

# RUM JUNGLE – REFINEMENT OF LDWQTVs

BRISBANE | PERTH | SINGAPORE | PAPUA NEW GUINEA

PREPARED FOR NORTHERN TERRITORY DEPARTMENT  
OF PRIMARY INDUSTRY AND RESOURCES (NT DPIR)



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# EXECUTIVE SUMMARY

This report complements previous studies completed at the former Rum Jungle mine. In previous reports, Hydrobiology (2013b, 2013c) identified environmental values (EVs) and set water quality objectives (WQOs) for receiving waters downstream from the former mine facilities in accordance with the ANZECC/ARMCANZ (2000) methodology. Water quality data collected up to 2015 were also analysed as part of an impact assessment (Hydrobiology, 2016). The following report presents an update of water quality data collected since the last Hydrobiology review of the dataset in 2015. Results of the temporal trend analysis indicated minor increases in aluminium and manganese in Zone 2 between 2017 and 2019. The concentrations of all other metals and metalloids investigated had decreased or remained unchanged in wet season samples since 2015. The concentrations of all other metals and metalloids investigated had decreased or remained unchanged in wet season samples since 2015 in Zones 1 (upstream) and 3 (further downstream from Zone 2).

As a former uranium mine, uranium and selenium levels have been historically elevated downstream of the mine as the metals/metalloids formerly sequestered in the subsurface have been mobilised. Despite this, no guidelines were proposed for uranium and selenium during earlier studies because they were not identified as priority substances of concern. This report addressed this gap by deriving locally relevant site-specific water quality objectives (LDWQOs) appropriate for the unique characteristics of the site for uranium and the most conservative and applicable international guidelines for selenium. The new LDWQOs for U were based on recent toxicity data derived in locally relevant areas and are summarised per zone in Table ES 1. The U LDWQOs remain interim due to the limited dissolved organic carbon data available to date and required to refine the LDWQOs. It is recommended that a minimum of 18 data values be collected over 2 years to derive final LDWQOs (as per DES, 2018a).

Table ES 1 LDWQOs for selenium and uranium at Rum Jungle

Zone	Se LDWQO (µg/L)	Interim U LDWQO (µg/L)
<b>Zone 1</b>	2	2.8
<b>Zone 2</b>	2	31*
<b>Zone 3</b>	2	22.5*
<b>Zone 4</b>	2	13.2
<b>Zone 5</b>	2	2.7*
<b>Zone 6</b>	2	2.9
<b>Zone 7</b>	2	2.7*

Notes: \* low-reliability LDWQO as defined by ANZG (2018)

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# 1. INTRODUCTION

## 1.1 BACKGROUND

The former Rum Jungle Mine site was mined in the 1950s-1970s, then rehabilitated during the 1980s. Monitoring of landform stability and water quality has continued since that time. The current project (under a Partnership Agreement between the Northern Territory and Commonwealth Governments) aims to provide a more permanent reduction in environmental impacts from the site due to acid and metalliferous drainage (AMD) by adopting leading practice rehabilitation methods.

In 2012, Hydrobiology completed a study that identified environmental values (EVs) and set appropriate water quality objectives (WQOs) in accordance with the ANZECC/ARMCANZ (2000) methodology (Hydrobiology 2013a, 2013b). This involved setting of EVs and WQOs and defining a monitoring plan and impact assessment that would provide support for the proposed EVs and WQOs, and inform the development of locally derived water quality objectives (LDWQOs) in accordance with relevant frameworks and guidelines.

As a former uranium mine, uranium and selenium levels have been historically elevated downstream of the mine as the metals/metalloids formerly sequestered in the subsurface have been mobilised. Despite this, no guidelines were proposed for uranium and selenium during earlier studies as they were not identified as priority substances of concern compared with other elements and salts mobilised by the AMD. This report aims to address this gap by deriving site specific water quality objectives appropriate for the unique characteristics of the site.



## 1.2 PROJECT LOCATION

The former Rum Jungle mine is located approximately 105 km (by road) south of Darwin, near Batchelor in the Northern Territory. The site is accessed from Batchelor Road, Rum Jungle Road and Litchfield Park Road via the Stuart Highway from Darwin. The water quality monitoring sites used for the project are listed in Table 1-1 and their locations are shown in Figure 1-1. Note that by convention in this report the east branch of the Finniss River is referred to simply as the East Branch. Also shown in the table and on the figure are the locations of river zones that were defined by Hydrobiology (2013a) for the purpose of setting water quality objectives. Those zones are used in this report to refer to groups of sites with differing levels of mine site -related inputs and dilution of them, because the zone boundaries are defined by sources of inputs and by major tributary junctions that will afford some dilution and geochemical alteration of waters from the mine lease area.

