Gringer Creek Aquatic Fauna Survey & Interim Site-Specific Guideline Values

Newmont Boddington Gold

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Executive Summary

Newmont Boddington Gold (NBG) operates the Boddington Gold Mine (BGM), located 17 km northwest of the town of Boddington, and around 100 km to the southeast of Perth in WA. Mine tailings are stored in Residue Disposal Areas (RDAs), and the active tailings facility (F1/F3 RDA) will reach its current approved maximum capacity in mid-2025 at 600Mt. NBG submitted approvals to increase the existing facility to 750Mt in July 2023, and is also investigating options for placement and storage of tailings material once the current RDA reaches a maximum storage capacity of 750Mt from mid-2029. Newmont is currently completing studies to support approval applications for a second storage facility in the adjacent Saddleback tree farm, to be called RDA 2. The proposed location of the new RDA 2 facility overlies the mid-reaches of Gringer Creek, a seasonal tributary of the Bannister River, which flows into the Hotham River southeast of the NBG project area.

The aim of this project was to document and assess the aquatic ecological values of Gringer Creek, in order to provide supporting documentation required under the Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act) and Part IV of the WA Environmental Protection Act 1986 (EP Act). Previous baseline aquatic surveys on Gringer Creek were conducted by Wetland Research & Management (WRM) in 2011, those results are compared to the current study to determine differences between the two sampling times. In addition, options for discharge of excess water, either dewatered from the developing pit or treated process water, will be explored, with options including discharge to Gringer Creek or another small tributary to the Hotham River, Boggy Brook. As part of environmental investigations into water discharge, SLR were also commissioned to derive interim site-specific guideline values (SSGVs) for the receiving creekline for discharge to the environment using available and new water quality data.

Three key components of the study were completed:

1. A brief desktop review of aquatic ecosystem values of Gringer Creek and nearby creeks and rivers;
2. A baseline field survey, documenting water quality, habitat, and aquatic fauna
3. Analysis of long-term water quality monitoring data and assessment against default guideline values (DGV) (ANZG 2018), and derivation of interim SSGVs.

Key findings of the baseline survey:

- Water quality at Gringer Creek was characterised by high electrical conductivity, and low concentrations of metal analytes, with exception of local elevations of cobalt and manganese. Elevated Co and Mn were detected in both 2011 and 2023 baseline studies.
- Gringer Creek supported a moderately diverse macroinvertebrate fauna, including locally endemic species, as well as three species of native fish and two species of native crayfish. Gringer Creek was found to support fish breeding in both 2011 and 2023 baseline surveys.

In 2023, conductivity was high (5,960 µS/cm), typical of streams in the region with legacy of catchment clearing and secondary salinization. Otherwise, water quality measured at Gringer Creek in 2023 was generally good, with circum-neutral pH, dissolved oxygen at around 100% saturation, and with the majority of toxicant analytes well below DGVs. Some exceptions included cobalt, recorded above the DGV of 0.0014 mg/L at several sites, and elevated manganese was recorded at GRDS02. Elevated cobalt was also recorded during 2011

baseline surveys, as was elevated manganese at GRDS02 and particularly GRDS06. The source of cobalt and manganese at these sites is unknown.

A total of 72 macroinvertebrate taxa were recorded from Gringer Creek in 2023. There were no formally listed conservation significant macroinvertebrate taxa recorded, however there were a number of south-west endemic species including Gondwanan damselflies *Archiargiolestes* sp., several endemic beetles, stoneflies (Plectoptera) *Leptoperla australica* and several leptocerid and hydroptilid caddisflies (Trichoptera). Altogether, the macroinvertebrate community of Gringer Creek featured a moderately diverse assemblage of taxa, of which at least a quarter are known endemic species, including species of Plecoptera and Trichoptera, known to be sensitive to changes in water quality.

A total of 275 fish of three endemic species were captured and released from Gringer Creek, including the nightfish *Bostockia porosa*, western minnow *Galaxias occidentalis* and the western pygmy perch *Nannoperca vittata*. All three species were recorded in the 2011 survey, and there were no introduced fish detected during the current survey. Several gravid females of the western minnow and nightfish were recorded in the upper reaches of Gringer Creek, particularly GRCK01 – 04. Both species are known to undergo annual breeding migrations in winter and spring, entering small tributary creeks and moving to the upper reaches to spawn (Pen & Potter 1990; 1991). During the 2011 baseline studies (conducted slightly later in October) the native fish catch was dominated by juvenile size classes. Evidently, Gringer Creek is being used by local fish populations (of the Bannister River, and potentially the Hotham River) as spawning and nursery habitat, therefore continued connectivity between Bannister River/Hotham River and this breeding habitat in the headwaters should be maintained.

Long term water quality

Water quality monitoring data supplied by NBG and WRM/SLR baseline data (2011 to 2023) were examined to define any long-term trends in water quality analytes from Gringer Creek, and to determine whether ANZG (2018) DGVs are appropriate for ongoing monitoring under future mine water discharge scenarios, or if interim site-specific guideline values need to be developed.

Generally, most analytes were recorded below ANZG (2018) 95% species protection DGVs and the default guidelines remain appropriate. Several analytes did record elevations over time, but the DGV was still appropriate for use. Several analytes had 80th percentile values greater than the DGV, therefore it was recommended to implement an interim SSGV rather than the default value. A summary of key analyte background concentrations is presented in Table E1, and interim SSGVs are presented in Table E2.

Table E1. Summary of analytes above the ANZG (2018) 95% species protection DGV during long-term monitoring at Gringer Creek (2012 - 2023).

Analyte	Exceedances	Recommendation
Al	Frequent spot elevations in winter months, likely due to natural enrichment within the catchment. 80^{th} %ile < DGV	Retain ANZG (2018) 95% DGV Implement seasonal interim SSGV to account for natural releases with onset of winter rain.
As	Isolated elevations in Nov-19 and Dec-19, below the 80% protection level.	Retain ANZG (2018) 95% DGV
Co	Frequently elevated at Gringer Creek, suggesting higher 'background' levels. Four exceptional outliers recorded at GRCK 06	Interim SSGV Removal of four outliers from 80^{th} percentile to improve conservatism
Cu	Limited data with appropriate LOR restricted statistical analysis. Some isolated elevatios.	Adopt updated ANZG guidance from 2023, in anticipation of publication. SSGV requires local DOC data.
Mn	Occasionally elevated, to a maximum of 22.2 mg/L dissolved Mn. Source of Mn is unknown. 80^{th} %ile < DGV	Retain ANZG (2018) 95% DGV
Zn	At default hardness = 30 mg/L $CaCO_3$ 45% of measurements were above the DGV, however implementation of a HMTV (recommended) reduced exceedances to 4% of observations	Interim SSGV is HMTV at hardness = 450mg/L as $CaCO_3$ Further measurement of hardness is recommended (currently 26 records)

Interim SSGVs

In most instances, the ANZG (2018) DGV for 95% species protection remained appropriate for use as interim SSGV at Gringer Creek. However, there were some analytes that had elevated 'background' levels (relative to DGV). The bioavailability of some metals is affected by toxicity modifying factors (e.g. hardness, dissolved organic carbon), therefore the DGV applying the appropriate correction factors were used as required.

- Hardness modified trigger values were derived using the minimum hardness recorded (450 mg/L) for Cd, Cr, Ni, and Zn. However, there were few hardness records available (*n* = 26).
- The DGV was retained for Al in the dry season, however, in acknowledgement of pulses of Al with the onset of winter rains, a seasonal interim SSGV is proposed, applicable to increased flow in response to rainfall.
- The interim SSGV for ammonia was derived using the 80^{th}%ile pH (7.1) and temperature (20ºC). Using a higher value for pH and temperature (in this case, the 80^{th}%ile rather than the median) is a more conservative approach (ANZG 2023a), and is recommended to be more protective.
- Without local DOC data, updated guidance on Cu in freshwater provide DGVs standardised at DOC =< 0.5 mg/L[1] (ANZG 2023b). The standardised DGVs are likely to be overly conservative, especially given high hardness, however derivation of an operation SSGV will require regular collection of DOC data.
- The proposed interim SSGV for temperature comprises the 20^{th} – 80^{th}%ile values. Water discharge from the operations should not increase stream temperature.
- pH has an upper and a lower DGV, as excessive alkalinity (high pH) or acidity (low pH) can be detrimental. The lower DGV for pH was altered to match the 20^{th}%ile value.

[1] 0.5 mg/L DOC is likely to be conservatively low, jarrah/marri forest stream are likely to have much higher DOC. Once the data are available, the SSGV can be modified.

Table E2. Interim site-specific guideline values for use at Gringer Creek. DGVs for non-metal toxicants are denoted with (T) to distinguish from those associated with eutrophication (E).

Analyte	Unit	ANZG 95%	ANZG 99%	80th%ile value	Interim SSGV
Al (dry season)	mg/L	0.055	0.027	**0.05**	**0.055**
Al (wet season)[A]	mg/L	0.055	0.027	**0.19**	**0.19**
Alkalinity	mg/L	N/A	N/A	**64**	**-**
As	mg/L	0.024	0.001	**0.00012**	**0.024**
B	mg/L	0.94	0.34	**0.11**	**0.94**
Ba	mg/L	N/A	N/A	**0.09**	**0.09**
Cd	mg/L	0.0002	0.00006	**<LOR**	**0.002[B]**
Co	mg/L	0.0014	N/A	**0.013**	**0.012[C]**
Cr (IV)	mg/L	0.001	0.00001	**<LOR**	**0.01[B]**
Cu	mg/L	0.00047	0.0002	**0.002**	**0.00047[E]**
Cyanide - free	mg/L	0.007	0.004	**<LOR**	**0.007**
DO	%	80 - 120		**83 – 102**	**80 - 120**
EC	(µS/cm)	120-1500^	N/A	**10,503**	**-**
Fe – dissolved	mg/L	N/A	N/A	**1.0**	**1.0**
Hardness	mg/L	-	-	**1,500**	**450 – 2400[F]**
Mn	mg/L	1.9	1.2	**1.3**	**1.9**
Mo	mg/L	0.034	N/A	**<LOR**	**0.034**
N-NH_3* (T)	mg/L	0.76	0.25	**0.1**	**0.76[G]**
N-NO_3 (T)	mg/L	2.4	1	**0.09**	**2.4[H]**
N-NO_X (E)	mg/L	0.2	-	**0.08**	**0.2**
N-total (E)	mg/L	1.2	-	**0.6**	**1.2**
Ni	mg/L	0.011	0.008	**0.002**	**0.11[B]**
P-total (E)	mg/L	0.065	-	**0.02**	**0.065**
pH	H+	6.5 - 8	N/A	**6.3 – 7.2**	**6.3 – 7.2[I]**
Pb	mg/L	0.0034	0.001	**<LOR**	**0.0034**
S-SO4	mg/L	-	-	**299**	**299**
Se	mg/L	0.011	0.005	**<LOR**	**0.011**
Temperature	C			**10.4 - 20.2**	**-**
TDS	mg/L	-	-	**7,405**	**7,405**
TSS	mg/L	-	-	**<LOR**	**5**
Turbidity	NTU	10 - 20		**9.3**	**20**
U	mg/L	0.0005	-	**<LOR**	**0.0005**
V	mg/L	0.0006	-	**0.0002**	**0.0006**
Zn	mg/L	0.008	0.0024	**0.02**	**0.08[B]**

A – 'wet season' as determined by rainfall, including first flush winter rains and proceeding large rainfall events.
B – Hardness modified trigger value (HMTV) calculated at hardness = 450 mg/L as $CaCO_3$.
C – Interim SSGV is the 80th%ile value, excluding high outliers.
D – Calculated from Chemcentre data only (*n* = 19).
E – ANZG (2023b) DGV at DOC =< 0.5 mg/L used, in absence of local DOC data. Measurement of local DOC advised.
F – Hardness values are min-max range recorded from Gringer Creek (*n* = 26). Recommend further collection of hardness data to support full operational SSGVs.
G – Calculated using 80th%ile pH (7.1) and temperature (20ºC). See ANZG (2023b), and rationale in section 5.2.
H – ANZG (2018) recommend use of 'grading' trigger values presented in Hickey et al., (2013) as 95% DGV, *in lieu* of updated guidance specific to Australia.
I – 20th to 80th%ile range for Gringer Creek.

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Acronyms and Abbreviations

ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
34MB	34 Mile Brook
ANOVA	Analysis of variance
ANZECC	Australian and New Zealand Environment and Conservation Council
ANZG	Australian and New Zealand Guidelines
BGM	Boddington Gold Mine
BOM	Bureau of Meteorology
CL	Carapace length
CPUE	Catch per unit effort
DBCA	Department of Biodiversity, Conservation and Attractions
DGV	Default guideline value
DoW	Department of Water
DPIRD	Department of Primary Industries and Regional Development
DWER	Department of Water and Environmental Regulation
EC	Electrical conductivity
EPA	Environmental Protection Authority
EWR	Ecological water requirement
HMTV	Hardness Modified Trigger Value
IUCN	International Union for the Conservation of Nature
LoE	Lines of evidence
LWD	Large woody debris
NBG	Newmont Boddington Gold
nMDS	non-metric multidimensional scaling
NWQMS	National Water Quality Management Strategy
RDA	Residue Disposal Area
SIMPROF	Similarity profiles
SL	Standard length
SLR	SLR Consulting Australia
SSGV	Site-specific guideline value
SWA	South-western Australia
TDS	Total dissolved solids
TL	Total length
TSS	Total suspended solids
WoE	Weight of evidence
WQMF	Water Quality Management Framework
WRM	Wetland Research and Management

1.0 Introduction

Newmont Boddington Gold (NBG) operates the Boddington Gold Mine (BGM), located 17 km northwest of the town of Boddington, and around 100 km to the southeast of Perth in WA. Open pit oxide mining and stockpile processing was undertaken at the site from 1987 to 2002. After a period of care and maintenance, construction of a large scale open pit mining operation to exploit the hard rock ore body was commenced by Newmont in 2006. 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 co-located Residue Disposal Areas (RDAs). The active tailings facility, known as the F1/F3 RDA, will reach its current maximum approved capacity in mid-2025 at 600Mt (at stage 18). NBG is currently progressing approval applications to increase the existing facility to 750Mt.

NBG is also investigating options for placement and storage of tailings material once the current F1/F3 RDA reaches a maximum storage capacity of 750Mt from mid-2029. Newmont is currently completing studies to support approval applications for a second storage facility in the adjacent Saddleback tree farm, to be called RDA2 (Figure 1). The RDA2 was approved under MS971 and EPBC 2012/6370 in 2014. However, due to a better understanding of footprint requirements and dam design methodology, it is anticipated the project will be referred again to the environmental regulators in 2024. As a result, Newmont is reviewing baseline studies and, as water treatment is currently being explored by the study, potential water quality discharge criteria are being considered. The proposed location of the new RDA facility lies adjacent to the mid-reaches of Gringer Creek, a seasonal tributary of the Bannister River, which flows into the Hotham River southeast of the NBG project area.

SLR Consulting were commissioned by Newmont Australia to document and assess the aquatic ecological values of Gringer Creek, in order to provide supporting documentation required under the Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act) and Part IV of the WA Environmental Protection Act 1986 (EP Act). The report discusses the results of the latest 2023 Spring survey as well as historical surveys along the same stretch of this creek.

In addition, options for discharge of excess water from the developing pit will be explored, options including discharge to Gringer Creek and/or another small tributary to the Bannister River, Boggy Brook. As part of environmental investigations into water discharge, SLR were also commissioned to derive interim site-specific guideline values (SSGVs) for receiving waters subject to discharge to the environment using available and new water quality data.

1.1 Scope of Work

The scope of work for the current project included a review of available current knowledge on the aquatic ecosystems of Gringer Creek and surrounds, and a repeat of 2011 baseline surveys of Gringer Creek within the proposed RDA2 project area. Gringer Creek has been identified as a potential receiving location for discharge water, thus the scope also includes development of Water Quality Guidelines/limits for this system.

Specifically, the current study included:

- Desktop review of existing aquatic fauna reports to inform current aquatic surveys.
- Systematic sampling of aquatic fauna (macroinvertebrates, fish and crayfish) and water quality (in situ parameters, ions, nutrients and metals) for Gringer Creek.
- Water quality reported against the Australia New Zealand Guidelines (ANZG 2018), part of the National Water Quality Management Strategy (NWQMS). The NWQMS is an update of the previous Australian and New Zealand Environment and Conservation Council

(ANZECC) and Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) Water Quality Guidelines (ANZECC/ARMCANZ 2000).

- Sampling design, methods and general approaches were consistent with the following:
  - EPA (2018) Environmental Factor Guideline: Inland Waters. Environmental Protection Authority, Western Australia. 29 June 2018.
  - Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018);
  - EPA Position Statement No. 3, Terrestrial Biological Surveys as an Element of Biodiversity Protection (EPA 2002); and,
  - EPA Guidance No. 56, Terrestrial Fauna Surveys for Environmental Impact Assessment in Western Australia (EPA 2004).
- All aquatic fauna to be identified to species level, where possible.
- Conservation status of each species recorded, and an assessment of any changes in distribution compared to previous baseline surveys conducted in 2011-13.
- Development of interim SSGVs for discharge to the environment.

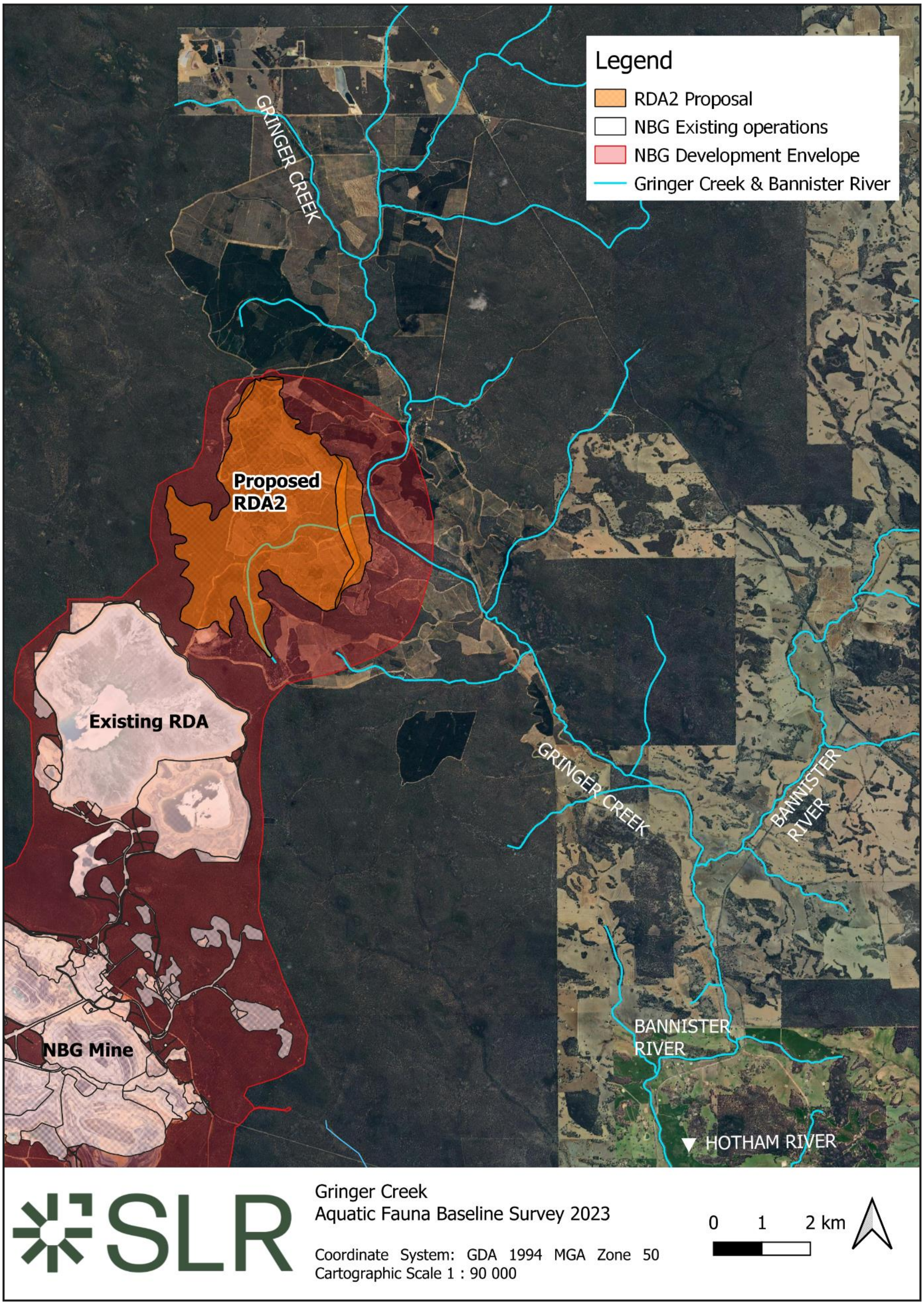


Figure 1. Location of the proposed RDA2 footprint in relation to Gringer Creek and existing NBG operations.

