 34 Mile Brook, however the appropriateness and potential impacts would require further consideration.

Provided discharge rates are kept to within the maximums assumed under this hazard assessment for the Hotham River, direct discharge to the river during the wet season is expected to undergo sufficient dilution and thus pose limited threat to the receiving ecosystem (noting Co and N_NOx as PAoC), provided real discharge concentrations are not higher than modelled by Piteau (2024). Decisions regarding the discharge location would need to assess the balance of risk posed by PAoC in discharge to the Hotham, against discharge to a tributary including minimal dilution, potential loss/alteration of aquatic environmental values, and likely erosive impacts especially where creeklines traverse farmland. Discharge to a tributary creek may also pose the risk of direct toxicity to aquatic fauna, due to minimal dilution of PAoC in discharge. Any tributary receiving this discharge would need to be considered a sacrificial mixing zone due to the reduced dilution.

10.0 Recommendations

Based on current knowledge of background water quality conditions of the Hotham River, and the hazard assessment using modelled discharge water quality provided by NBG:

- It is recommended that mine discharge only occur during the wet season months (June to October), when flows are above the median recorded at the Hotham Weir. Discharge during the dry season or during low flows would ideally be avoided.
- Two key analytes of potential concern were identified, cobalt (toxicant) and nitrite/nitrate-N (stressor). Reduced concentrations of these analytes in discharge water would reduce potential risk to the receiving environment.

[16] Median wet season flow in 2023 at the Gringer Creek upstream gauge was 150 kL/hr, to a maximum of 2,970 kL/hr; at Gringer downstream 435 kL/hr, and maximum 3,025 kL/hr. Data supplied by NBG.

- Further investigation into discharge plume modelling and extent of mixing zones, to determine the spatial extent of discharge influence on the river, taking into account the distinct seasonality of the Hotham River.

Further water quality data requirements to support hazard assessment and future monitoring:

- Additions to the regular water quality monitoring suite for the Hotham River have recently commenced including hardness, dissolved organic carbon, nitrite/nitrate-N, and free cyanide at an LOR < 0.004 mg/L, including monthly replicates from monitoring sites on the Hotham River. These data will be incorporated into future updates of SSGVs. Ideally, the ANZG (2018) recommends 24 months of data for development of baseline SSGVs.
- Laboratory LORs for toxicants have been recently reviewed to ensure sufficient baseline and monitoring data is comparable to ANZG (2018) DGVs.
- To support hazard assessment, modelled discharge concentrations for free cyanide, electrical conductivity, total dissolved solids and total suspended solids should be provided.
- Implementation of an aquatic fauna baseline study including macroinvertebrates, to commence 3 years prior to the commencement of discharge, to underpin monitoring throughout discharge operations. Depending on the discharge location, baseline surveys of fish may also be required.
- Predicted discharge quality is presently under review, however if cobalt concentrations continue to be PAoC then ecotoxicity testing for cobalt on a range of south-west species from a range of trophic levels should be considered. Ecotoxicity testing is the best way to predict likely faunal toxicity thresholds of cobalt in the Hotham River, and to better predict likely consequences of exceeding thresholds.

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