Table 1-1 Sampling sites used for the current study

Site Code	Site Name	Zone
<b>EB@LB</b>	East Branch at Mine Boundary	1
<b>FC@LB</b>	Fitch Creek at Mine Boundary	1
<b>EB@G_Dys</b>	East Branch at Dyson's gauging station	2
<b>EB@GS200</b>	East Branch at gauging station GS8150200	2
<b>TC@LB</b>	Tailings Creek at Mine Boundary	2
<b>EB@GS327</b>	East Branch at gauging station GS8150327	3
<b>EB@GS097</b>	East Branch at gauging station GS8150097	3
<b>EBusFR</b>	East Branch upstream of the Finniss River Confluence	4
<b>FRUSMB</b>	Finniss River Upstream Mount Burton mine	5
<b>FRDSMB</b>	Finniss River Downstream Mount Burton mine	5
<b>FR@GS204</b>	Finniss River at gauging station GS8150204	6
<b>FRusFC</b>	Finniss River upstream of Florence Creek	6
<b>FRdsFC</b>	Finniss River downstream Florence Creek	7

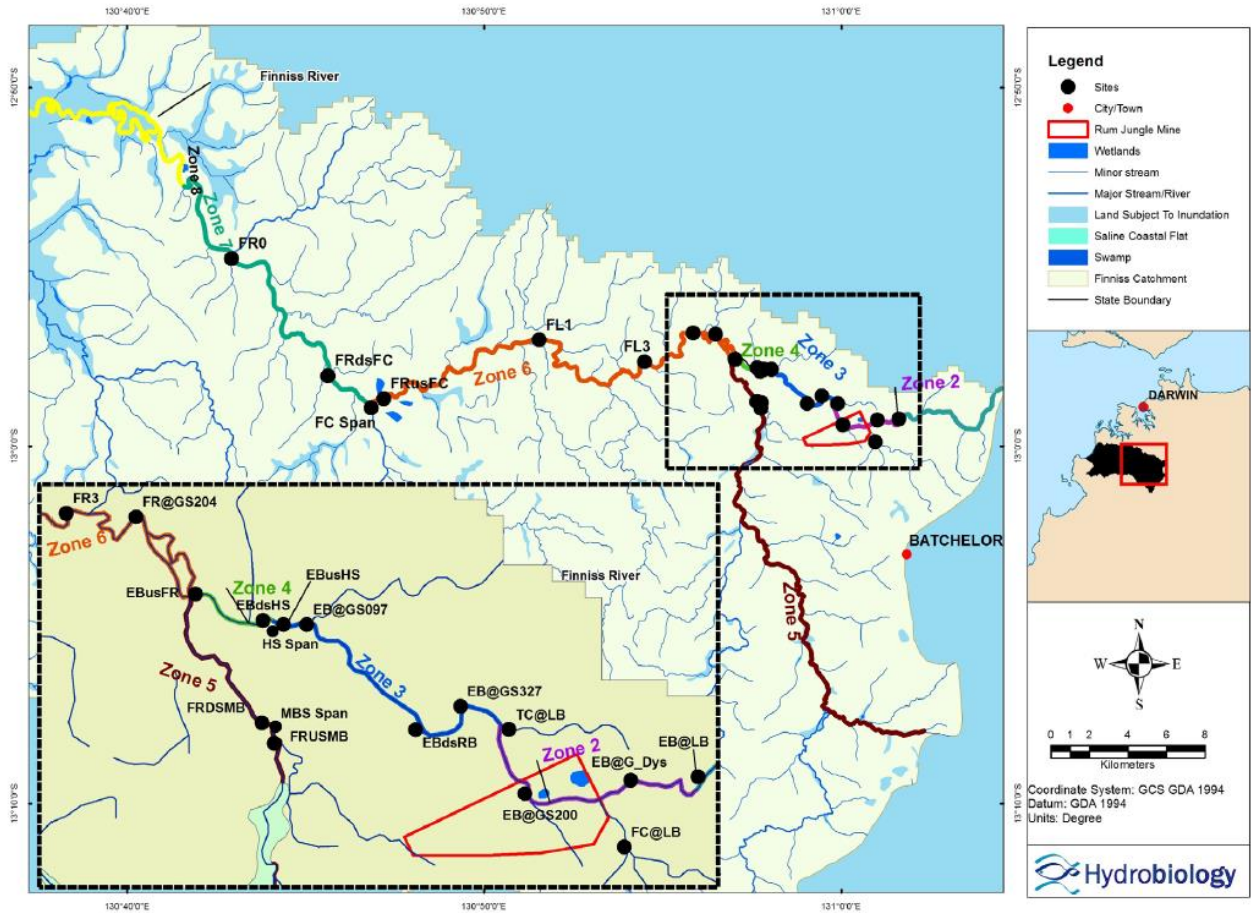


Figure 1-1 Location of key monitoring sites used for the current study

### 1.3 SCOPE

The specific objectives of this report were to:

- Review water quality data collected since the last Hydrobiology review of the dataset in 2015;
- Undertake similar assessment of the water quality characteristics to those provided in the 2015 impact assessment report (Hydrobiology, 2016); and
- Undertake refinement of LDWQOs including the development of LDWQOs for Selenium and Uranium.