2.0 Environmental Setting

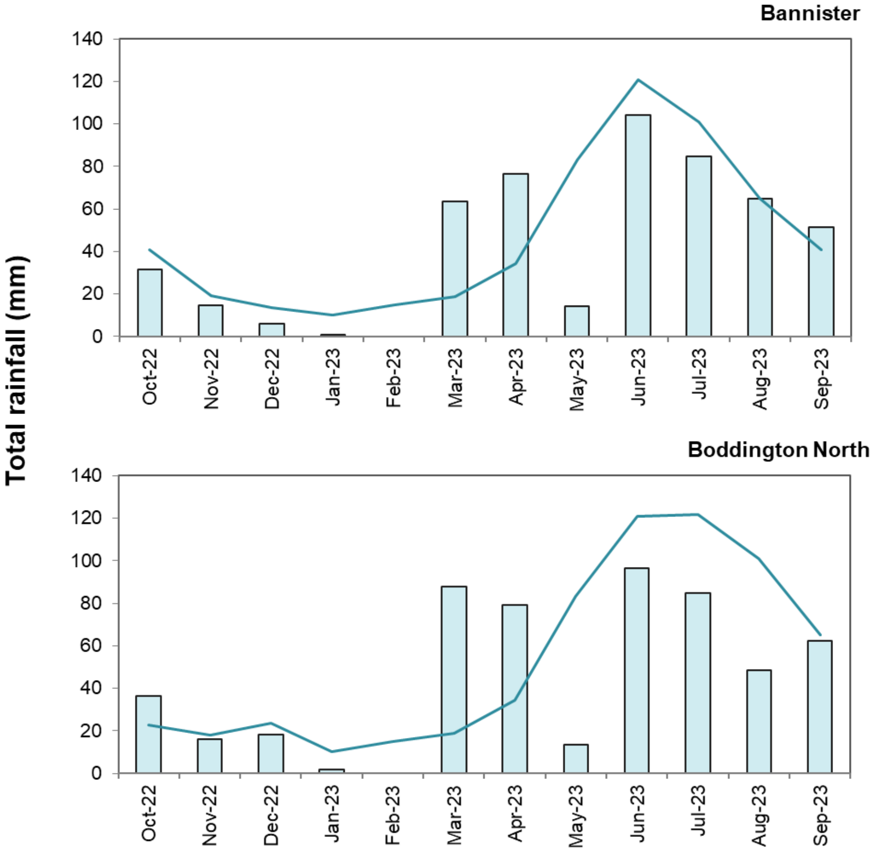
2.1 Climate and Rainfall

The south-west region of Western Australia 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 exception of occasional summer storms. As such, many low order 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 (Figure 2). Annual rainfall for the catchment varies between 560 – 635 mm, and was below average in 2023 (Figure 3). When compared to long-term average data, rainfall in the study area was below average for 2023, with an unusually dry May (

Figure 2. Total monthly rainfall (Oct-22 to Sep-23) overlain with long-term average rainfall for Bannister (1891 – 2023; BOM 009507) and Boddington North (2011 - 2023; BOM 109516) (BOM 2023).

). A continuing trend of declining winter rainfall has been observed across the south-west since approximately 1970, and is expected to continue to intensify as climate change progresses (McFarlane et al., 2020).



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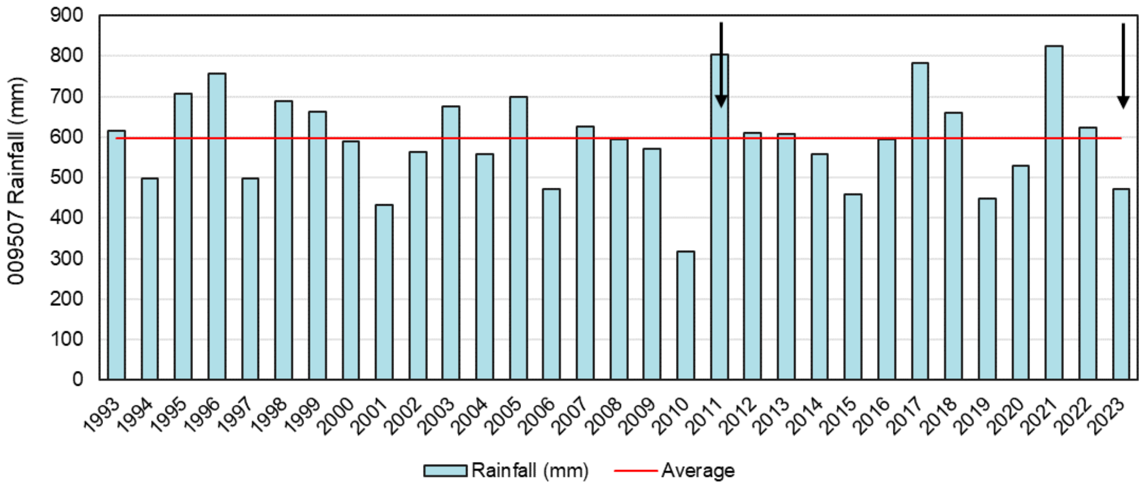


Figure 3. Annual rainfall recorded from Bannister between 1993 – 2023 (BOM 009507). Average (1993 – 2023) total is also given. Black arrows indicate timing of aquatic baseline surveys in 2011 and 2023.

2.2 Surface Hydrology

Gringer Creek is located in the Hotham River catchment, which is a sub-catchment of the greater Murray River catchment. Streamflow in Gringer Creek is highly seasonal, with flows commencing in late May/early June, peaking in late winter and early spring, usually ceasing by late November or December (Figure 4). Summer rainfall tends to generate only short-lived flow events, if at all (Figure 4).

The closest long-term flow monitoring weir is located on the Hotham River at Marradong Bridge (Figure 5). When compared to long-term average data, winter streamflow in the Hotham River was below average in 2023. As Gringer Creek is a tributary of the Hotham River (via the Bannister River), it can be assumed that flows in Gringer Creek were also below average in 2023.

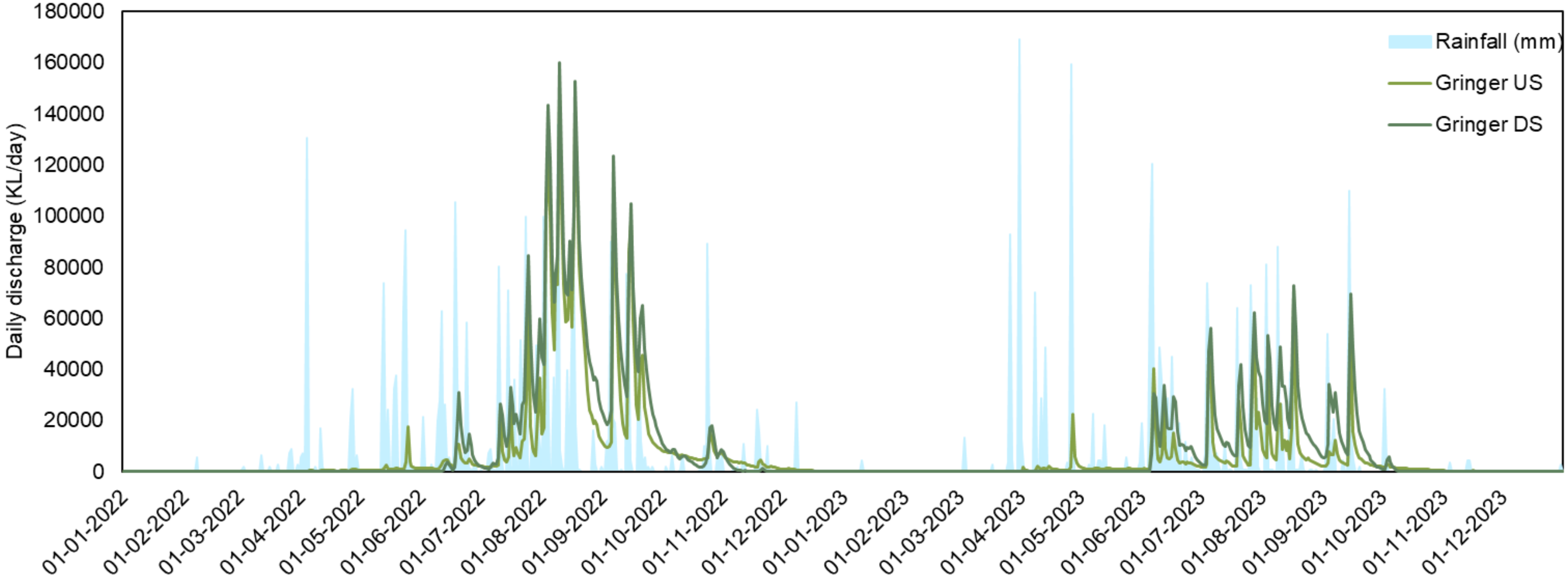


Figure 4. Daily flow hydrographs (KL/day) for Gringer Creek (upstream and downstream gauges) for 2022 and 2023, overlain with daily rainfall observations (mm) for Bannister (BOM station 009507).

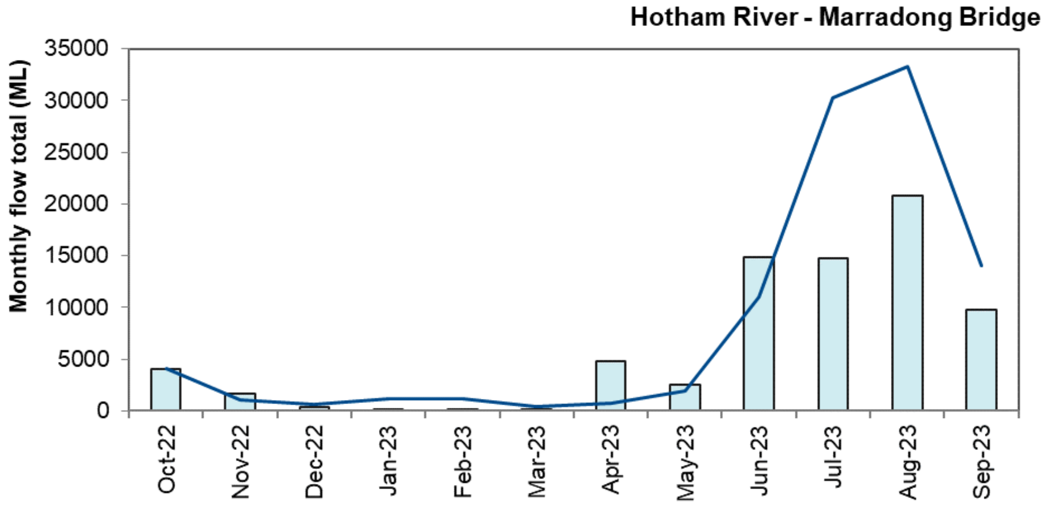


Figure 5: Total monthly streamflow (ML) for the Hotham River (Marradong Road Bridge 614224) including long-term average streamflow (solid line) (DWER 2023).

3.0 Desktop review

3.1 Methods and guidance

A review of aquatic ecosystem values within the vicinity of the project area is provided below, including relevant records from published scientific literature, unpublished reports, and other grey literature (Table 1). The majority of available literature focusses in and around the Hotham River near Boddington and surrounds. Data reviewed here also includes the WRM (2011) baseline survey of Gringer Creek.

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) and,
- 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.

All fauna recorded during the field surveys were also assessed using these conservation listings and databases (see sections 4.2.3, 4.2.4, 4.2.5).

Table 1. Summary of available literature assessing aquatic fauna in vicinity of the proposed RDA2.

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	2024	Hotham River Fish Monitoring 2023	Hotham River	Fish populations, hydroregime, long-term analysis
WRM	2012a	Gringer Creek baseline 2011	Gringer Creek	WQ, macroinvertebrates, fish & crayfish
WRM	2012b	Hotham Farm WQ & aquatic fauna survey	Jungellan Creek	WQ, macroinvertebrates, fish & crayfish, physical form
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

3.2 Summary of results

3.2.1 Water quality records

Several broader scale regional assessments overlapping the project area have been conducted that include an assessment and discussion of water quality, including that of the Hotham and Williams rivers (WRM 2019, 2020) and the wider Avon-Hotham catchments (Sharafi et al., 2005). Generally, waterways in the vicinity of the project area 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), 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). Boggy Brook is a minor creekline draining a forested catchment, and is fresh (EC 159 – 262 µS/cm), whereas study reaches in proximity to cleared farmland were brackish (>2,500 µS/cm) (WRM 2012c).

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, it is almost certain 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 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.

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 in vicinity of the RDA2 area indicate circum-neutral to slightly alkaline pH, well oxygenated waters, and good clarity (WRM 2012a,b; 2013). This regional context informs assessment of prior baseline studies of water quality at Gringer Creek (WRM 2012a). Water quality in Gringer Creek sites was characterized as brackish to saline (5,420 - 11,800 µS/cm EC), neutral to slightly acidic pH (6.3 – 7.1), generally well oxygenated (daytime dissolved oxygen DO 79 – 101%) and with oligotrophic nutrient status. In addition, water quality attributes included high hardness (750 – 1,900 mg/L hardness as $CaCO_3$), and most metal toxicant concentrations at or below laboratory detection limits recorded in October 2011. Ionic composition indicated Na and Cl to be the dominant ions, accounting for 35% of the total cation and 49% of the total anion equivalence, respectively. Ca^{2+} and $SO4^{2-}$ were sub-dominant at all sites.

Interestingly, the above description of water quality applied to all sites on Gringer Creek, except one site which had poor water quality. Upstream site GRCK04 (WRM 2012a site GC4/S04) differed to the others in having particularly low pH (4.7), low DO (32.5%) and low alkalinity (<1 mg/L), and relatively low hardness (580 mg/L $CaCO_3$). Daytime DO levels (32.5% saturation) were far below the lower DGV (90%) and below levels known to cause stress in aquatic fauna, particularly fish (*i.e.* <50%; Connolly *et al.* 2004, Flint *et al.* 2014). The low daytime levels at GRCK04 indicate anoxia (zero DO) was likely to occur overnight. Dissolved Al was 6 times greater than the default 95% TV (0.055 mg/L) and twice the default 80% TV (0.15 mg/L), and dissolved iron was also high (1.7 mg/L Fe). Low pH likely explains the increased concentrations of these metals, which are known to occur as more soluble (and bioavailable/ toxic) forms under these acidic, reduced conditions (ANZG 2018). Because this site is situated short distances between upstream and downstream sites, which all had comparatively good water quality, this site may indeed reflect local habitat conditions, as it is located in a section of stream that supports a teatree swamp community, which contributes organic matter and tannins which would work to reduce pH and DO levels through decomposition. Also of note, teatree communities contribute DOC to streams, which moderates the toxicity of metals such as Cu.

3.2.2 Aquatic Invertebrates

A total of 47 macroinvertebrate taxa[2] were recorded from Gringer Creek in October 2011. 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 *Paramerina levidensis*, *Procladius paludicola*, *Chironomus* aff. *alternans* and *Tanytarsus* sp. and biting midge larvae *Ceratopogoninae* sp. were particularly abundant. Other common taxa included the amphipod *Austrochiltonia subtenuis* and larvae of the diving beetle *Sternopriscus* sp.. Bunn and Davies (1992) and WRM (2011a) recorded similar dominance of *Tanytarsus* species and *Austrochiltonia subtenuis* in the salinised Thirty Four Mile Brook and Hotham River. Bunn and 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 at Gringer Creek in 2011. Of total fauna, 14% were considered south-west endemics, or likely south-west endemics, including the damselfly *Archiargiolestes pusillus*, the diving beetle *Megaporus solidus*, the caddis-fly *Oecetis* sp. and the chironomid *Paramerina levidensis*. Other species recorded near the study area include Western Australian endemic beetle *Necterosoma darwini*, *Symphitoneuria wheeleri* which is a Western Australian endemic, salinity-tolerant caddisfly (unusual among a group of taxa that are typically sensitive to poorer water quality) and the dragonfly larvae *Procordulia affinis*, which is a south-west endemic species (WRM 2020). Altogether, whilst no listed species were recorded, the presence of several regionally endemic taxa does afford this macroinvertebrate assemblage conservation significance, especially considering regional pressures from development and climatic drying.

3.2.3 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, one native estuarine species the Swan River goby (*Pseudogobius olorum*) and two introduced species,

[2] 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).

Mosquitofish (*Gambusia holbrooki*) and redfin perch (*Perca fluviatilis*) are also known to occur within the catchment (WRM 2016, 2017, 2021). Limited studies have been undertaken upstream of Lions Weir, at Boddington townsite, and the headwater catchment (e.g. Morgan & Beatty 2004), with the majority of historical records from studies downstream of Lions Weir, but upstream of the Marradong Road bridge (WRM 2016, 2017, 2018, 2020, 2021; SLR 2024).

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). Although minnows are a freshwater species, studies indicate that, like western pygmy perch, adult minnows 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. Although tolerant to 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; WRM 2012d). Both species are known to migrate to headwater tributaries to spawn, with western minnow breeding generally occurring 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) thus a significant risk to riverine populations is loss of connectivity to seasonal tributaries (WRM 2012d).

Nightfish are also widespread in SWWA. 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). This 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 species 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 bodies 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). It is possible that cobbler may move into Gringer Creek when flows and depths are sufficient for their passage, particularly in the lower reaches in proximity to the Bannister River. However it is not known whether Gringer Creek would be preferred habitat for cobbler.

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)

Two introduced species have been recorded in the region surrounding Gringer Creek. 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).

3.2.4 Crayfish

There are six native crayfish species from the genus *Cherax* and five from the genus *Engaewa* in SWWA, 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, 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 Gringer Creek. 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.

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. During the 2011 Gringer Creek baseline studies, koonacs, gilgies and smooth marron were recorded. Koonacs and gilgies are also known from other nearby intermittent creeklines, including Boggy Brook, Jungelan Brook and 34 Mile Brook (WRM 2012b,c).

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 and Knott 1996). Koonacs tend to dominate assemblages where water tables fluctuate markedly (Beatty et al. 2006). They burrow to avoid summer drying, and may remain in their burrows for long periods, only leaving when surface waters return in early winter.

The yabby is native to south eastern Australia, and was first introduced to farm dams in south west Western Australia in 1932. Currently, the distribution of the yabby is from the Hutt River in the north to Esperance in the southeast, with isolated occurrences further inland, such as along the railway line to Cue, and in a Gully system near Leonora (Lynas et al. 2004, 2006, Beatty et al. 2005).

3.2.5 Other Aquatic fauna

3.2.5.1 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. There are few available data on the spatial distribution of water rats along the Hotham River or its tributaries. WRM have confirmed presence of the water rat at 34 Mile Brook, which is 800m upstream of Tullis Bridge and 7kms downstream of the Boddington Town Pool, and it was recorded in October 2023 at the Hotham River during a fish monitoring survey (SLR 2024). 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.

Based on the absence of permanent water or pools within the study area, it is considered there was a low likelihood that habitats within Gringer Creek support important breeding and feeding habitat for the regional population.

3.2.5.2 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 as conservation significant. It has a relatively widespread distribution throughout the south-west, from Hill River in the north to Esperance on the southern coast. 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.

Chelodina oblonga is endemic to the south-west of WA and is listed on the IUCN Redlist of Threatened Species as Near Threatened (IUCN 2023), although it is not listed at State or National level. This species is restricted to the south-west of Western Australia, between the Hill River in the north, Blackwood River in the south, and east to the Sussetta River (Cann 1998). 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).

4.0 Field Survey

4.1 Methods

4.1.1 Guidance and general approach

4.1.1.1 EPA Environmental Factor Guideline: *Inland Water*

The 2023 baseline aquatic ecosystem survey at Gringer Creek was 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).

4.1.1.2 Technical guidance

There are currently no prescriptive guidance statements at the state level outlining surface water quality and aquatic fauna sampling design, methods and general approaches / bioindicators. In alignment with current research however, field surveys conducted in 2023 were consistent with the following:

- 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);
- EPA Position Statement No. 3, Terrestrial Biological Surveys as an Element of Biodiversity Protection (EPA 2002);
- EPA Guidance No. 56, Terrestrial Fauna Surveys for Environmental Impact Assessment in Western Australia (EPA 2004).
- 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.

Aquatic fauna sampling methods were also similar to the following:

- Van Looj E, Storer T 2009. *Inception Report Volume 1*, Department of Water, Western Australia.
- Storer T, White G, Galvin L, O'Neill K, van Looj E, Kitsios A. 2010. *The Framework for the Assessment of River and Westland Health (FARWH) for flowing rivers of south-west Western Australia: method development, Final report.* Water Science Technical Series, report no. 40, Department of Water, Western Australia.

- Storer T, White G, O'Neill K, Galvin L, van Looj E. 2020. *South-West Index of River Condition, Method Overview 2020*, River Science Technical Series 1, *Healthy Rivers* program, Department of Water and Environmental Regulation, Perth.

Fauna sampling was conducted under DBCA Fauna Taking (Biological Assessment) licence BA27000899 and DPIRD Fisheries Exemption 251151923.

Surveys were undertaken between the 5th and 9th of September 2023.

4.1.2 Survey sites

A total of 11 sites were surveyed on Gringer Creek (Figure 6). Site selection aimed to revisit those used during original baseline surveys conducted in 2011 (WRM 2012a), and sites currently monitored by NBG (Table 2). Where possible, site nomenclature prioritised existing NBG naming conventions.

Sites were classified as either upstream reference sites (GRCK01-04), impact sites within the proposed RDA2 footprint (GRCK05-07) or downstream sites that would be potentially exposed to upstream effects (GRCK08, and new sites GringerDS09-11). NBG water monitoring point GRCK09 was not sampled as part of this survey, however long-term water quality monitoring data were incorporated in derivation of interim SSGVs (section 5.0).

Table 2. Locations of aquatic monitoring sites on Gringer Creek, including SLR previous baseline sites, and new sites on downstream reaches of Gringer Creek.

WRM site ID	Newmont Site ID	Final site ID	Reach	East_MGA94Z50	North_MGA94Z50
S01	GRCK01	GRCK01	Upstream	444735	6392961
S02	GRCK02	GRCK02	Upstream	445407	6392127
S03	GRCK03	GRCK03	Upstream	445586	6391018
S04	GRCK04	GRCK04	Upstream	445040	6390732
S05	GRCK05	GRCK05	Impact	445588	6387588
S06	GRCK06	GRCK06	Impact	446250	6386340
S07	GRCK07	GRCK07	Impact	446516	6386075
S08	GRCK08	GRCK08	Downstream	447982	6384738
GringerDS09	N/A (new site)	GringerDS09	Downstream	448898	6384119
GringerDS10	N/A (new site)	GringerDS10	Downstream	449193	638468
GringerDS11	N/A (new site)	GringerDS11	Downstream	449958	6382146

Field surveys included systematic sampling of water quality, macroinvertebrates, fish and crayfish, where water depth and open space (for fyke nets) allowed (Table 3).

Table 3. Summary of methods successfully used at each site.

Site ID	Reach	Water quality	Macro.	Box traps	Fyke nets
GRCK01	Upstream	✓	✓	✓	✓
GRCK02	Upstream	✓	✓	✓	✓
GRCK03	Upstream	✓	✓	✓	✓
GRCK04	Upstream	✓	✓	✓	✓
GRCK05	Impact	✓	✓	✓	Insufficient depth
GRCK06	Impact	✓	✓	✓	✓
GRCK07	Impact	✓	✓	✓	✓
GRCK08	Downstream	✓	✓	✓	✓
GringerDS09	Downstream	✓	✓	Too deep	Too deep
GringerDS10	Downstream	✓	✓	✓	✓
GringerDS11	Downstream	✓	✓	✓	Insufficient space

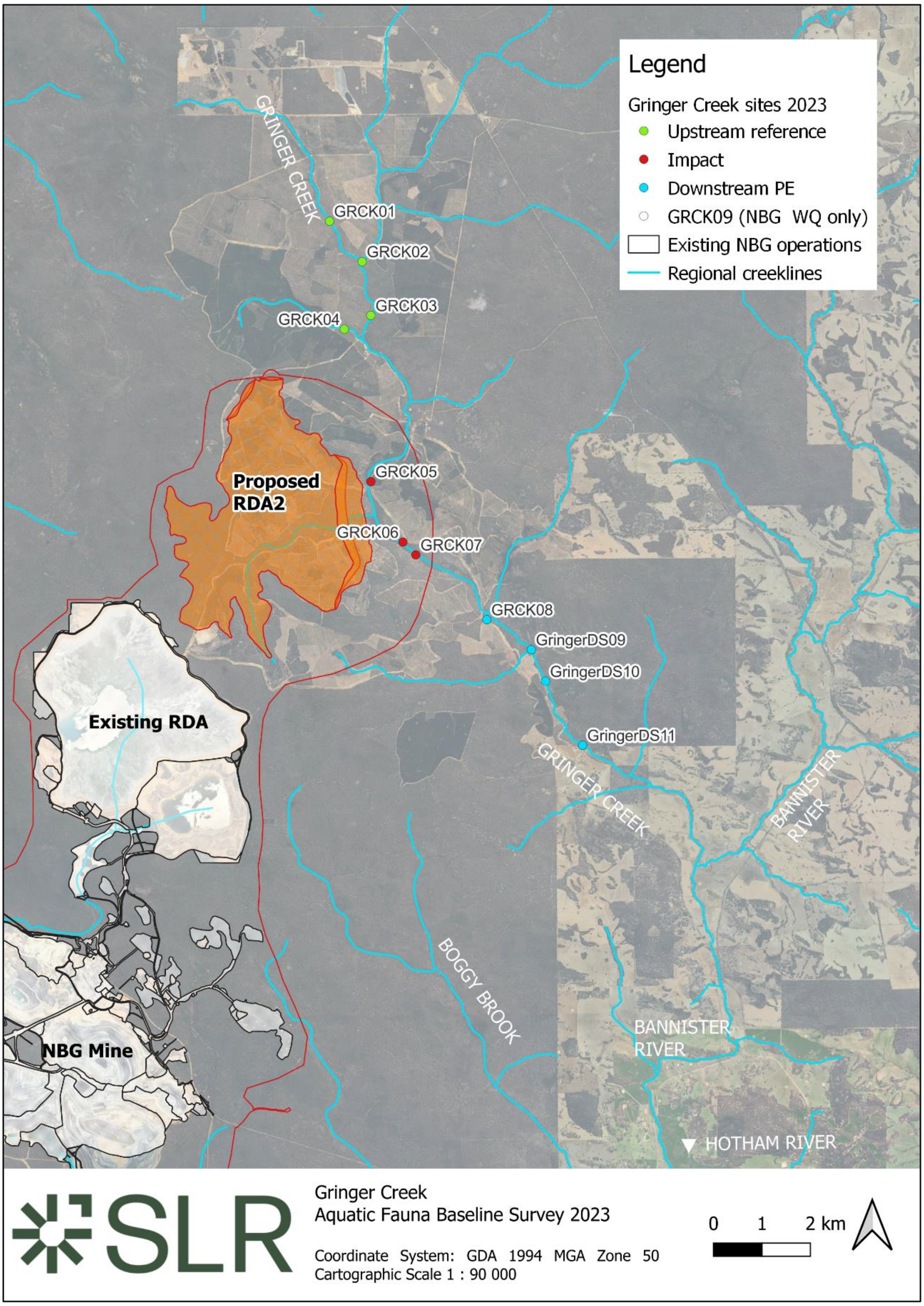


Figure 6. Baseline aquatic survey sites on Gringer Creek sampled in 2023, with indicative location of the RDA2 development in red. Sites GRCK01 – 08 were sampled as part of 2011 baseline surveys (WRM 2012a).

4.1.3 Field methods & data analysis

4.1.3.1 Water Quality

Water quality included *in situ* measurements of pH, conductivity, dissolved oxygen and temperature. At each site, undisturbed water samples were taken for laboratory analysis of hardness and alkalinity (as $CaCO_3$), total dissolved solids, total suspended solids, ions, nutrients (including total and dissolved forms) and dissolved metals. Field filtering for filtered samples was conducted through 0.45 µm Millipore filters. Water samples were immediately placed on ice in the field, and then refrigerated (general ions and filtered metals) or frozen (filtered nutrients) prior to transport to ChemCentre (a NATA-accredited laboratory).

Water quality data recorded in 2023 were compared to the ANZG (2018) default guideline values (DGV) for 99% and 95% species protection (Table 5). The 95% DGVs are recommended for slightly to moderately disturbed systems (ANZG 2018) and are thus appropriate for Gringer Creek given the legacy of catchment clearing and salinisation. The 99% DGVs (which are recommended for use in pristine-minimally disturbed systems) are also given for context.

4.1.3.2 Habitat characterisation

Qualitative data on aquatic habitat conditions at each site were collected using visual assessment. These data aid in the interpretation of aquatic fauna diversity and abundance. Habitat characteristics recorded included percent cover by inorganic sediment, submerged macrophyte, floating macrophyte, emergent macrophyte, algae, large woody debris, detritus, roots and trailing vegetation. Visual appraisal of substrate composition was also conducted, including percent cover by bedrock, boulders, cobbles, pebbles, gravel, sand, silt and clay. SLR have specific worksheets for this task so that recordings between sites remain as comparable as possible. General observations regarding the condition of site habitat and disturbance were also made, and site photographs taken (Site photographs are provided in Appendix 1).

4.1.3.3 Macroinvertebrates

One composite macroinvertebrate sample was collected from each site. Macroinvertebrates were sampled using a 250 µm mesh FBA d-frame dipnet. All habitats present (e.g. littoral areas, open channel, macrophyte, inundated riparian vegetation) were sampled, with the objective of maximising the number of species recorded by sampling across as many habitats as possible. All samples were preserved in the field in 100% ethanol for laboratory processing. In the laboratory, macroinvertebrates were removed from samples by sorting under microscopes. Specimens were then identified to the lowest taxonomic level (typically genus or species) and enumerated to $\log_{10}$ scale abundance classes (i.e. 1 = 1 - 10 individuals, 2 = 11 - 100 individuals, 3 = 101-1000 individuals, 4 = >1000).

Patterns in macroinvertebrate richness and assemblage composition was further explored using univariate and multivariate techniques. Spatial differences were compared between sites, and also temporal differences between 2011 and 2023 baseline surveys to elucidate potential spatio-temporal patterns, and also the comparability of the two datasets as a combined baseline for ongoing monitoring. Taxonomy was standardised between the two datasets prior to temporal analysis, allowing for advances in taxonomy since 2011.

All univariate analysis was conducted using TIBCO Statistica (v. 14), and multivariate analyses using Primer-E (v. 7).

4.1.3.4 Fish & Crayfish

Fish and crayfish were surveyed using a standardised catch per unit effort (CPUE) approach based on SWIRC (Storer et. al. 2020), using fyke nets and baited box traps deployed overnight, where water depth was sufficient, and (for fyke nets) there was sufficient open space. Box traps were placed partially out of the water, to protect any air-breathing bycatch (e.g. turtles) that may enter traps. All species of fish and crayfish, and any by-catch were identified in the field, measured by standard length[3] (SL mm, for fish) or carapace length (CL mm, for crayfish) and then released alive. Crayfish were sexed, and whilst it is not possible to determine sex in most native fish species, reproductive indicators including breeding colours or presence of gravid females were noted, where applicable. Fish nomenclature followed that of Allen et al. (2002).

It was not possible to set fyke nets at GRDS05 or GringerDS11 due to insufficient water depth and/or open space in the water channel (Table 2). In addition, fyke nets (which rely on blocking the channel) are not effective when water depth is too great (e.g. GringerDS09).

Abundance data for each species were tabulated according to site, and length-frequency histograms were plotted to determine approximate population structure for each species at Gringer Creek. Length was also used to group fish species into broad age-classes based on published information for southwest fish populations (see Table 4). Nightfish display sexual dimorphism in the rates at which they mature, meaning males and females of the same size can represent different age-classes (Pen & Potter 1990a). Therefore, assigning specific age-classes to nightfish is problematic without sexing specimens, and distinction is only made between ≤ 56mm SL (0-1 year) and >56 mm SL (1+ years). Length is not a reliable determinate of age for crayfish.

Table 4. Age-size classes of native fish species in southwest Western Australia, based on Pen & Potter (1990a, b, 1991), Gill et al. (1996) and Beatty et al. (2010).

Scientific name	Common name	Age class (year)	Size (TL mm)
Bostockia porosa	Nightfish	0+ - 1	≤ 56
		1+	>56
Galaxias occidentalis	Western minnow	0+ - 1	≤ 75
		2+	75 - 110
		3+	> 110
Nannoperca vittata	Western pygmy perch	0+ - 1	45
		2+	52
		3+	53
		4+	63
		5+	65

[3] Standard length (SL) = tip of the snout to the posterior end of the last vertebra (*i.e.* this measurement excludes the length of the caudal fin). Carapace length (CL) = anterior tip of the rostrum to the posterior median edge of the carapace.

4.2 Results

4.2.1 Water Quality

There were few analytes recorded above the 95% species protection DGVs recorded in September 2023, with no additional exceedances of the 99% DGVs. Of note, conductivity (salinity) was well above the 95% DGV at all sites (Table 5). However, values recorded are comparable to other creeks in the region affected by historic catchment clearing. Additionally, the DGV of 250 µS/cm may be considered conservative (Hart et al., 1991; Horrigan et al., 2005). Horrigan *et al.* (2005) reported that significant shifts in macroinvertebrate community composition can occur as salinity reaches 800 – 1000 µS/cm, although it is generally considered that biota of freshwater ecosystems will not be adversely affected until salinity reaches ~1,500 µS/cm (Hart 1991). Most south-west freshwater fish and crayfish can tolerate salinities up to 8,000 µS/cm (Morrissy 1978, Beatty *et al.* 2008, 2010).

Aside from conductivity, cobalt was recorded above the 95% DGV of 0.0014 mg/L at GRCK01-04, GRCK06 and GRCK07, to a maximum of 0.026 at GRCK02 (Table 5). Historically, Co has periodically been elevated at Gringer Creek, which is discussed further in section 5.3.2. Manganese was also recorded at high levels (by comparison to the 95% DGV) at GRDS02 (Table 5).

Table 5. Summary of water quality values recorded in September 2023 at Gringer Creek, compared to ANZG (2018) DGVs for 99% and 95% species protection (or ANZECC/ARMCANZ (2000) DGV applicable to southwest rivers). All units are mg/L unless indicated.

ANALYTE	95% DGV	99% DGV	*LOR*	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
LABORATORY														
Al	0.055	0.027	*0.005*	<0.005	<0.005	<0.005	<0.005	<0.005	0.039	<0.005	<0.005	<0.005	<0.005	<0.005
Alkalinity	-	-	*1*	31	21	25	5	23	15	17	16	15	16	16
As	0.024	0.0001	*0.00005*	0.00012	0.00009	0.00008	0.00011	0.00006	0.00009	0.00007	0.00009	0.00008	0.00008	0.00008
B	0.94	0.34	*0.02*	0.06	0.04	0.05	0.03	0.04	0.04	0.03	0.03	0.03	0.03	0.03
Ba	-	-	*0.002*	0.068	0.058	0.055	0.096	0.042	0.044	0.036	0.036	0.036	0.037	0.042
CO_3	-	-	*1*	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Ca	-	-	*0.1*	37.4	73.6	58.3	48.8	59.5	69.1	60.9	69.1	70.2	69.3	75.8
Cd[A]	0.0002	0.00006	*0.0001*	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Cl	-	-	*1*	1450	2060	1830	1780	1910	1990	1880	2030	2070	2020	2060
Co	0.0014	-	*0.0001*	0.0035	0.026	0.01	0.0099	0.0012	0.0041	0.001	0.0012	0.0008	0.0008	0.0008
Cr[A]	0.001	0.0001	*0.0005*	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005	<0.0005
Cu[B]	0.0014	0.001	*0.0001*	0.0004	0.0001	0.0002	0.0001	0.0002	0.0002	0.0002	0.0002	0.0002	0.0002	0.0002
Fe	-	-	*0.005*	0.94	1.8	0.31	3.5	0.078	0.048	0.027	0.034	0.03	0.068	0.074
HCO_3	-	-	*1*	38	25	30	6	28	19	20	20	19	19	19
Hardness	-	-	*1*	450	750	630	630	650	740	660	770	780	780	840
K	-	-	*0.1*	11.9	7.2	9.3	3.9	7.8	7.9	7.3	7.3	7.3	7.2	7.7
Mg	-	-	*0.1*	86.2	138	117	124	122	138	124	144	148	147	157
Mn	1.9	1.2	*0.001*	0.26	2.1	1	0.6	0.2	0.38	0.1	0.11	0.073	0.081	0.087
Mo	0.034	-	*0.001*	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
N_NH_3[B]	0.9	0.32	*0.01*	<0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
N_NO_2	-	-	*0.01*	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
N_NO_3[C]	2.4	1.0	*0.01*	0.06	0.01	0.04	<0.01	0.02	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
N_NOx	0.2	-	*0.01*	0.06	0.01	0.04	<0.01	0.02	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
N_total	1.2	-	*0.01*	0.3	0.21	0.27	0.07	0.18	0.11	0.11	0.15	0.17	0.09	0.12
Na	-	-	*0.1*	748	1020	920	821	923	954	882	947	953	955	988
Ni[A]	0.011	0.008	*0.001*	0.001	0.004	0.002	0.002	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
P_total	0.065	-	*0.005*	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005	<0.005
Pb	0.0034	0.001	*0.0001*	<0.0001	<0.0001	<0.0001	0.0002	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
S	-	-	*0.1*	38	52	48	29	47	47	45	46	48	46	44
SO4_S	-	-	*0.1*	115	156	145	86.9	142	140	134	139	142	138	133
Se	0.011	0.005	*0.001*	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Si	-	-	*0.05*	2.7	5.1	3.9	5.3	4.2	3.6	3.6	3.8	3.8	4	4
Sr	-	-	*0.002*	0.36	0.55	0.46	0.49	0.48	0.59	0.51	0.58	0.59	0.59	0.64
TDS_calc	-	-	*5*	2500	3500	3200	3000	3300	3400	3200	3400	3500	3500	3500
TSS	-	-	*1*	4	3	3	2	1	1	1	6	1	<1	1
U	0.0005	-	*0.0001*	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
V	0.006	-	*0.0001*	0.0002	0.0002	0.0001	0.0002	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001

ANALYTE	95% DGV	99% DGV	*LOR*	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
Zn[A]	0.008	0.0024	*0.001*	<0.001	0.001	<0.001	0.002	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
IN SITU														
pH (H^+)	6.5 - 8.0	-		7.02	6.96	7.24	7.35	7.19	7.27	7.33	7.23	7.41	7.1	7.03
DO%	80 - 120	-		103.1	101.4	102.5	100.6	81.7	95.5	108.1	111	102.1	103.4	97.6
DO (ppm)	-	-		10.96	8.55	6.62	9.17	6.77	7.51	9.42	9.68	10.77	9	8.61
Turbidity (NTU)	10 - 20	-		1.76	2.32	2.77	2.19	0.05	n.p.	0.12	0.001	0.001	0.001	0.001
Conductivity (µS/cm)	120 - 300	-		4300	5870	5400	5210	5560	5820	4890	5190	5830	5960	5880

A – DGV provided here applies to hardness at the default 30 mg/L $CaCO_3$. Where analytes exceed DGV, a hardness modified trigger value should be applied (Warne et al., 2018).

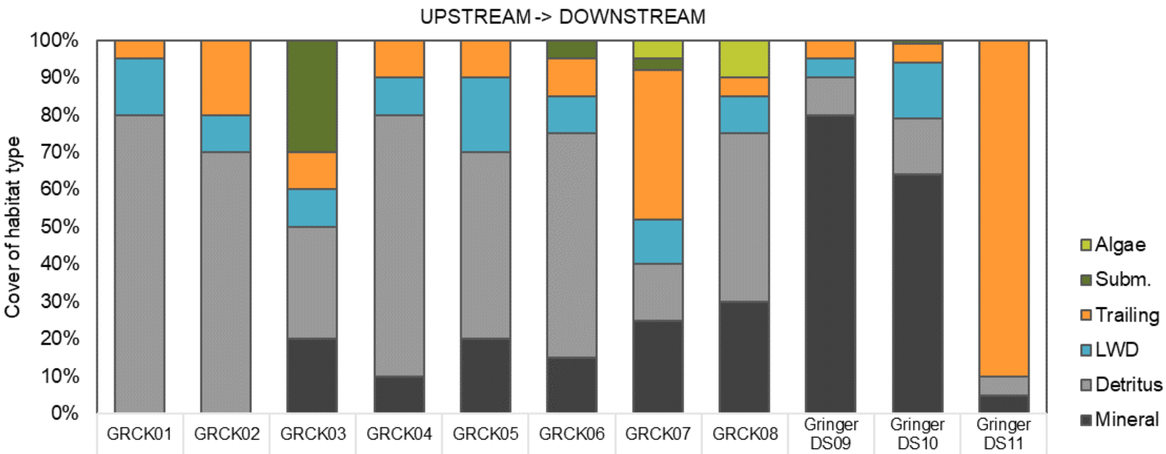
B – Draft guidance for Cu and NH_3-N are expected to be published in future. The current DGV is applied here.

C – the ANZECC/ARMCANZ (2000) toxicity DGV for nitrate is erroneous. ANZG recommend use of updated guidance for New Zealand (Hickey et al., 2013) *in lieu* of Australian guidelines.

4.2.2 Habitat Characteristics

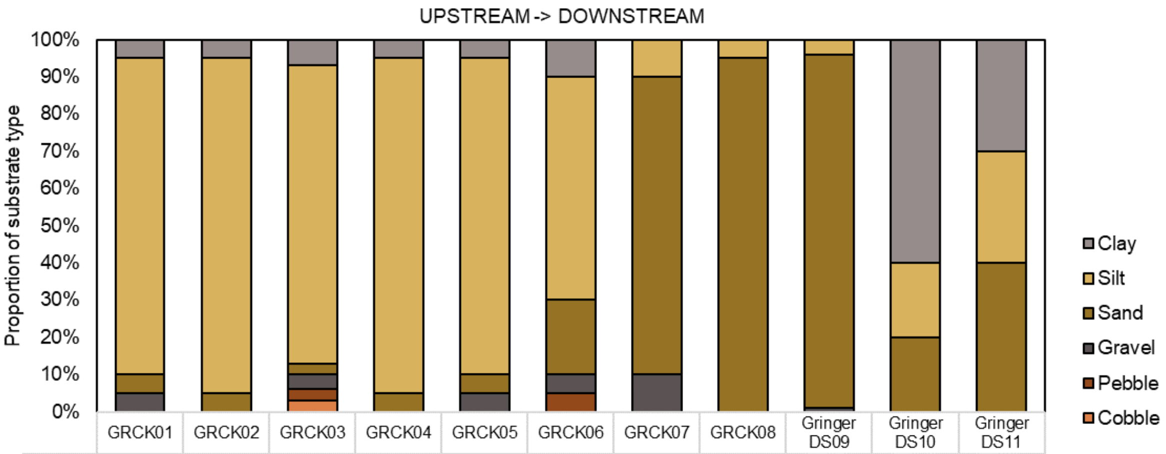
Gringer Creek traverses silviculture plantations along much of its length within the project area, and thus has a variable width of buffering and riparian zone native vegetation. Most sites sampled had moderately dense riparian fringing vegetation, particularly *Melaleuca* sp. shrubs which formed a closed canopy in places, which is typical of good riparian condition in jarrah/marri forest headwater streams (Appendix 1, 2). A high proportion of instream habitats reflected riparian composition, with detritus, woody debris (LWD), and trailing vegetation dominant at most sites, providing structurally complex habitats for fish and macroinvertebrates. Several sites also supported submerged vegetation including water ribbons *Triclochin* sp. Algal cover was scarce, as would be expected for shaded, headwater streams. There was an approximate gradient in habitat composition between upper reaches, dominated by detritus and woody debris, to the lower reaches where the creek was wider, had more open space (or 'mineral substrate') and the riparian zone supported more trailing streamside vegetation (Figure 7). Most sites were dominated by sand or silt substrates, except for GringerDS11 which was dominated by clay (Appendix 2). Habitat diversity was moderate to high, with all sites supporting three or more of the eight broad habitat types used to characterise aquatic ecosystems, to a maximum of six identified habitat types at GRCK07.

Similarly to habitat composition, an upstream to downstream gradient in substrate composition was identified, with the upper reaches dominated by silty substrates (GRCK01 – 05; Figure 8), gradually shifting between GRDS06 to GRDS07 to sand dominated substrates (GRCK07 – Gringer DS09). The lowest reaches comprised a variation of clay > silt > sand substrate (GringerDS10 & GringerDS11; Figure 8).



	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
Algae	0	0	0	0	0	0	5	10	0	0	0
Subm.	0	0	30	0	0	5	3	0	0	1	0
Trailing	5	20	10	10	10	10	40	5	5	5	90
LWD	15	10	10	10	20	10	12	10	5	15	0
Detritus	80	70	30	70	50	60	15	45	10	15	5
Mineral	0	0	20	10	20	15	25	30	80	64	5

Figure 7. Composition of in-stream habitats along a longitudinal gradient in Gringer Creek.



	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
Clay	5	5	7	5	5	10	0	0	0	60	30
Silt	85	90	80	90	85	60	10	5	4	20	30
Sand	5	5	3	5	5	20	80	95	95	20	40
Gravel	5	0	4	0	5	5	10	0	1	0	0
Pebble	0	0	3	0	0	5	0	0	0	0	0
Cobble	0	0	3	0	0	0	0	0	0	0	0

Figure 8. Proportion of substrate composition along a longitudinal gradient in Gringer Creek.

4.2.3 Macroinvertebrates

4.2.3.1 Taxa richness and assemblage composition

A total of 72 macroinvertebrate taxa were recorded from Gringer Creek in 2023 (69 excluding microcrustaceans; Appendix 3). This includes a number of taxa that could not be identified to species level due to incomplete taxonomic information (e.g. most Diptera families, some Coleoptera) or immature specimens, therefore species richness per site is likely higher than reported. In addition, three groups of microcrustaceans were present in the samples, however as sampling did not specifically target microcrustacea, and their identifications require external taxonomic expertise, an accurate count of taxa is not included here.

The macroinvertebrate assemblage was dominated by insects and their larvae, with larval Diptera (true flies) the most diverse group with 31 taxa, of which 17 were larval Chironomidae (non-biting midges), and 11 Coleoptera (aquatic beetles; Table 6). Common taxa included the salt tolerant amphipod *Austrochiltonia subtenuis* at GRDS05-GringerDS11, the chironomid *Tanytarsus* sp., Ceratopogoninae sp. (biting midge) larvae and Scirtidae beetle larvae. The highest abundance per site was recorded at GRCK06 and GringerDS11 (30 taxa each), and the lowest abundance was recorded at GRDS04 (Figure 9). A species accumulation curve suggested that further sampling may continue to identify additional taxa, however appeared to be nearing asymptote at 10 to 11 samples (Figure 10). A suite of common taxa were identified across samples, thus any new species would likely be collected as singletons.

Table 6. Summary of key taxonomic groups recorded as present in the 2023 surveys. *Sampling did not target microcrustacean specifically, therefore only very high level identifications are given.

Taxonomic group	Common name	*n* taxa
Oligochaeta & Nematoda	Worms	2
Gastropoda	Aquatic snails	2
Amphipoda	Side-swimmers	1
Acarina	Water mites	6
Collembola	Springtails	3
Coleoptera	Aquatic beetles	11
Diptera	True-flies	31
(Chironomidae)	Midges	17
Hemiptera	True-bugs	2
Odonata	Dragonflies & damselflies	4
Plectoptera	Stoneflies	1
Trichoptera	Caddisflies	5
Total		**68**
*Microcrustacea	Ostracoda, Cladocera, Copepoda	3

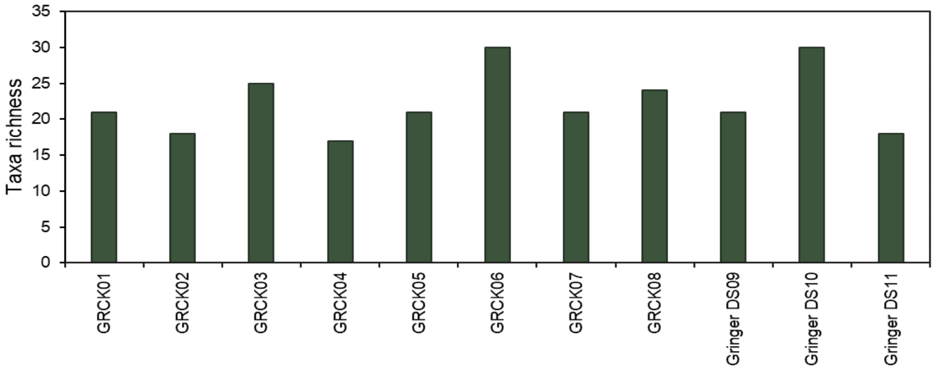


Figure 9. Summary of macroinvertebrate taxa richness per site recorded in 2023.

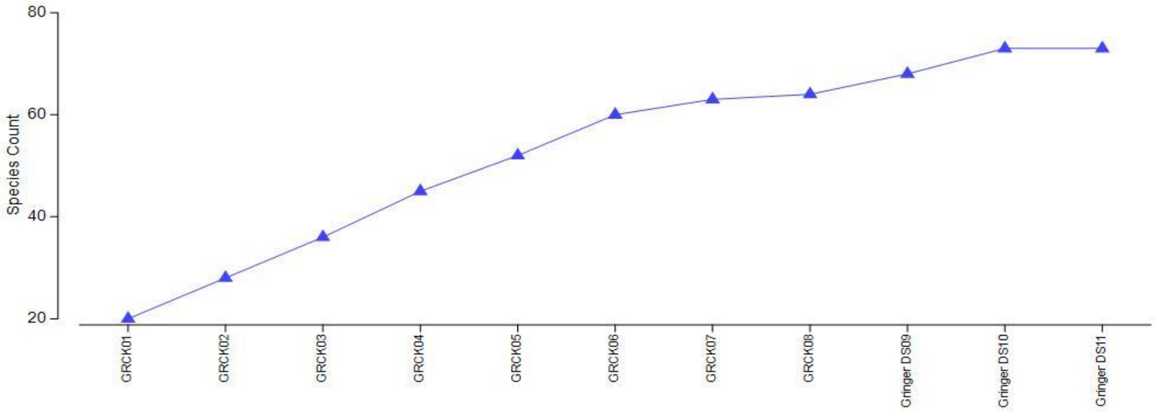


Figure 10.Species accumulation curve for Gringer Creek in 2023, showing accumulation of 'new' taxa per sample.

Ordination of macroinvertebrate assemblages recorded in 2023 found a clear separation of the upstream sites GRCK01 – 03 from the remaining sites, which did not differ (Figure 11). Similarity profiles (SIMPROF) identified two groups at a correlation of rho = 0.82; which were significantly different when permuted using Analysis of Similarity (On-way ANOSIM; Global R = 0.81, p = 0.006 on 165 permutations). Key taxa contributing to differences between groups included:

- Beetle *Sternopriscus marginatus*, chironomid *Paratanytarsus* sp., caddisfly *Oecetis* sp. and immature dragonfly nymphs, high frequency of occurrence at GRCK01 – 03, and rare or absent downstream of GRCK03;
- Larval Scirtidae beetles, endemic chironomid *Paralimnophyes pullulus* and amphipod *Austrochiltonia subtenuis* were frequent below GRCK04 and absent from sites GRK01 – 03.

Differences in habitat structure and hydrology may be contributing to differences in assemblage composition between the two sections of Gringer Creek. For example, differences in groundwater connectivity between upstream and downstream reaches may be influencing distribution of *Austrochiltonia subtenuis*, which is usually associated with groundwater.

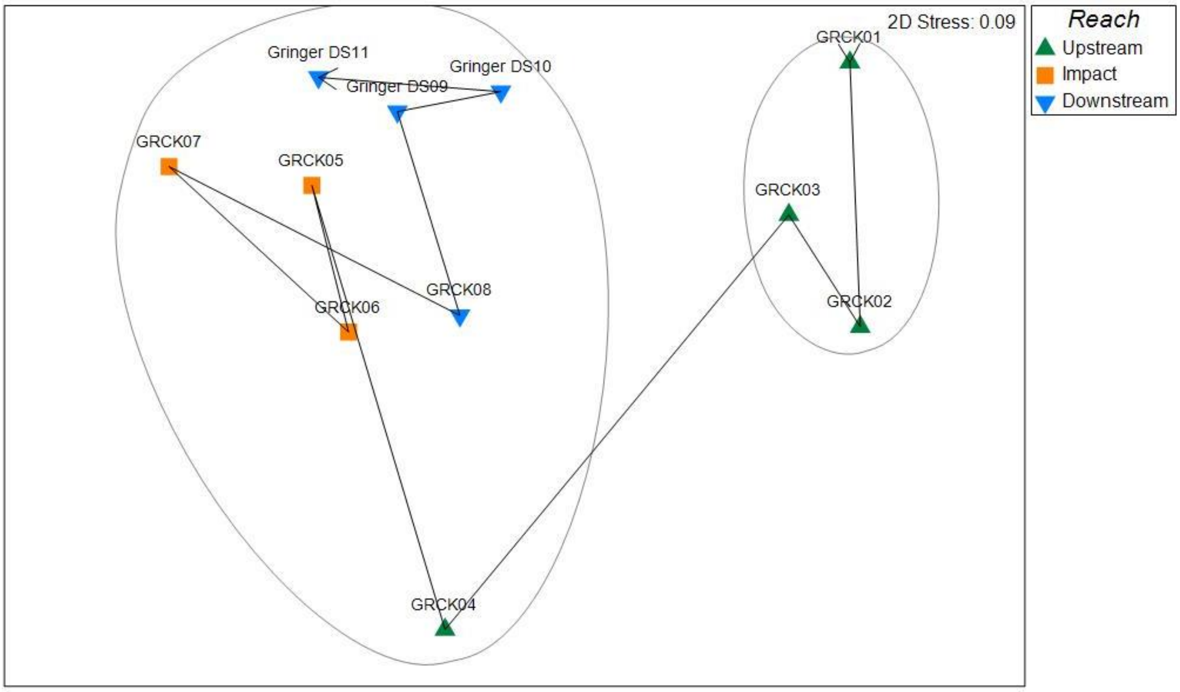


Figure 11. nMDS ordination of macroinvertebrate assemblage composition recorded at Gringer Creek in 2023. Trajectory overlay indicates site position of site in relation to unidirectional flow. SIMPROF overlay indicates grouping at a cophenetic correlation of 0.82.

4.2.3.2 Conservation significance

No formally listed macroinvertebrate taxa were recorded from Gringer Creek in 2023. However, there were a number of south-west endemic species which are of conservation and/or scientific interest, summarised below.

The damselfly *Archiargiolestes* sp., is a southwest endemic of ancient Gondwanan origin, and was found at several sites. Taxonomic revision of *Argiolestes pusillus* (recorded in 2011; WRM 2012a) found the taxon to comprise a complex of three species belonging to a genus endemic to south-west WA, including *Archiargiolestes pusillus*, *A. pusillissimus* and *A. parvulus* (Keller & Theischinger 2013). The latter two species have been identified by DBCA as potential priority species of concern, as they have not been observed since the 1970s (Pennifold 2018), however have not yet appeared on formal conservation lists. The IUCN list *A. pusillissimus* and *A. parvulus* as Near Threatened, and *A. pusillus* as Least Concern. The *Archiargiolestes* sp. recorded from Gringer Creek is evidently tolerant of moderate levels of salinity. The SWWA endemic dragonfly *Austroaeschna anacantha* was recorded (as a singleton) at GRCK03. *A. anacantha* was formerly the most common dragonfly in the Northern Jarrah forest (Bunn 1988).

Several endemic beetles were also recorded, including *Megaporus solidus* (previously *Macroporus solidus*), *Sternopriscus browni* and *S. marginatus*, and *Haliplus fuscatus/gibbus* (both endemic species that are inseparable based on external features; Watts & Hamon 2015). In addition, a number of undescribed Orthocladiinae chironomids were present. Many SWA Orthocladiinae are yet to be described, and many are endemic to the region (Leung et al., 2011). In addition, there were taxa that are Australian endemic, such as the dragonfly *Anax papuesis*, and the corixid (backswimmer) *Diaprepocoris barycephalus* which has a southern ‘Gondwanan’ distribution.

Several endemic insect taxa recorded belong to the orders Plecoptera and Trichoptera, which alongside Ephemeroptera (or ‘EPT’) taxa are among the most sensitive freshwater invertebrates, and are a widely used as a bioindicator of ecosystem health (Marchant et al*,.* 1995; Marshall et al*,.* 2001; Walsh 2006). Their presence is usually associated with good environmental condition and low anthropogenic disturbance, and they are often absent in degraded systems. GRCK07 had the highest number of EPT taxa recorded (Figure 12). The stonefly *Leptoperla australica* is able to breed in intermittent streams, and has desiccation resistant eggs (Carey et al., 2021). In perennial systems, they undergo synchronised emergence in April (Bunn 1988), however in streams that have become intermittent they are able to switch their life histories to develop and emerge in winter and early spring, depositing desiccation resistant eggs in the sediment. Five Trichoptera were recorded at Gringer Creek. The vast majority of Trichoptera in SWA are locally endemic, estimated at over 75% of total species (Cartwright et al., 2012), thus taxa identifiable to genus level only (*Oecetis* sp., *Oxyethira* sp.) are likely to be endemic. The stick-caddis *Triplectides* sp. AV21 is distributed throughout forested catchments of the Darling Scarp and southern jarrah forests (St Clair 2002). There were however no Ephemeroptera recorded during the survey, suggesting salinity (most likely) is in above of this group’s tolerance thresholds.

Altogether, the macroinvertebrate community of Gringer Creek has considerable ecological value, including a moderately diverse assemblage of taxa, of which at least a quarter are known SWA endemic species (more are likely to be endemic; Figure 13), including species from sensitive Plecoptera and Trichoptera groups.

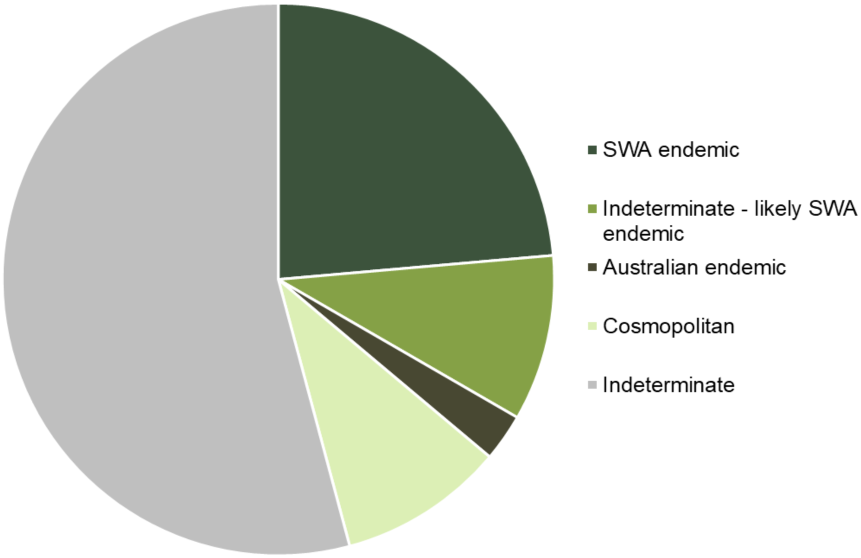


Figure 12. Composition of macroinvertebrates according to distribution, including south-west (SWA) endemic species, Australian endemic and cosmopolitan/likely cosmopolitan (i.e. distribution beyond Australia) taxa. Many taxa were classified as indeterminate due to higher level identification (i.e. immature specimens), however some are likely to be SWA endemic due to known distributions (e.g. SWA Trichoptera are almost entirely endemic; Cartwright et al., 2012).

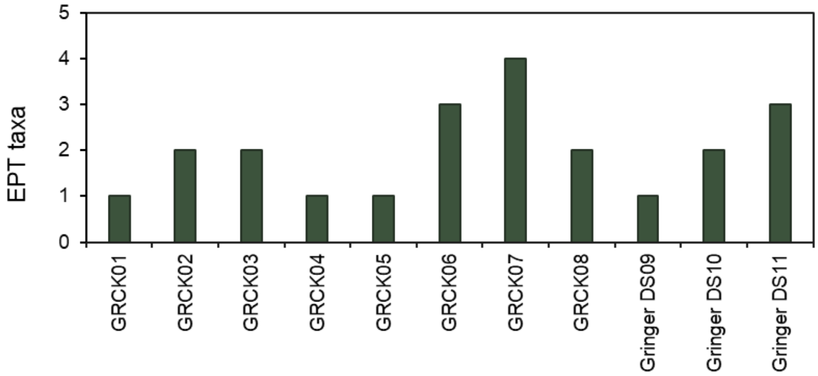


Figure 13. Richness of EPT taxa (Ephemeroptera, Plecoptera and Trichoptera) at Gringer Creek sites in 2023.

4.2.3.3 Temporal comparison

Mean taxa richness in 2023 (17.25) was significantly greater than that recorded in 2011 (13.1) (t_{14} = -1.8, p = 0.046; Figure 14). Ordination shows clear separation of assemblage composition between years, which was statistically significant (One-way ANOSIM Global R = 0.87, p = 0.001; Figure 15). Because taxonomy was standardised between the species lists prior to analysis, it is likely that these represent real differences between surveys rather than improved taxonomic resolution. Importantly, the 2011 survey was conducted later in spring, which in an ephemeral system may be sufficient time difference to cue life history stages (e.g. emergence of adult insects from aquatic nymphs). This is evident in the differences in assemblage composition, with entire groups absent in 2011 that were present in 2023, including several insects with winged adults (e.g. caddisflies, stoneflies) whereas taxa in common included obligate swimmers *Austrochiltonia subtenuis*, and a number of chironomid taxa. Alternatively, differences in hydroperiod between years may have resulted in lower richness in 2011.

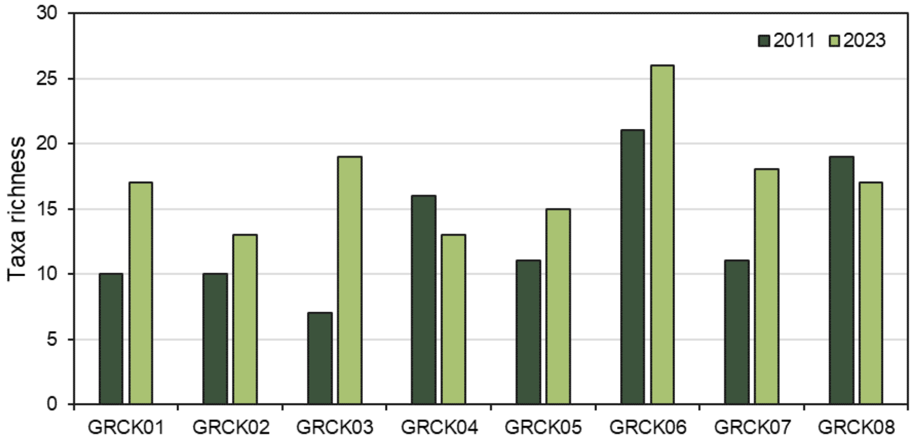


Figure 14. Summary of macroinvertebrate taxa richness recorded between 2011 and 2023 from sites in common.

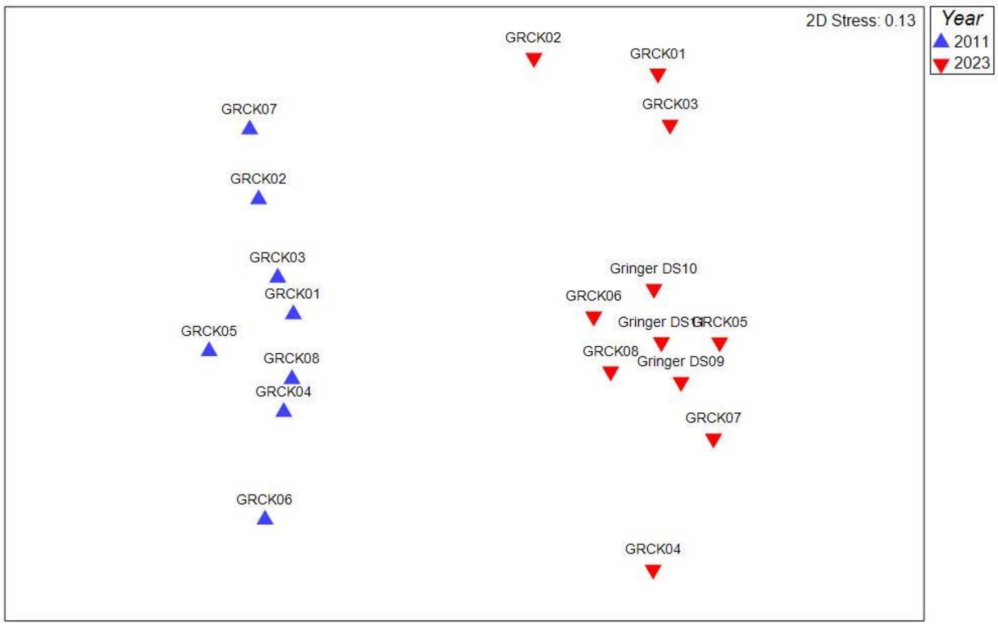


Figure 15. nMDS ordination of standardised macroinvertebrate taxa richness recorded at sites on Gringer Creek in 2011 and 2023.

4.2.4 Fish

4.2.4.1 Species composition and conservation significance

A total of 275 fish of three species were captured and released during this survey**Error! Reference source not found.**. No fish were recorded at GRCK05, GringerDS09 and

GringerDS11 (owing to inability to set fyke nets, and no capture in box traps), and no fish were captured at GRCK07. The greatest abundance of fish was recorded at GRCK02 (Table 7; Figure 16). All three species are SWA endemic, including the nightfish *Bostockia porosa*, western minnow *Galaxias occidentalis* and the western pygmy perch *Nannoperca vittata*. All three species were recorded in the 2011 survey, and there have been no apparent losses of native fish species since the first baseline survey (WRM 2012b). There were no introduced fish detected during the current survey (e.g. the mosquitofish *Gambusia holbrooki*, which was detected in 2011; WRM 2012b).

No species recorded are considered rare or restricted in distribution. The western minnow, western pygmy perch, and nightfish are the most common native fish in the southwest, occurring across a wide range of habitats. Populations of all three are present in the nearby 34 Mile Brook and the Hotham River (WRM 2011; WRM 2012a,b; WRM 2022). Western minnows are typically found in schools in both open water and enclosed areas amongst riparian vegetation. Western pygmy perch generally prefer waterbodies with overhanging riparian and/or emergent vegetation and rarely remain in open water. In contrast to western minnows and western pygmy perch, the nightfish is a solitary, bottom dwelling fish, more active at night. During the day, it typically shelters under ledges, rocks, logs or amongst root mats and inundated vegetation (Thorburn 1999).

Although all three species are freshwater species, they are all somewhat tolerant of salinity. Acute salinity tolerance trials by Beatty *et al*. (2008) indicate western minnows can tolerate salinities up to ~14,000 mg/L (equivalent to ~25,000 µS/cm). Morgan *et al.* (2003) reported that the acute salinity tolerance of western pygmy perch is similar to that of western minnows, but pygmy perch appear to prefer habitats with salinities of <~5,000 mg/L (equivalent to ~10,000 µS/cm). Thorburn (1999) reported that nightfish also appear to have a preference for lower salinity water (<1,000 µS/cm). However, recent work by Beatty *et al*. (2010) on the Brockman River, suggests nightfish can tolerate and possibly breed in salinities of at least ~8,000 µS/cm.

There were clear spatial patterns evident in abundances of nightfish and pygmy perch between the upper and lower reaches. Western pygmy perch were primarily detected in sites GRCK01 to GRCK03, with a maximum abundance of 47 at GRCK02. Nightfish were also more frequent in upper reaches than lower. Known patterns of migration in these species may explain this spatial differentiation, as these species are known to migrate upstream into upper reaches of tributaries through winter and spring to spawn, followed by downstream migrations of adults and juveniles in late spring as systems dry. Western minnows however did not show clear spatial patterning between the upper and lower reaches. Western minnow and nightfish are known to undertake their upstream migrations earlier in the year (beginning early-mid winter) than do pygmy perch (beginning early spring), which is thought to minimise inter-specific competition (Pen & Potter 1990b; Pen & Potter 1991).

Table 7. Summary of fish species recorded at Gringer Creek in 2023.

Section	Site	Nightfish	W. minnow	W. pygmy perch	Total
Upstream Reference	GRCK01	1	7	3	**11**
	GRCK02	7	38	47	**92**
	GRCK03	4	32	17	**53**
Impact	GRCK04	0	11	0	**11**
	GRCK06	1	34	0	**35**
	GRCK07	0	0	0	**0**
Potentially exposed	GRCK08	1	52	0	**53**
	GringerDS10	0	19	1	**20**
	Total	14	193	68	**275**

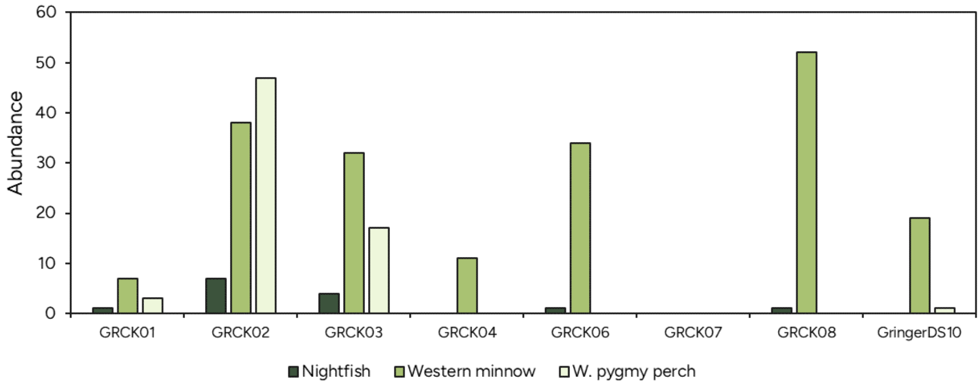


Figure 16. Abundance of native fish species per site.

4.2.4.2 Population demographics

A range of size classes of all three fish species were recorded, and size class distribution was as expected for healthy populations of native fish (Figure 177).

Several gravid females of western minnow and nightfish were recorded in the upper reaches of Gringer Creek during the survey (Figure 18, Figure 19), indicating these reaches are spawning grounds for these species in the subcatchment. As above, this concurs with understood breeding phenology of these species, which are known to migrate into tributary creeks during early to late winter, followed by downstream migration of adults and juveniles back to riverine environments during spring (i.e. before these creek begin to dry) (Pen & Potter 1990a, b). Therefore connectivity to the upper creekline is an important environmental value for western minnow and nightfish in the local region, and connectivity should be maintained between lower and upper reaches during and after any potential construction for the RDA2. Native fish populations of the Bannister River, and potentially the Hotham River, are likely using Gringer Creek as spawning habitat. Previous baseline surveys also noted a high proportion of juvenile native fish, supporting the same conclusion (WRM 2012b). Therefore, importance should be placed on maintaining fish passage as an aquatic environmental value, during the RDA2 construction and operation phases. Loss of connectivity may reduce available breeding habitat and may result in reduced breeding success, and ultimately declines in populations in the main river.

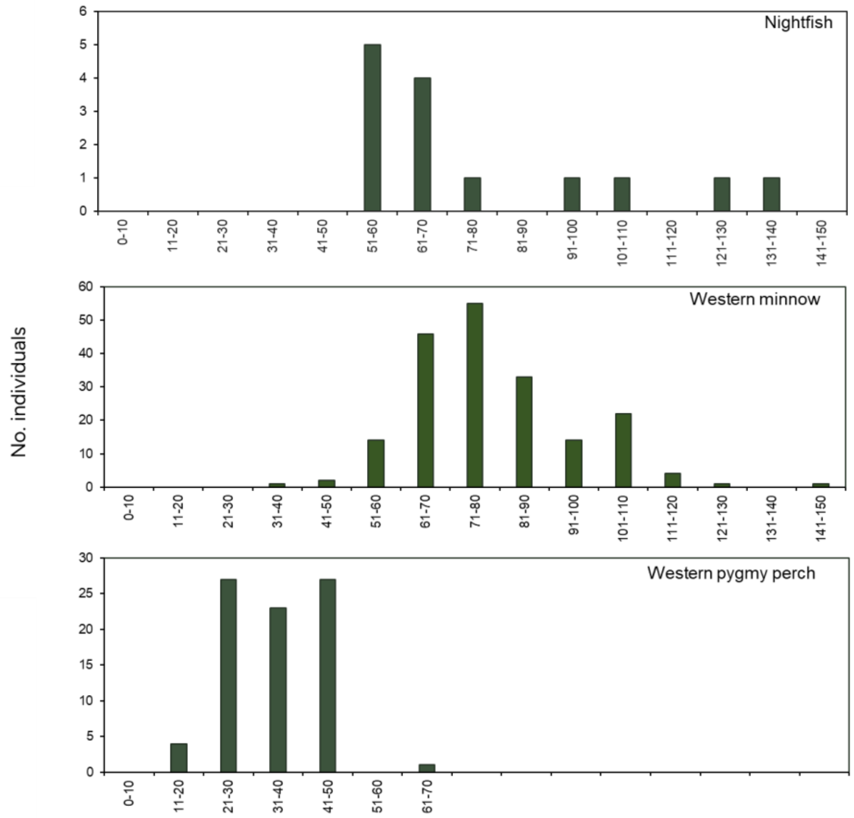


Figure 17. Length-frequency (SL mm) plots for fish species recorded in Gringer Creek 2023.

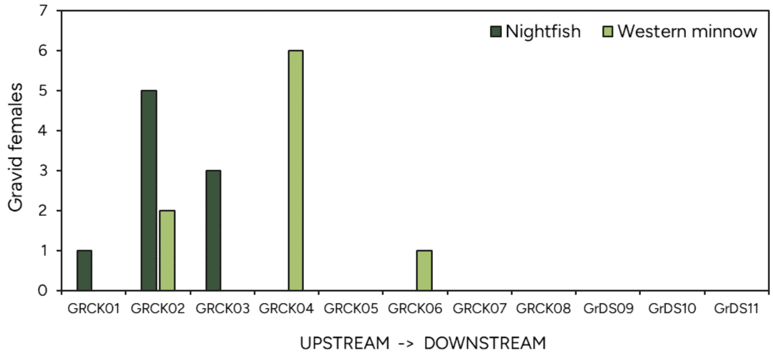


Figure 18. Distribution of gravid female nightfish and western minnows recorded at Gringer Creek.

Figure 19. Gravid western minnow captured at GRCK04.

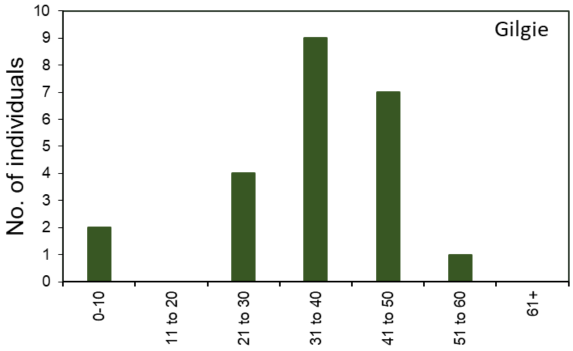
4.2.5 Crayfish

A total of 268 crayfish were captured and released (Table 8). Two native species were recorded, the Gilgie *Cherax quinquecarinatus* and Koonac *Cherax preissii*, as well as unidentified juveniles, and no introduced species were detected (e.g. the yabby *Cherax destructor*, detected in the 2011 survey; WRM 2012b). Marron were caught at GRCK07 and GRCK08 during the 2011 baseline surveys, with one individual per site. It is likely that population density was very low, and it is not possible to determine whether marron were present in the system in 2023. However, it is worth noting that unlike koonacs and gilgies (which can burrow), marron do not have strategies to survive drying in intermittent streams, and therefore rely on permanent pools to survive summer drying (Austin & Knott 1996). If pools completely dry over summer (which is possible in dry years) then potential exists for local extirpation of marron.

Both gilgies and koonacs are common and abundant in the habitats where they reside, and neither are conservation listed. Gilgies have a range that extends from Moore River in the north to Bunbury in the south of WA (Shipway 1951). They are known to exploit almost the full range of freshwater environments and can be found in habitats that range from seasonal swamps to deep rivers (Austin & Knott 1996). Gilgies have a well-developed burrowing ability that allows them to retreat in burrows during periods of low water level, and thereby survive seasonal waterbodies, so long as the burrows remain permanently moist. Gilgies are unlikely to have the same migratory capabilities of larger crayfish such as marron and may not be able to readily recolonise if local populations are lost during periods of prolonged drought. Koonacs occupy a wide-range of freshwater habitats but are most prevalent in seasonally inundated wetlands and other aquatic environments where water tables fluctuate (Beatty et al., 2006; Morgan et al., 2011). Koonacs are well adapted to seasonally dry systems, constructing deep burrows to stay moist over summer. Koonacs are known to mate and spawn in their burrows, returning to the surface following winter inundation to feed and grow (Beatty et al., 2006).

Table 8. Summary of crayfish recorded across Gringer Creek sites in 2023.

Section	Site	Gilgie	Koonac	Juv.	Total
Upstream Reference	GRCK01	2	8	-	**10**
	GRCK02	4	-	-	**4**
	GRCK03	15	5	2	**22**
	GRCK04	-	13	3	**16**
Impact	GRCK05	-	10	-	**10**
	GRCK06	-	79	-	**79**
	GRCK07	-	5	1	**6**
Potentially exposed	GRCK08	-	24	4	**28**
	GringerDS10	-	88	4	**92**
	GringerDS11	-	1	-	**1**



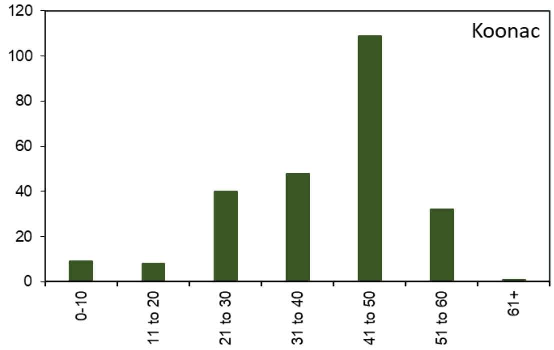


Figure 20. Length frequency distribution (CL mm) of crayfish species gilgie (GRDS01 – 03) and koonacs (all sites) recorded in spring 2023. NOTE: individuals under 10mm were identified to genus level only, thus for sites where gilgies were present it was not possible to confirm species level identity. All juveniles at GRDS04-GringerDS11 are assumed to be koonacs.

Figure 21. Koonacs *Cherax preisii* recorded during the 2023 surveys at Gringer Creek. Photos by Simon Ong (SLR Consulting).

5.0 Temporal water quality trends and interim SSGVs

5.1 Data compilation and methods

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 are persistently elevated above 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.

For the purposes of this analysis, each toxicant or stressor with sufficient data available were assessed based on rate of 95% DGV exceedances occurring as ‘background’ levels. The 95% DGVs are recommended for use in slightly to moderately disturbed ecosystems, and are thus appropriate for Gringer Creek given catchment clearing for ongoing agricultural and silvicultural land uses, and associated legacy impacts including secondary salinisation. In addition, each analyte was also compared to the more conservative 99% DGV for context.

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 Gringer Creek for physico-chemical stressors and toxicants were supplied by NBG (2012 to 2023) and baseline WQ data from WRM/SLR (2011, 2023) were used to calculate summary statistics including:

- Median, 20th and 80th percentiles,
- Minimum and maximum analyte concentrations,
- Proportion of data that is above the 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.

Prior to analysis, all data were screened for appropriateness for inclusion in interim SSGV calculations. Data were either gathered in field (e.g. physical attributes DO, pH), or supplied from one of three NATA accredited laboratories: ALS or MPL (NBG data), or ChemCentre (WRM/SLR), over a twelve year period. This introduced error in some datasets, for example differences in laboratory limits of detection/limits of reporting (LORs) between vendors. Criteria for exclusion included: laboratory limits of detection/ limits of reporting (LORs) too high to be comparable to ANZG DGVs, erroneous values or units, malfunctioning equipment, or data derived from ‘reliable estimates’, and removal of duplicates. A summary of availability of cleansed data is provided in Table 9.

Table 9. Summary of water quality data from Gringer Creek, valid for use in calculation of SSGVs. Including data ranges and total number of samples (*n*) from NBG and WRM/SLR collections.

Analyte	NBG			WRM/SLR		
	Start	**End**	***n***	**Start**	**End**	***n***
Al	31-08-2012	03-07-2023	*359*	31-10-2011	08-09-2023	*19*
Alkalinity	22-07-2013	03-07-2023	*353*	31-10-2011	08-09-2023	*19*
As (total)	22-07-2013	03-07-2023	*396*	05-09-2023	08-09-2023	*11*
B	20-07-2016	03-07-2023	*328*	31-10-2011	08-09-2023	*19*
Ba	20-07-2016	03-07-2023	*328*	31-10-2011	08-09-2023	*19*
Cd	22-07-2013	03-07-2023	*353*	31-10-2011	08-09-2023	*19*
Co	31-08-2012	03-07-2023	*368*	31-10-2011	08-09-2023	*19*
Cr (IV)	20-07-2016	03-07-2023	*328*	31-10-2011	08-09-2023	*19*
Cu^	22-07-2013	03-07-2023	*73*	31-10-2011	08-09-2023	*19*
Cyanide (free)	-	-	-	-	-	-
DO	31-08-2012	05-08-2023	*416*	31-10-2011	08-09-2023	*19*
EC	31-08-2012	05-08-2023	*414*	31-10-2011	08-09-2023	*19*
Fe	22-07-2013	03-07-2023	*353*	31-10-2011	08-09-2023	*19*
Hardness	22-07-2013	22-07-2013	*7*	31-10-2011	08-09-2023	*19*
Mn	31-08-2012	03-07-2023	*400*	31-10-2011	08-09-2023	*19*
Mo	22-07-2013	03-07-2023	*353*	31-10-2011	08-09-2023	*19*
N-NH_3	22-07-2013	03-07-2023	*706*	31-10-2011	08-09-2023	*19*
N-NO_3	22-07-2013	03-07-2023	*350*	05-09-2023	08-09-2023	*11*
N-NO_X	20-07-2016	03-07-2023	*295*	31-10-2011	08-09-2023	*19*
N Total	22-07-2013	03-07-2023	*354*	31-10-2011	08-09-2023	*19*
Ni	22-07-2013	03-07-2023	*353*	31-10-2011	08-09-2023	*19*
P Total	23-10-2019	03-07-2023	*278*	31-10-2011	08-09-2023	*19*
pH	31-08-2012	05-08-2023	*412*	31-10-2011	08-09-2023	*19*
Pb	22-07-2013	03-07-2023	*353*	31-10-2011	08-09-2023	*19*
S-SO_4	31-08-2012	03-07-2023	*400*	31-10-2011	08-09-2023	*19*
Se	31-08-2012	03-07-2023	*448*	31-10-2011	08-09-2023	*19*
Temperature	14-08-2012	05-08-2023	*377*	31-10-2011	08-09-2023	*19*
TDS	05-05-2012	05-08-2023	*735*	31-10-2011	08-09-2023	*19*
TSS	31-08-2012	03-07-2023	*389*	31-10-2011	08-09-2023	*19*
Turbidity	12-02-2020	05-08-2023	*359*	05-09-2023	08-09-2023	*10*
U	20-07-2016	03-07-2023	*328*	31-10-2011	08-09-2023	*19*
V	20-07-2016	03-07-2023	*328*	05-09-2023	08-09-2023	*11*
Zn	31-08-2012	13-01-2023	*144*	31-10-2011	08-09-2023	*19*

^Limits of reporting for Cu for ALS and MPL supplied data (0.001 mg/L) is too high for comparison to DGV. Only values >0.001 (i.e. high readings) could be included in plots, and statistics were limited to Chemcentre supplied data (LOR = 0.0001 mg/L).

5.2 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 and total ammonia is based on improved understanding of interactions between these toxicants and hardness, pH, temperature (for total ammonia) or dissolved organic carbon (DOC) (for Cu), in determining actual bioavailability to aquatic organisms (ANZG 2023a; ANZG 2023b). These updated guidances are still in draft form, but are expected to be published in near future thus are included in the derivation of interim SSGVs for Gringer Creek.

The requirement for paired toxicant, hardness, pH and temperature (for total ammonia) and DOC (for Cu) data to assess the actual toxicity of these analytes at Gringer Creek is a limitation in the current dataset, which can be addressed using approximations given in ANZG (2023a and 2023b) (see below). In particular, hardness (defined as the aqueous concentration of calcium and magnesium ions, and expressed as mg/L CaCO3) in freshwaters has an ameliorating effect on further metal toxicants, including Cd, Cr, Ni and Zn, to which hardness-modified trigger values (HMTVs) are applied in waters with hardness > 30mg/L $CaCO_3$ (Warne et al., 2018; ANZG 2018). High hardness is also indicated in amelioration of toxicity effects of nitrate (NO_3) in Pilbara waterways (defined as >160 mg/L; van Dam et al., 2022), however specific guidance beyond that region is yet to exist. Currently, measurements of hardness at Gringer Creek are limited to records from WRM/SLR in 2011 and 2023, and sporadic measurements supplied by NBG (*n* = 26 samples). Given the importance of hardness in calculating specific toxicity values for a number of analytes, this omission in regular monitoring is a limitation in the current dataset. Collection of these data prior to construction of the RDA2 and commencement of dewatering discharge should be undertaken to address these gaps, to assist formalisation of interim SSGVs to full SSGVs for ongoing use at Gringer Creek.

The following approximations were used to derive interim SSGVs, following Warne et al., (2018); ANZG (2023a); ANZG (2023b), with the aim of being as conservative as reasonable. Minimum hardness recorded at Gringer Creek was 450 mg/L, up to 2400 mg/L, which is considered very to extremely hard (ANZECC/ARMCANZ 2000). In this case, the default hardness thresholds (usually 30 mg/L) for DGVs in systems where hardness is unknown are overly conservative for use at Gringer Creek. Where applicable, the minimum recorded hardness (450 mg/L $CaCO_3$) is used for calculations of adjusted 95% DGVs and interim SSGVs.

<u>Ammonia:</u>

Updated guidance is available for ammonia based on new ecotoxicity data for total ammonia-N (hereafter TAN) 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 95% and 99% 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 TAN mg N/L are provided for a range of pH and temperatures (ANZG 2023a, Appendix D). Generally, the laboratory derived total ammonia (expressed as NH_3-N) is directly comparable to the DGVs for total ammonia[5].

[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).

The SSGV for Gringer Creek was derived using the whole dataset of pH and temperature (Table 9), applying the 80th percentile values for each (pH 7.1, temperature 20ºC) to the table provided in ANZG 2023a (Appendix D). Although this is not too dissimilar to the values given in absence of pH/temperature data, it is recommended that local data be used to derive SSGVs as closely as possible (ANZG 2023a). Should the distribution of pH and or temperature change substantively following commencement of discharge to Gringer Creek, then the toxicity of ammonia present would also change, and a new interim SSGV would need to be calculated and applied.

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. 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. These values are applicable at pH 6.5 – 8.0 and hardness of 2 – 200 mg/L. Ideally, future water quality monitoring would include regular collection of DOC data with which to derive actual toxicity of Cu in Gringer Creek[6], as the default values are likely to be overly conservative.

Cyanide:

Cyanide (measured as free/uncomplexed HCN) has an ANZG (2018) 95% DGV for freshwater ecosystems of 0.007 mg/L. Data supplied for analysis of free cyanide at Gringer Creek has an LOR of 0.01 mg/L, which is too high for comparison to DGVs, thus precluding statistical analysis, and is suited to the far more lenient guideline values of recreational water quality (0.1 mg/L; ANZECC/ARMCANZ 2000). Total cyanide is a measure that includes measurable components bound in inorganic and organic compounds. Weak acid dissociable cyanide (WAD cyanide) refers to free cyanide plus acid dissociable complexes. Both of these measures include cyanide complexes that are not necessarily bioavailable. Therefore, cyanide toxicity cannot be commented on in this report, nor free cyanide data be compared to ANZG (2018) guidelines using the data available. It is highly recommended that background data for free cyanide be gathered, at an appropriate LOR, prior to commencement of mining discharge should there be potential for discharge to contain cyanide. The ANZG (2018) DGV is retained as SSGV.

Hardness modified trigger values (HMTVs) for heavy metals:

Default guideline values for Cd, Cr, Ni and Zn 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; see Table 10). To derive an interim SSGV for these analytes, the lowest hardness value recorded (26 samples - 450 mg/L) was used. This was considered a conservative approach aimed at providing accuracy as well as being sufficiently conservative to protect aquatic ecosystem values. However, it is again recommended that further collection of hardness data be conducted at Gringer Creek, given the likely important role of hardness in ameliorating toxicity of several analytes in the system.

[6] Where Cu concentrations remain well below the DGV (DOC =<0.5 mg/L), then an adjusted DGV is unlikely to be necessary (ANZG 2023b, pg. 29). However, in situations where dissolved Cu exceed the DOC adjusted guideline values, then copper speciation should be conducted and the 'bioavailable' Cu tested against the non-adjusted DGV (i.e. DOC=<0.5 mg/L; ANZG 2023b).

5.3 Summary statistics and timeseries of selected analytes

Generally, most analytes were recorded at low concentrations (usually below 99% or 95% DGVs), or below laboratory LORs. For the vast majority of analytes, the 95% DGV for slightly to moderately disturbed ecosystems remain appropriate for use at Gringer Creek, and are included as the interim SSGV for potential future mining discharge to Gringer Creek. A summary of statistical analysis for each analyte is presented in Table 10. Interim SSGVs are presented in Table 11 (section 5.3.2).

5.3.1 Comparisons against ANZG DGVs

There were several stressors and toxicants recorded during the period of monitoring (2011 – 2023) that were above default guidelines, ranging from isolated exceedances, to continually elevated levels. These included metal toxicants, non-metal toxicants and stressors. However, further examination indicates not all are analytes of concern, nor demand an interim SSGV be instated over the ANZG (2018) defaults, as discussed below. Summary statistics for all key analytes are presented in Table 10.

<u>Aluminium</u>

Dissolved Al was generally below detection limits, and the 80th %ile of monitoring data was below the 95% DGV (0.05 mg/L; Table 10). However, there were periodic increases in Al concentrations that appeared as ‘pulsed’ events and occurred across sites almost simultaneously, suggesting release from the catchment. Timing indicates the release was likely in response to winter rainfall (Figure 22). During these events, concentrations of Al were temporally above the ANZG 95% DGVs, such that the 95th percentile is 6x DGV (0.33 mg/L) and maximum value 11x DGV (0.67 mg/L; GRCK01 17-08-2017). Seasonal increases in Al were apparent when comparing monthly average data (2012 to 2023; Figure 23), which showed that mean Al in July (0.09 ± 0.02) and August (0.16 ± 0.03) was much above the long-term 80th%ile. Mean concentrations then decline between September (0.07 mg/L) and October (0.03 mg/L). To account for fluctuations in Al with the onset of flows, the July, August and September data were analysed separately to derive a seasonal 80th%ile (0.19 mg/L; Table 10). Because Al appears to spike in response to winter rainfall/catchment runoff, a ‘wet season’ SSGV is proposed, defined by the first flush of rainfall or large rainfall events. During dry months the DGV of 0.055 mg/L should be applied, however, when the system is flowing in response to recent rainfall, the wet season SSGV of 0.19 mg/L should be applied.

Aluminium is naturally abundant in the environment, and is released from the weathering of rocks and deposited in the sediments of riverine, estuarine and coastal environments. Anthropogenic effluents may also release significant Al to environments exposed to industrial wastes (ANZG 2018; Botté et al., 2022). Similarly to other toxicants, water hardness reduces bioavailability (thus toxicity) of Al, noting Gringer Creek has recorded high hardness (450 – 2400 mg/L) it is probable that hardness is at least in part ameliorating toxicity to fauna.

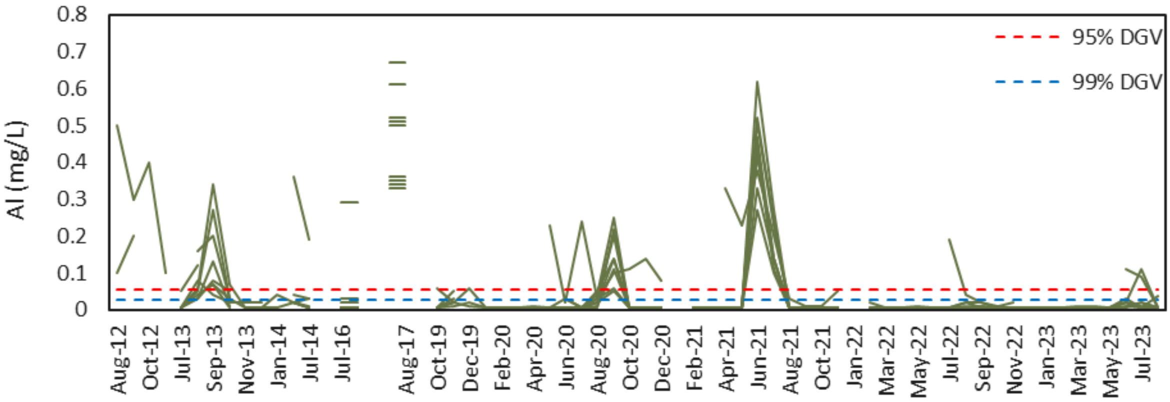


Figure 22. Timeseries of dissolved Al concentrations at Gringer Creek aquatic monitoring sites (GRCK01-08) and NBG water quality monitoring site GRCK 09.

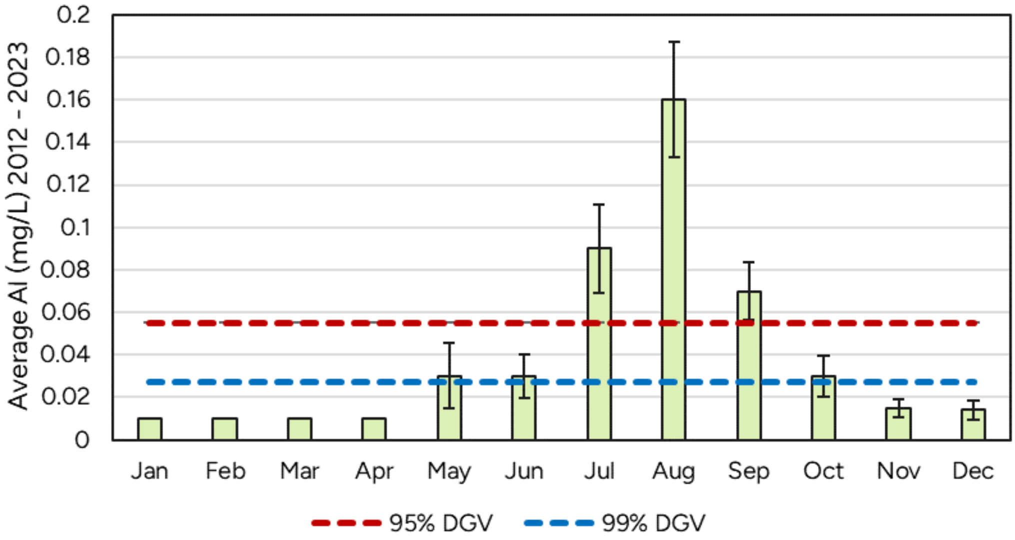


Figure 23. Monthly average Al concentrations (mean ± SE) recorded at Gringer Creek (2012 – 2023). The ANZG (2018) 95% DGV (0.0055 mg/L) and 99% DGV (0.027 mg/L) shown.

Arsenic

Nearly all As observations were below the LOR (ALS = 0.001 mg/L), and thus below the 99% DGV (0.001 mg/L) and well below the 95% DGV (0.024 mg/L). Data collected by WRM/SLR analysed by ChemCentre with an LOR of 0.0001 also failed to detect concentrations above the 99% (or 95%) DGV. This is with notable exceptions in 2019, with three instances where As was above the 95% DGV, to a maximum of 0.25 mg/L As detected at GRCK09 in Nov-19, and 0.095 mg/L at GRCK08 in Dec-19 (Figure 24). These concentrations were not greater than the 80% protection level (340 mg/L), noting this DGV is designed to be protective against acute effects, not chronic effects (ANZG 2018). Where As exceeds DGV/SSGVs, further laboratory analysis should be performed to speciate total As into its less toxic As(III) and more toxic As(V) forms, and As(V) values then compared to the DGVs (ANZG 2018; Warne et al., 2018). That As exceedances only occurred over two sampling occasions in 2019, this suggests an environmental release event (natural, anthropogenic, or combination of both), or, that sample contamination and/or data transcription error occurred. Any further elevations detected prior to RDA2 construction should undergo speciation at a NATA accredited laboratory and the source of As further investigated, to determine whether there is a pre-existing source of As in the catchment.

Because there were no further exceedances (or even detections) of As other than in 2019, the ANZG 95% DGV (0.024 mg/L) is recommended to be retained as the interim SSGV for Gringer Creek.

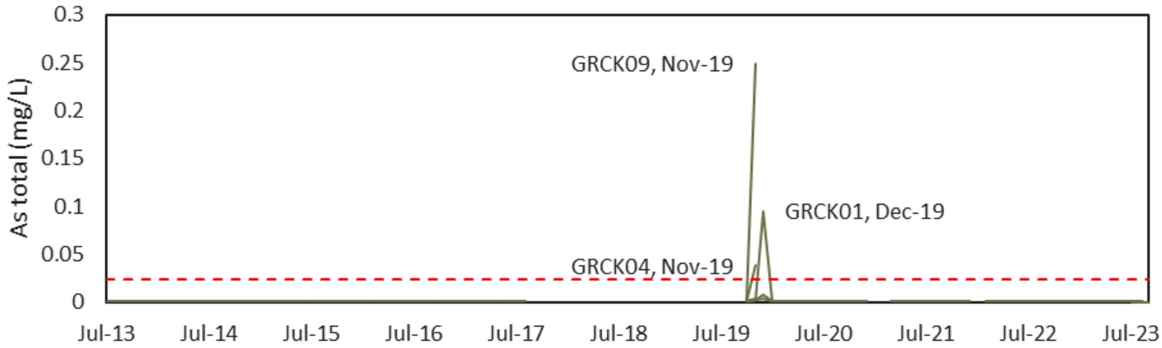


Figure 24. Timeseries of dissolved As concentrations at Gringer Creek aquatic monitoring sites (GRCK01-08) and NBG water quality monitoring site GRCK 09. The ANZG (2018) DGV for 95% species protection is indicated by the red dashed line (0.024 mg/L).

Cobalt

Dissolved Co was frequently above the DGV at Gringer Creek, over all years of monitoring (Figure 25). The maximum recorded value was 0.366 mg/L at GRCK06 (24-10-2019), however this was by far an outlier with the second highest reading at 0.09 mg/L, also at GRCK06 (19-09-2012). All outliers (above 0.05 mg/L) were recorded from GRCK06, suggesting a point source at or near this site (Figure 26). However, the whole creekline frequently recorded high dissolved Co, with the median (0.006 mg/L) and 80th %ile (0.013 mg/L) greater than the DGV of 0.0014 mg/L. This was especially true in the upper reaches (GRCK01 – 04; Figure 26). It is worth noting that the DGV of 0.0014 mg/L is a **low reliability** trigger threshold, due to contradictions in the ecotoxicity data for acute and chronic effects on biota (ANZG 2018). Nevertheless, these high levels of Co are within the ranges that would be expected to have deleterious impacts on aquatic fauna (ANZG 2018). Given the high background levels of Co detected consistently across sites at Gringer Creek, an interim SSGV was derived using the 80th %ile value. However, to increase conservatism, four outlier values from GRCK06 (>0.05 mg/L) were removed, revising the 80th %ile down to 0.012 mg/L. Again, Co levels at or above this value are unlikely to be protective of toxic effects on biota, but is in acknowledgement of pre-existing levels or legacy issues in the catchment, and current aquatic biota likely reflects the acute/chronic toxicity of the currently elevated Co. A further increase in Co could lead to a decline in aquatic biota in Gringer Creek.

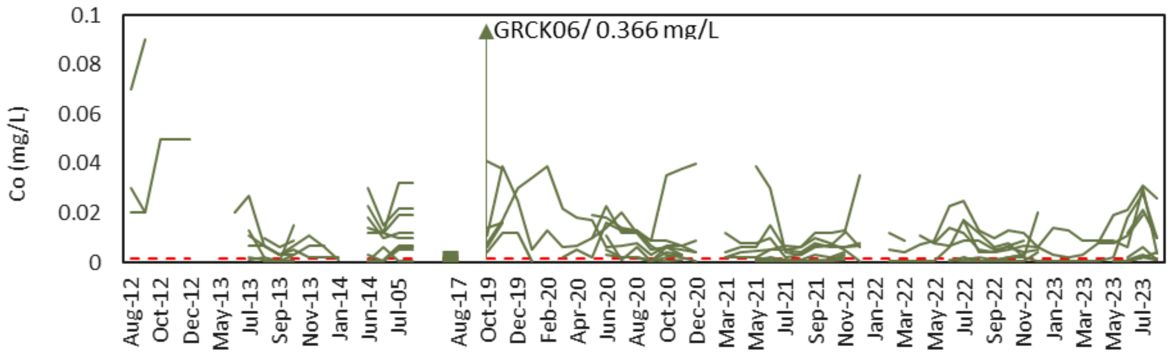


Figure 25. Timeseries of dissolved Co concentrations at Gringer Creek aquatic monitoring sites (GRCK01-08) and NBG water quality monitoring site GRCK09. The ANZG (2018) DGV for 95% species protection is indicated by the red dashed line (0.0014 mg/L).

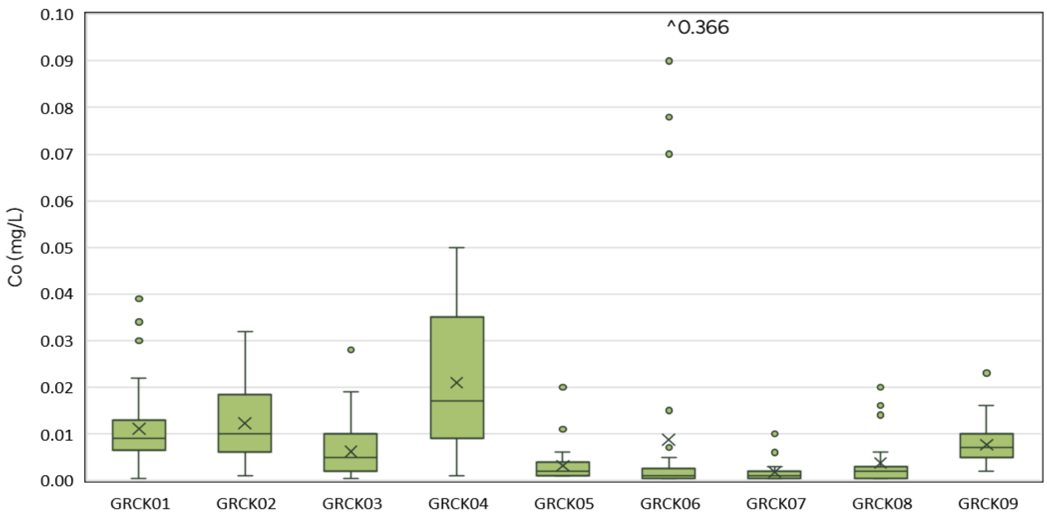


Figure 26. Distribution of Co (mg/L) concentrations across Gringer Creek aquatic survey sites (GRCK01-08) and NBG water quality monitoring site GRCK09.

Copper

There were limitations regarding LORs in the dissolved Cu dataset from ALS/MPL that restricted statistical analysis to data collected by WRM/SLR (*n* = 19; Table 10). Data supplied by ALS/MPL were included in plots where detections were above LOR (>0.001 mg/L; over 2x the 95% DGV of 0.00047 mg/L), but excluded from statistical analysis.

Similarly to As, Cu concentrations were generally low, below LOR and/or DGVs, however several very high readings were detected in November and December 2019, with a maximum value of 1.26 mg/L recorded at GRCK01 (Figure 27). A second peak in Cu concentration occurred in Aug-20, with between 0.029 mg/L (GRCK08) and 0.112 mg/L (GRCK09) recorded (note, GRCK08 and GRCK09 are not situated proximally to each other; Figure 2). These levels of Cu are significant, and exceed the ANZG 2023b guideline for 80% species protection (0.0013 mg/L[7]) by several orders of magnitude, suggesting acute negative effects to aquatic biota are likely during these events. However, updated guidance also stipulates that DOC

[7] High Cu records are also above the 2018 DGV (0.0014 mg/L), which is anticipated to be superseded in near future once updated guidance is published.

ameliorates Cu toxicity, and without local DOC data for Gringer Creek, there is potential that the DGV of 0.00047 mg/L is overly conservative.

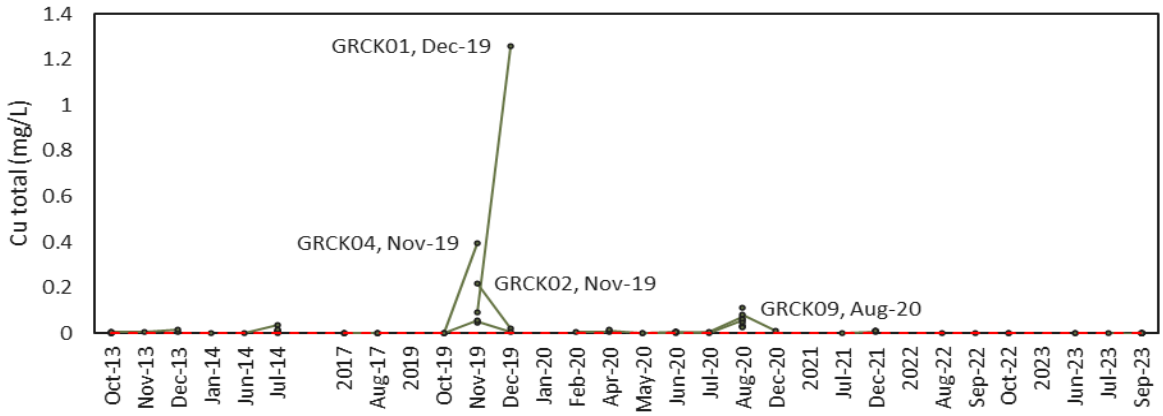


Figure 27. Timeseries of dissolved Cu concentrations at Gringer Creek aquatic monitoring sites (GRCK01-08) and NBG water quality monitoring site GRCK 09. The ANZG (2018) DGV for 95% species protection is indicated by the red dashed line (0.00047 mg/L).

<u>Manganese</u>

Generally, dissolved Mn was below the 95% DGV (1.9 mg/L) at Gringer Creek, however some notable concentrations were recorded (Figure 28). The maximum recorded value was 22.2 mg/L at GRCK06 (24-10-2019), and the four highest measurements were also recorded at GRCK06, including Oct-11 (12 mg/L), Sept-12 (7.7 mg/L), and Aug-12 (6.7 mg/L), with further high values recorded at GRCK01 (5.8 mg/L, 12-02-2020). However, high concentrations have occurred across monitoring sites, therefore it is hard to distinguish point-sources (i.e. in proximity to GRCK06) from the possibility of background Mn levels being elevated relative to ANZG (2018) DGVs. Given the 80th%ile value remains below DGV (1.3 mg/L; Table 10), it is recommended that the default guideline be retained as interim SSGV. It is noteworthy that Mn and Co were both elevated at site GRCK06, indicating a likely groundwater source at this site.

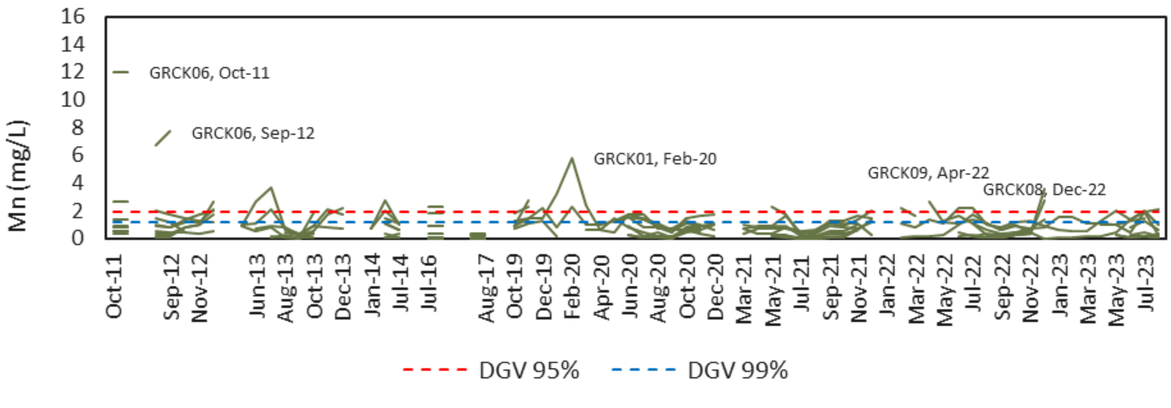


Figure 28. Timeseries of dissolved Mn concentrations at Gringer Creek aquatic monitoring sites (GRCK01-08) and NBG water quality monitoring site GRCK 09. The ANZG (2018) DGV for 95% species protection is indicated by the red dashed line (1.9 mg/L).

Total nitrogen, NOx and total phosphorus (as stressors)

Various forms of nitrogen are assessed against DGVs as either direct toxicants (e.g. nitrate NO_3-N) or as stressors, whereby levels are not directly toxic, but their prolonged elevation will likely cause habitat loss due to the effects of eutrophication (Boulton et al., 2014; ANZG 2018). The ANZG (2018) default guidelines give two values for both total-N, recommended for use below elevation of 150m ('lowland' rivers, 1.2 mg/L) and above 150m ('upland' rivers, 0.45). The same is given for oxides NOx, above (0.2 mg/L) and below 150m (0.15 mg/L). However, such a distinction was not supported by local data analysed as part of the Framework for the Assessment of River and Wetland Health (SW FARWH; van Looj et al., 2009). Furthermore, true 'reference' state data for nitrogen in rivers is difficult to obtain for SWA, due to the widespread legacy of clearing for agriculture (van Looj et al., 2009; Storer et al., 2010). The total-N categories currently accepted under the SWIRC[8] align more closely with the 'lowland' river DGVs (ANZECC/ARMCANZ 2000).

Using the DGV of 1.2 mg/L, there were few individual values of total-N recorded from Gringer Creek that were above the DGV (approx. 4% of observations). There were 20 records of nitrite/nitrate-N (NOx) above the stressor DGV (0.2 mg/L), or around 6% of the total observations, with a maximum of 1.45 mg/L (Table 10). Despite these occasional high values being recorded, the 80th%ile values for both total-N (0.6 mg/L) and NOx (0.08 mg/L) remained below the DGV, and the 95th%ile were just below (total-N, 1.14 mg/L) or just above (NOx, 0.22 mg/L) their respective DGVs. It is therefore recommended that the DGVs is retained as the interim SSGV for Gringer Creek. It is also important to note that for stressors, individual spot measurements above the DGV do not constitute an exceedance. ANZG (2018) expects that a systematic increase in a stressor is required for an exceedance to occur. Generally the median of monthly data for a 12 month period are tested against the DGV, and if the median of monitoring data is above the DGV it is accepted there has been a systematic increase that poses a threat and constitutes an exceedance.

Similarly to total-N, the SW FARWH identified no distinction between systems above and below 150m with regard to total-P concentrations, and took the same approach to classifying total-P categories[9]. Using the 'lowland' river DGV, there were 5 records of total-P over the 12+ years of monitoring at Gringer Creek that were above the DGV (or approx. 2% of observations), and the 80th%ile value was well below the DGV (0.02 mg/L), as was the 95th%ile value (0.04 mg/L). Therefore, it is recommended the DGV 0.065 mg/L be retained as interim SSGV for Gringer Creek.

pH

Over the period of monitoring, Gringer Creek has generally had slightly acidic to neutral pH, with a 20th – 80th%ile range for pH at Gringer Creek of 6.3 – 7.2 (Table 10). This is slightly below the ANZG (2018) guidelines (6.5 – 8.0) and well within the range considered normal for streams in the southwest of WA. Approximately 35% of the observations fell outside this range, with a minimum pH of 4.67 (GRCK01, 23-06-2014) and a maximum pH of 11.3 (GRCK06, 31-10-2011).

Although pH can have direct effects on aquatic biota, perhaps more importantly is the relationships many toxicants have with pH, often having higher bioavailability, or higher proportions of toxic valencies, at pH levels outside of the range 6.5 to 8.0 (ANZG 2018). Because several toxic analytes are present at Gringer Creek which are likely less bioavailable at circum-neutral pH, it is recommend that the interim SSGVs use the 20th – 80th%ile ranges rather than the DGVs. Should discharge substantially alter pH, then further alterations to the bioavailability of other toxicants may also occur.

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8 Total-N <0.75 Low; 0.75 – 1.2 Moderate; 1.2 – 2.0 High; >2.0 Very high (van Looj et al., 2009).

9 Total-P <0.02 Low; 0.2 – 0.08 Moderate; 0.8 – 2.0 High; >2.0 Very high (van Looj et al., 2009).

Turbidity

Gringer Creek typically has low turbidity (81% of observations are below the lower DGV of 10 NTU). Occasionally, high turbidity has been recorded, typically during the summer months. As Gringer Creek is an intermittent stream, undergoing disconnection and drying over summer, increases in turbidity (and other attributes such as TSS) may be expected as pools recede, and processes associated with organic matter decomposition and algal growth occur. However, as turbidity is a stressor, continuous elevation above the DGV (rather than spot-exceedances) would be required to justify a SSGV.

Total suspended solids (TSS)

No formal DGV exists for TSS. Occasional pulses were evident in Gringer Creek, with a maximum of 111 mg/L TSS recorded (Figure 29). However, generally TSS was below LOR (80^{th} %ile <5mg/L; Table 10). Temporary pulses of TSS can be a natural occurrence in streams, or as a result of anthropogenic disturbance (such as clearing within the catchment), however a sustained increase in TSS loads can directly impact aquatic fauna by smothering gills and can cause indirect impacts from habitat degradation. Because TSS was generally very low at Gringer Creek, 5 mg/L (equivalent to current LOR) is proposed as interim SSGV.

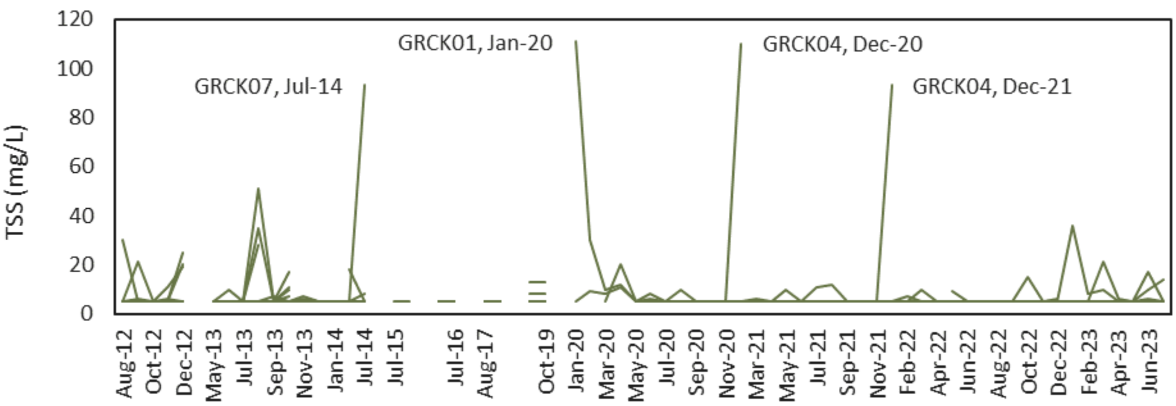

Figure 29. Timeseries of total suspended solids (TSS) concentration at Gringer Creek aquatic monitoring sites (GRCK01-08) and NBG water quality monitoring site GRCK 09.

Temperature

There is currently no accepted default guideline for temperature, nor the SW FARWH does not prescribe a specific temperature guidance for SWWA rivers (van Looj et al., 2009). However, diel temperature fluctuations are recommended to be kept to under 4ºC, as fluctuations greater than this would be expected to cause direct mortality to aquatic biota. Furthermore, discharge of mine water should not increase stream temperatures, as biota have direct sensitivity to temperature, and temperature is also implicated in the toxicity status of a number of analytes, particularly ammonia, Cu and HCN (ANZG 2018; Storer et al., 2010; ANZG 2023a).

Temperatures measured at Gringer Creek have ranged from a minimum of 4.2ºC to a maximum of 29.8ºC. Seasonal variation in stream temperatures show June and July to have the lowest temperatures (a mean of approximately 10ºC) and December and January the highest (mean of 24.9ºC) noting that fewer measurements are taken in the warmer months due to Gringer Creek becoming dry. That there are seasonal fluctuations in stream temperatures is axiomatic; however, any prospective discharge operations should not increase stream temperatures above seasonal norms (Figure 30). If temperature is identified as a potential risk to the receiving environment, then derivation and implementation of seasonal SSGVs should be considered. Monitoring should compare end of pipe discharge water temperature to ambient stream water temperature to test for potential to exceed the SSGV, and thus potential for thermal stress to aquatic biota.

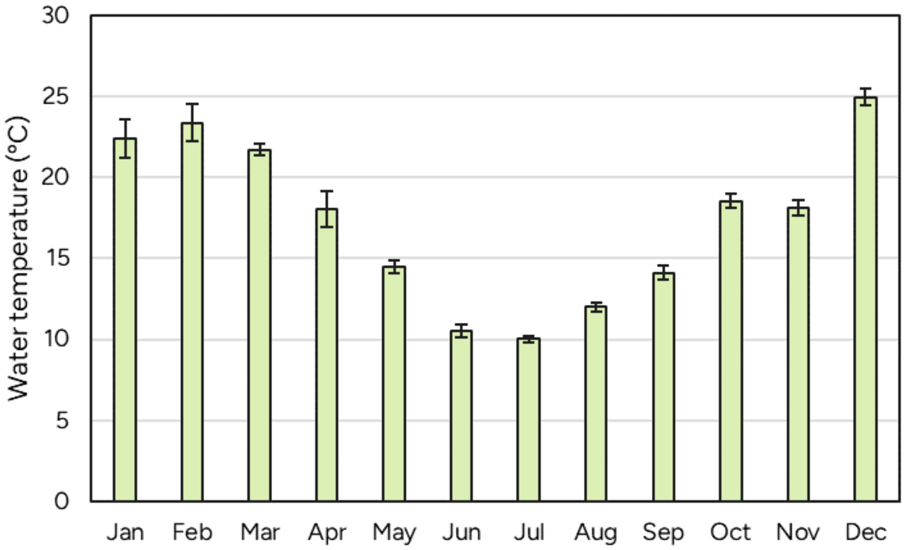

Figure 30. Monthly average stream temperature (mean ± SE) recorded at Gringer Creek 2011 – 2023.

Table 10. Summary statistics for key water quality analytes, alongside DGVs and proportion of samples elevated compared to 95% DGVs. The 95% species protection are considered appropriate for slightly to moderately disturbed systems, 99% DGVs are given for context. Stressor DGVs for south-western Australia (ANZECC/ARMCANZ 2000) are included under the 95% DGV column. Units are mg/L unless specified in parentheses. <LOR = below laboratory limits or reporting. Non-metal toxicants denoted with -T to distinguish from nutrient stressors related to eutrophication (E). Hardness modified TVs presented where appropriate, shaded green (see footnotes).

Analyte	*N samples*	DGV 95%	DGV 99%	Median	80th %ile	Max.	% > DGV 95%	Comments
Al	*378*	0.055	0.027	<LOR	0.05	0.67	18.5%	80th%ile value below DGV 95%. Exceedances are periodic and likely associated with pre-existing catchment sources.
Al (wet season)	*165*	0.055	0.027	0.02	0.19	0.67	34%	Analysis includes July, August and September data
Alkalinity	*370*	-	-	23	30	64	-	
As^ (CC)	*11*	0.024	0.001	0.0005	0.0005	0.00012	-	Chemcentre (CC) LORs much lower than ALS/MPL, therefore analysis split between laboratory sources.
As (ALS/MPL)	*396*	0.024	0.001	<LOR	<LOR	0.25	<0.01%	
B	*347*	0.94	0.34	<LOR	<LOR	0.11	-	
Ba	*348*	-	-	0.074	0.094	0.204	-	
Cd#^	*307*	0.0002	0.00007	<LOR	<LOR	0.0009	3%	95% HMTV = 0.0023 mg/L. There were no exceedances at this level
Co	*387*	0.0014	-	0.006	0.013	0.37	78%	
Cr#^	*314*	0.001	0.00001	<LOR	<LOR	0.002	0%	95% HMTV = 0.001 mg/L
Cu (CC)	*19*	0.00047	0.0002	<LOR	0.0002	0.0004	0%	LOR from ALS/MPL too high for comparison to DGVs. All detections (*n* = 73) were above DGV. Only ChemCentre data retained for analysis. Require DOC data to develop SSGV.
Cu (ALS/MPL)	*73*	0.00047	0.0002	N/A	N/A	1.26	N/A	
Cyanide - free	*7*	0.007	0.004	<LOR	<LOR	<LOR	N/A	The data provided by MPL has an LOR = 0.01 which is far too high for comparison against ANZG (2018) DGVs. WAD and total cyanide is not directly comparable
DO (% sat.)	*19*	80 - 120	-	100.3	83.3 - 102.7	32.5 - 111	11%	20th - 80th %ile values are well within DGVs for southwest rivers.
EC (µS/cm)	*433*	120-1500	N/A	8,130	10,503	17,850	99%	Elevated EC is a known legacy of catchment clearing in SWA
Fe	*372*	N/A	N/A	0.24	1	83.1	-	No DGV for Fe
Hardness	*26*			780	660 - 1500	450 - 2400		Hardness is likely protective against toxicity for numerous analytes at Gringer Creek. Discharge should not reduce hardness. More data required.
Mn	*419*	1.9	1.2	0.68	1.3	22.2	8%	
Mo	*372*	0.034	N/A	<LOR	<LOR	0.12	<0.01%	One high record at GRCK01 16-12-2019 (0.12 mg/L)
N-NH_3 (T)†	*359*	0.76	0.25	0.03	0.1	3.5	2%	DGVs applicable at pH 7.1 and temperature 20°C
N-NO_3 (T)**	*363*	2.4	1.0	0.03	0.09	1.45	0%	
N-NO_X (E)	*314*	0.2	-	0.02	0.08	1.45	6%	
N Total (E)	*373*	1.2	-	0.3	0.6	2.6	3.5%	
Ni#	*372*	0.008	0.011	0.001	0.002	0.027	1%	95% HMTV = 0.11. There were no exceedances at this level.
P Total (E)	*297*	0.065	-	0.01	0.02	0.16	2%	
pH (H^+)	*431*	6.5 - 8	N/A	6.74	6.29 - 7.21	4.67 - 11.3	35%	
Pb		0.0034	0.001	<LOR	<LOR	0.003	0%	

Analyte	*N samples*	DGV 95%	DGV 99%	Median	80th %ile	Max.	% > DGV 95%	Comments
S-SO_4	*419*	-	-	207	299	545	-	
Se	*459*	0.011	0.005	<LOR	<LOR	<LOR	-	
Temp (°C)	*396*	-	-	14.25	10.4 - 20.2	4.2 - 29.8	-	
TDS	*754*	-	-	5,454	7,405	11,610	-	Salinity as TDS – reflects high EC
TSS	*389*	-	-	<LOR	<LOR	111	-	LOR = 5mg/L, proposed as interim SSGV.
Turbidity (NTU)	*369*	10 - 20	-	3.6	9.3	150	10%	
U	*295*		-	<LOR	<LOR	<LOR	0%	
V	*11*	0.0006	-	0.0001	0.0002	0.0002	0%	
Zn#	*163*	0.008	0.0024	0.008	0.02	0.2	45%	95% HMTV = 0.08. there were 7 exceedances (approx. 4%) at this level.

^DGVs apply to the more toxic valencies of these analytes, if exceedances are detected then further speciation of analytes should be conducted to determine the relative proportions of more toxic and less toxic forms.

#Hardness modified TVs calculated at hardness = 450 mg/L as $CaCO_3$. %exceedances are against the default DGV at 30 mg/L hardness as $CaCO_3$. Ongoing collection of hardness data is advised to support analysis of toxicity.

*Updated guidance for Cu toxicity requires corresponding DOC data, as DOC is now known to have an ameliorating effect on Cu. *In lieu* of this data, the recommended DGV at DOC =< 0.5 mg/L is used.

** The ANZECC/ARMCANZ (2000) DGVs for nitrate toxicity are erroneous, ANZG (2018) recommend guidelines provided in Hickey et al., (2013) derived for New Zealand waterways in absence of data specific to Australian systems.

† Draft updates to ANZG guidance on the toxicity of total ammonia-N (as NH3-N) is used to derive these DGVs (ANZG 2023a). The 80th%ile values for pH (7.1) and temperature (20ºC) at Gringer Creek were used in calculations. Updated guidance has been adopted here in anticipation of it being published in near future.

5.3.2 Interim SSGVs

Table 11. Interim site-specific guideline values for use at Gringer Creek. DGVs for non-metal toxicants are denoted with (T) to distinguish from those associated with eutrophication (E).

Analyte	Unit	ANZG 95%	ANZG 99%	80th%ile value	Interim SSGV
Al (dry season)	mg/L	0.055	0.027	**0.05**	**0.055**
Al (wet season)A	mg/L	0.055	0.027	**0.19**	**0.19**
Alkalinity	mg/L	N/A	N/A	**64**	**-**
As	mg/L	0.024	0.001	**0.00012**	**0.024**
B	mg/L	0.94	0.34	**0.11**	**0.94**
Ba	mg/L	N/A	N/A	**0.09**	**0.09**
Cd	mg/L	0.0002	0.00006	**<LOR**	**0.002B**
Co	mg/L	0.0014	N/A	**0.013**	**0.012C**
Cr (IV)	mg/L	0.001	0.00001	**<LOR**	**0.01B**
Cu	mg/L	0.00047	0.0002	**0.002**	**0.00047E**
Cyanide - free	mg/L	0.007	0.004	**<LOR**	**0.007**
DO	%	80 - 120		**83 – 102**	**80 - 120**
EC	(μS/cm)	120-1500^	N/A	**10,503**	**-**
Fe – dissolved	mg/L	N/A	N/A	**1.0**	**1.0**
Hardness	mg/L	-	-	**1,500**	**450 – 2400F**
Mn	mg/L	1.9	1.2	**1.3**	**1.9**
Mo	mg/L	0.034	N/A	**<LOR**	**0.034**
$N\text{-}NH_3$* (T)	mg/L	0.76	0.25	**0.1**	**0.76G**
$N\text{-}NO_3$ (T)	mg/L	2.4	1	**0.09**	**2.4H**
$N\text{-}NO_X$ (E)	mg/L	0.2	-	**0.08**	**0.2**
N-total (E)	mg/L	1.2	-	**0.6**	**1.2**
Ni	mg/L	0.011	0.008	**0.002**	**0.11B**
P-total (E)	mg/L	0.065	-	**0.02**	**0.065**
pH	H+	6.5 - 8	N/A	**6.3 – 7.2**	**6.3 – 7.2I**
Pb	mg/L	0.0034	0.001	**<LOR**	**0.0034**
S-SO4	mg/L	-	-	**299**	**299**
Se	mg/L	0.011	0.005	**<LOR**	**0.011**
Temperature	C			**10.4 - 20.2**	**-**
TDS	mg/L	-	-	**7,405**	**7,405**
TSS	mg/L	-	-	**<LOR**	**5**
Turbidity	NTU	10 - 20		**9.3**	**20**
U	mg/L	0.0005	-	**<LOR**	**0.0005**
V	mg/L	0.0006	-	**0.0002**	**0.0006**
Zn	mg/L	0.008	0.0024	**0.02**	**0.08B**

A – 'wet season' as determined by rainfall, including first flush winter rains and proceeding large rainfall events.
B – Hardness modified trigger value (HMTV) calculated at hardness = 450 mg/L as $CaCO_3$.
C – Interim SSGV is the 80th%ile value, excluding high outliers.
D – Calculated from Chemcentre data only (*n* = 19).
E – ANZG (2023b) DGV at DOC =< 0.5 mg/L used, in absence of local DOC data. Measurement of local DOC advised.
F – Hardness values are min-max range recorded from Gringer Creek (*n* = 26). Recommend further collection of hardness data to support full operational SSGVs.
G – Calculated using 80th%ile pH (7.1) and temperature (20ºC). See ANZG (2023b), and rationale in section 5.2.
H – ANZG (2018) recommend use of 'grading' trigger values presented in Hickey et al., (2013) as 95% DGV, *in lieu* of updated guidance specific to Australia.
I – 20th to 80th%ile range for Gringer Creek.

5.4 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 Gringer Creek. 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. 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, 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.

6.0 Summary

This report summarised the findings of contemporary baseline surveys of Gringer Creek conducted in 2023, building on previous baseline studies conducted in 2011 and informed by a brief desktop review of aquatic ecosystems and fauna recorded in the area. Below summarises the findings of the baseline survey, development of interim SSGVs, and provides recommendations based on these findings to fill data gaps, and for ongoing monitoring under RDA2 operation and discharge scenarios.

Water quality

Water quality data were compared to ANZG (2018) DGV for 95% species protection. The 95% DGVs are recommended for use in slightly to moderately disturbed ecosystems, and are thus appropriate for Gringer Creek given catchment clearing for ongoing agricultural and silvicultural land uses, and associated legacy impacts including secondary salinisation.

Water quality measured at Gringer Creek in 2023 was generally good, with circum-neutral pH, dissolved oxygen at around 100% saturation, and with the majority of toxicant analytes well below DGVs. Some exceptions included Cobalt, recorded above the DGV of 0.0014 mg/L at several sites, and elevated Mn (relative to the 95% DGV) recorded at GRDS02. Otherwise, most notable water quality attributes at Gringer Creek were high conductivity (up to 5,960 µS/cm), considered brackish, and very likely due to prior land clearing. Most south-west freshwater fish and crayfish can tolerate salinities up to 8,000 µS/cm (Morrissy 1978, Beatty et al. 2008, 2010). Nevertheless, salinity at Gringer Creek would be limiting to many species for which the habitat would otherwise be suitable. In addition, hardness at Gringer Creek was very high, between 450-2400 mg/L CaCO3. High hardness is not of concern at Gringer Creek in terms of direct impact to fauna, and there are no associated DGVs for hardness. However,

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

hardness is known to reduce the toxicity of several metal and non-metal analytes, including Cd, Cr, Cu, Ni, Zn and ammonia.

Macroinvertebrates

A total of 72 macroinvertebrate taxa were recorded from Gringer Creek in 2023 dominated by insects and their larvae, with larval Diptera (true flies) the most diverse group followed by taxa of 11 Coleoptera (aquatic beetles). Common taxa included the salt tolerant amphipod *Austrochiltonia subtenuis*, the chironomid *Tanytarsus* sp., Ceratopogoninae sp. (biting midge) larvae and Scirtidae beetle larvae. There was a clear spatial pattern occurring in macroinvertebrate assemblages between upper (GRCK01-03) and lower sites (GRCK04-GringerDS11), possibly due to differences in habitat structure and hydrology (i.e. differences in groundwater connectivity).

There were no formally listed macroinvertebrate taxa recorded from Gringer Creek in 2023. However, there were a number of south-west endemic species which are of conservation and/or scientific interest, including SWA endemic, Gondwanan damselflies *Archiargiolestes* sp., and dragonfly *Austroaeschna anacantha*, several endemic beetles, including *Megaporus solidus*, *Sternopriscus browni* and S*. marginatus*, and *Haliplus fuscatus/gibbus*. Furthermore, several SWA endemic species belonged to the orders Plecoptera and Trichoptera, taxa known to be sensitive to changes in water quality (i.e. ETP taxa, Marchant et al., 1995) including stoneflies *Leptoperla australica* and several leptocerid and hydroptilid caddisflies. Altogether, the macroinvertebrate community of Gringer Creek featured a moderately diverse assemblage of taxa, of which at least a quarter are known SWA endemic species, including species from sensitive Plecoptera and Trichoptera groups.

Fish and Crayfish

A total of 275 fish of three endemic species were captured and released from Gringer Creek, including the nightfish *Bostockia porosa*, western minnow *Galaxias occidentalis* and the western pygmy perch *Nannoperca* vittata. Fyke nets could not be set at GRCK05, GringerDS09 or GringerDS11, owing to incompatible depth or channel width, and no fish were recorded at GRCK07. All three species were recorded in the 2011 survey, and there were no introduced fish detected during the current survey.

Several gravid females of the western minnow and nightfish were recorded in the upper reaches of Gringer Creek, particularly GRCK01 – 04. Both species are known to undergo annual breeding migrations in winter and spring, entering small tributary creeks and moving to the upper reaches to spawn (Pen & Potter 1990; 1991). Spawning generally occurs over a two to three month period, followed by downstream migration of adults and juveniles later in spring to riverine environments, prior to the commencement of drying in tributary creeks. It appears that Gringer Creek is being used by local fish populations (of the Bannister River, and potentially the Hotham) as breeding habitat, therefore continued connectivity between lower and upper reaches of Gringer Creek should be maintained, both during and after the construction of the RDA2. Consideration should be given for repeat monitoring of fish movement between upper and lower reaches to ensure breeding migrations continue to occur.

Long term water quality and interim SSGV development

Water quality monitoring data supplied by NBG and WRM/SLR baseline data (2011 to 2023) were examined to determine whether interim site-specific guideline values are more appropriate than ANZG (2018) DGVs for ongoing monitoring under future mine water discharge scenarios at Gringer Creek.

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. pH or oxygen) from a background or 'pre-impact' state. Analysis included calculation of median, 20th and 80th percentiles, minimum and maximum analyte concentrations, and the proportion of data above the 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.

Generally, most analytes were recorded below ANZG (2018) DGVs and the default guidelines remain appropriate. Several analytes had 80th%ile values greater than the DGV, therefore it was recommended to implement an interim SSGV rather than the default value. Further, updated guidance on some analytes is expected to be published in near future, and are adopted as interim SSGVs. A summary of proposed interim SSGVs are provided below:

- Hardness modified trigger values were derived using the minimum hardness recorded (450 mg/L) for Cd, Cr, Ni, and Zn. However, there were few hardness records available (*n* = 26).
- The DGV was retained for Al in the dry season, however, in acknowledgement of pulses of Al with the onset of winter rains, a seasonal interim SSGV is proposed, applicable to increased flow in response to rainfall.
- The interim SSGV for ammonia was derived using the 80th%ile pH (7.1) and temperature (20ºC). Using a higher value for pH and temperature (in this case, the 80th%ile rather than the median) is a more conservative approach (ANZG 2023a) and is recommended to be more protective.
- Without local DOC data, updated guidance on Cu in freshwater provide DGVs standardised at a conservatively low DOC =< 0.5 mg/L (ANZG 2023b). The standardised DGVs are likely to be overly conservative, especially given high hardness, however derivation of an operation SSGV will require regular collection of DOC data.
- The lower DGV for pH was altered to match the 20th%ile value.

6.1 Recommendations

1. Native fish including western minnow and nightfish appear to use the upper reaches of Gringer Creek as breeding habitat. Should their access to these reaches become obstructed, there may be further impacts on populations on the Bannister and Hotham River if recruitment is affected. Therefore:
   - Construction and operation of the RDA2 should maintain fish passage between the lower and upper reaches of Gringer Creek.
   - Consideration should be given to developing a fish monitoring plan to ensure fish passage is being maintained following construction of the RDA2.
2. Interim SSGVs were derived using the best available data. However, there are some data gaps to be addressed for the development of full operational SSGVs. Collection of data should start promptly and continue regularly prior to commencement of any discharge to Gringer Creek. These include:
   - Cyanide: Free cyanide needs to be analysed using LORs suitable for comparison against ecosystem protection DGVs (i.e. <0.0007 mg/L), not recreational water quality guidelines (which are an order of magnitude higher). This is especially pertinent if there is any potential for discharge water to contain cyanide.

- Dissolved organic carbon (DOC): the update to Cu guidelines provides DGVs based on a conservative DOC concentration, which has been retained as interim SSGV. Without local data, it is not possible to determine whether the interim SSGV for Cu is appropriate, or overly conservative.
- Hardness: Gringer Creek has recorded high hardness over baseline surveys. Given the role of hardness in the amelioration of toxicity for several analytes, and the input of this data in the derivation of DGVs/SSGVs, hardness should be included in regular monitoring to improve confidence in the threshold values used.
- Further investigation of surface and groundwater quality at site GRCK06, which may be a point source of Co and Mn, potentially upwelling of contaminated groundwater.

3 It is highly recommended that a risk assessment on potential impacts of mine water discharge to aquatic fauna should be carried out, once the composition of discharge is characterised. Characterisation of the discharge regime would also be beneficial to assess impacts of altered flow regimes.

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Appendix 1 Site photographs

Gringer Creek Aquatic Fauna Survey & Interim Site-Specific Guideline Values

Newmont Boddington Gold

GRCK01 - upstream

GRCK01 – downstream

GRCK02 – upstream

GRCK02 - downstream

GRCK03 – upstream

GRCK03 – downstream

GRCK04 – upstream

GRCK04 – downstream

GRCK01 - upstream

GRCK01 – downstream

GRCK05 – upstream

GRCK05 – downstream

GRCK06 – upstream

GRCK06 – downstream

GRCK07 – upstream

GRCK07 – downstream

GRCK08 – upstream

GRCK08 – downstream

GRCK01 - upstream

GRCK01 – downstream

GringerDS09 - upstream

GringerDS09 – downstream

GringerDS10 - upstream

GringerDS10 – downstream

GRCK01 - upstream

GRCK01 – downstream

GringerDS11 – upstream

GringerDS11 – downstream

GringerDS11 – roadway

Appendix 2 Habitat and substrate characteristics

Gringer Creek Aquatic Fauna Survey & Interim Site-Specific Guideline Values

Newmont Boddington Gold

Table A2.1 Percent cover of eight broad in-stream habitat types at each site on Gringer Creek. LWD = Large woody debris

Site type	Site	Mineral	Emergent Vegetation	Submerged Vegetation	Floating Vegetation	Algae	Detritus	Trailing Vegetation	LWD	Habitat Diversity Score
Upstream reference	GRCK01		-	-	-	-	80	5	15	**3**
	GRCK02	-	-	-	-	-	70	20	10	**3**
	GRCK03	20	-	30	-	-	30	10	10	**5**
	GRCK04	10	-	-	-	-	70	10	10	**4**
Impact	GRCK05	20	-	-	-	-	50	10	20	**4**
	GRCK06	15	-	5	-	-	60	10	10	**5**
	GRCK07	25	-	3	-	5	15	40	12	**6**
Potentially exposed	GRCK08	30	-	-	-	10	45	5	10	**5**
	Gringer DS09	80	-	-	-	-	10	5	5	**4**
	Gringer DS10	64	-	1	-	0	15	5	15	**5**
	Gringer DS11	5	-	-	-	0	5	90	-	**3**

Table A2.2 Percent cover of substrate particle sizes at each site on Gringer Creek. Size ranges for each particle size category given.

Site type	Site	Bedrock	Boulder <256 mm	Cobble 64-256mm	Pebble 16-64mm	Gravel 4-16mm	Sand 1-4mm	Silt <1mm	Clay	Substrate diversity score
Upstream reference	GRCK01	-	-	-	-	5	5	85	5	**4**
	GRCK02	-	-	-	-	-	5	90	5	**3**
	GRCK03	-	-	3	3	4	3	80	7	**6**
	GRCK04	-	-	-	-	-	5	90	5	**3**
Impact	GRCK05	-	-	-	-	5	5	85	5	**4**
	GRCK06	-	-	-	5	5	20	60	10	**5**
	GRCK07	-	-	-	-	10	80	10	-	**3**
Potentially exposed	GRCK08	-	-	-	-	-	95	5	-	**2**
	Gringer DS09	-	-	-	-	1	95	4	-	**3**
	Gringer DS10	-	-	-	-	-	20	20	60	**3**
	Gringer DS11	-	-	-	-	-	40	30	30	**3**

Appendix 3 Macroinvertebrates

Gringer Creek Aquatic Fauna Survey & Interim Site-Specific Guideline Values

Newmont Boddington Gold

PHYLUM/Class	ORDER	FAMILY	Lowest taxon	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
NEMATODA			Nematoda sp.	1	0	2	0	0	0	0	0	0	0	0
ANNELIDA			Oligochaeta	1	1	0	1	1	1	1	0	1	1	0
MOLLUSCA														
Gastropoda	Hygrophila	Planorbidae	*Glyptophysa* sp.	0	0	0	0	1	0	0	0	0	0	0
			Gyraulus sp.	0	0	0	0	0	0	0	0	1	0	0
ARTHROPODA														
Arachnida	Mesostigmata		Mesostigmata sp.	0	0	0	0	1	0	0	0	0	0	0
	Sarcoptiformes		Oribatida sp.	2	3	3	2	2	2	0	1	2	2	2
	Trombidiformes	Hydrodromidae	*Hydrodroma* sp.	0	0	2	0	0	0	0	0	0	0	0
		Hygrobatidae	*Australiobates* sp.	0	0	0	0	0	2	0	1	0	0	0
		Unionicolidae	*Koenikea* sp.	0	2	2	0	0	0	0	2	1	1	0
			Halacaroidea sp.	0	0	0	0	2	0	0	0	0	0	0
Collembola	Entomobryomorpha		Entomobryoidea sp.	0	0	0	2	2	1	2	0	0	0	0
	Poduromorpha		Poduroidea sp.	3	1	1	2	3	2	0	2	1	2	2
	Symphypleona		Symphypleona sp.	0	0	0	2	0	0	0	0	0	1	0
Branchiopoda	Diplostraca		Cladocera sp.	0	0	0	0	0	0	0	0	0	2	2
Malacostraca	Amphipoda	Chiltoniidae	*Austrochiltonia subtenuis*	0	0	0	0	3	4	3	4	4	3	3
			Amphipoda sp.	0	0	0	2	3	5	4	0	0	0	0
Maxillopoda			Copepoda sp.	0	3	4	2	3	0	0	3	0	2	0
Ostracoda			Ostracoda sp.	4	4	4	3	3	4	3	4	4	4	4
Insecta	Coleoptera	Chrysomelidae	Chrysomelidae sp.	0	0	0	0	0	0	0	0	0	1	0
		Dytiscidae	*Megaporus solidus*	0	0	0	0	0	0	0	0	0	1	0
			Necterosoma sp. (L)	0	0	0	0	0	0	0	0	1	2	1
			Platynectes sp. (L)	0	0	1	0	0	1	0	1	0	0	0
			Sternopriscus browni	0	0	0	0	0	0	0	0	1	1	0
			Sternopriscus marginatus	1	2	2	0	0	0	0	0	0	0	0

PHYLUM/Class	ORDER	FAMILY	Lowest taxon	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
			Sternopriscus sp. (L)	0	0	0	0	0	2	0	1	0	0	0
			Tiporus sp. (L)	0	0	0	0	0	0	0	0	0	1	0
			Haliplus fuscatus/gibbus	0	0	0	0	0	1	0	1	0	0	0
		Hydrophilidae	*Berosus* sp. (L)	0	0	0	0	0	1	0	0	0	0	0
		Scirtidae	Scirtidae sp. (L)	0	0	0	2	2	3	3	2	3	1	2
	Diptera	Cecidomyiidae	Cecidomyiidae sp.	2	0	0	0	0	0	0	0	0	0	0
		Ceratopogonidae	Ceratopogonidae sp. (P)	0	0	0	0	0	0	1	2	1	0	0
			Ceratopogoninae sp.	3	3	3	0	2	3	3	3	1	2	2
			Dasyheleinae sp.	2	0	1	0	2	0	2	1	0	2	1
			Forcipomyiinae sp.	1	0	0	0	0	0	0	0	0	0	0
		Chironomidae	Chironomidae sp. (P)	0	0	1	1	0	2	0	2	1	0	0
			Chironomus tepperi	0	0	0	2	0	0	0	0	0	0	0
			Cladotanytarsus sp.	0	0	0	0	0	0	1	0	0	0	1
			Corynoneura sp.	0	0	0	0	1	0	1	0	2	1	0
			Cryptochironomus griseidorsum	0	0	0	0	0	0	1	0	0	0	0
			Dicrotendipes sp.	0	0	0	0	2	2	0	0	0	1	0
			Larsia ?albiceps	0	0	3	0	0	0	0	2	2	0	0
			Orthocladiinae sp.	1	0	0	0	0	0	0	0	0	0	0
			Orthocladiinae sp.	0	2	0	0	0	0	0	0	0	0	0
			*Parakiefferiell*a sp.	0	0	2	0	2	3	2	0	3	2	2
			Paralimnophyes pullulus	0	0	0	0	3	4	2	3	2	1	2
			Paramerina levidensis	0	3	0	0	0	1	0	2	0	1	2
			Paratanytarsus sp.	2	2	3	0	0	0	0	0	0	0	0
			*Polypedilum nu*bifer	0	0	0	0	0	0	0	0	0	1	0
			*Procladius paludi*cola	2	4	4	1	0	1	0	3	2	2	0
			Stenochironomus sp.	0	1	0	0	0	0	0	0	0	0	0

PHYLUM/Class	ORDER	FAMILY	Lowest taxon	GRCK01	GRCK02	GRCK03	GRCK04	GRCK05	GRCK06	GRCK07	GRCK08	Gringer DS09	Gringer DS10	Gringer DS11
			Tanytarsus sp. (V6)	3	2	3	3	3	4	3	4	3	3	2
			Thienemanniella sp. (V19)	2	0	2	0	0	3	0	0	0	0	0
		Culicidae	*Culex* sp.	0	0	0	0	0	0	0	0	1	0	0
			Culicidae sp. (P)	0	0	1	0	0	0	0	0	0	0	0
		Empididae	Empididae sp.	2	0	0	0	0	1	0	0	0	0	0
		Muscidae	Muscidae sp.	1	0	0	2	0	0	0	1	0	0	0
		Simuliidae	Simuliidae sp.	0	0	0	0	0	2	2	0	0	0	1
		Stratiomyidae	Stratiomyidae sp.	0	0	0	0	0	1	0	0	0	0	0
		Tabanidae	Tabanidae sp.	1	0	1	0	0	0	0	0	0	1	0
		Tipulidae	Tipulidae sp.	1	0	1	0	0	0	0	0	0	2	0
	Hemiptera	Corixidae	*Diaprepocoris barycephalus*	0	2	0	0	0	0	0	0	0	0	0
		Gelastocoridae	*Nerthra* sp.	0	0	0	0	0	0	0	1	0	0	0
	Odonata	Aeschnidae	*Austroaeschna anacantha*	0	0	1	0	0	0	0	0	0	0	0
		Megapodagrionidae	*Archiargiolestes* sp.	0	0	0	2	0	0	1	0	0	0	0
			Anisoptera sp.	2	2	2	0	1	1	0	0	0	1	0
			Zygoptera sp.	0	0	0	2	0	1	0	0	0	0	0
	Plecoptera	Gripopterygidae	*Leptoperla aust*ralica	2	0	0	0	2	0	1	0	1	2	2
	Trichoptera	Hydroptilidae	*Hellyethira* sp.	0	0	0	0	0	3	2	2	0	0	0
			*Maydenoptila bay*nesi	0	0	0	0	0	1	2	0	0	0	0
			Oxyethira sp.	0	3	1	0	0	1	0	0	0	0	1
		Leptoceridae	*Oecetis* sp.	0	3	2	0	0	0	0	0	0	0	0
			Triplectides sp. AV21	0	0	0	1	0	0	2	2	0	2	2
			Total richness	**21**	**18**	**25**	**17**	**21**	**30**	**21**	**24**	**21**	**30**	**18**

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