# 2.

# TOXICITY PROFILES

A review of published and grey literature on the toxicity of Se and U was performed. This review aimed to identify their respective environmental behaviour, with a particular emphasis on their characteristic bioavailability in aquatic environments, and expected toxicity effects and concentrations reported around the Northern Territory and elsewhere.

## 2.1 SELENIUM

### 2.1.1 SELENIUM IN NATURAL FRESHWATER

Most Se in surface water is present as selenate or selenite (Maher *et al.*, 2010). Selenate tends to predominate in lotic (flowing water) systems, while the more reduced selenite may predominate in lentic (standing water) systems (Stewart *et al.*, 2010). Further reduced organic Se species, such as selenomethionine, may also occur in lentic systems. Organic Se species tend to be much more bioaccumulative than inorganic Se species, and selenite tends to be more bioaccumulative than selenate (Besser *et al.*, 1993). As such, Se bioaccumulation potential is highly dependent on Se speciation, which is highly dependent on site-specific water body characteristics.

### 2.1.2 EFFECT OF ENVIRONMENTAL PARAMETERS ON SELENIUM

Algae and other primary producers convert the inorganic Se taken up from the water to organic Se species that are then passed through the food web via trophic transfer. The most significant Se bioaccumulation step therefore occurs at the base of aquatic food webs. Selenium bioconcentration factors (BCFs), calculated as the ratio of Se concentrations in algae and water, may range from about 100 to >10,000 L/kg dry weight (dw) (Presser and Luoma, 2010; Stewart *et al.*, 2010). The high variability in BCFs reflects the importance of site-specific Se speciation and food web characteristics.

Trophic transfer factors (TTFs), calculated as the ratio of Se concentrations in a consumer organism to its food, typically range from <1 to about 2 for most invertebrates and fish (on a whole-body basis) (Presser and Luoma, 2010). As such, although trophic transfer is important for Se, it does not biomagnify at higher trophic levels.

### 2.1.3 SELENIUM TOXICITY

In aquatic systems, fish and birds are the most sensitive organisms to Se, while invertebrates and primary producers tend to be relatively insensitive (Janz *et al.*, 2010; USEPA, 2016). The critical exposure route for fish and birds is to diet-borne organic Se. In fish, maternally transferred Se to the ovaries and eggs can result in larval mortality, deformities, and oedema. Similarly, in birds, Se transferred to eggs can impair hatchability and cause embryo deformities. Given the greater sensitivity of fish and birds to Se relative to lower trophic level organisms, most regulatory Se guidelines are based on protection of these organisms (BCMOE, 2014; USEPA, 2016).

## 2.2 SELENIUM GUIDELINE VALUES

Because the potential for Se toxicity to fish and birds occurs via diet-borne exposures, and diet-borne Se concentrations are highly dependent on site-specific characteristics, most regulatory Se guidelines are not derived based on the traditional approaches used for other chemicals (e.g., water-only toxicity tests). Rather, water Se guidelines have been developed based on Se exposure and effects data from case studies or, more recently, guidelines have been developed that are expressed as Se concentrations in fish tissue or bird eggs. The following sections summarise freshwater Se guidelines in Australia and New Zealand, Canada, and the United States, the corresponding guideline values are shown in Table 2-1.

Table 2-1 Summary of Se guidelines in Australia and New Zealand, Canada, and the United States.

Jurisdiction	Water Se (µg/L)	Dietbourne Se (mg/kg dw)	Fish (mg/kg dw)			Birds (mg/kg dw)
			Eggs/Ovaries	Muscle	Whole Body	Eggs
<b>Australia and New Zealand</b>	5 (99% protection) 11 (95% protection) 18 (90% protection) 34 (80% protection)	-	-	-	-	-
<b>Canada (BC)</b>	1 (alert)* 2 (guideline)	4 (invertebrates)	11	4	4	6
<b>United States</b>	1.3 (lentic) 3.1 (lotic) Intermittent <sup>1</sup>	-	15.1	11.3	8.5	-

BC = British Columbia

dw = dry weight

- = not available

\* Alert is a concentration below the guideline but above which there may be a risk to some environments and/or species that are particularly sensitive to Se bioaccumulation.

1 The lentic and lotic criteria are expressed as 30-d average concentrations. USEPA (2016) also provided an intermittent criterion equation that would allow for short-term exceedance of the lentic and lotic criteria without exceedance by the 30-d average: Intermittent Criterion = [30-d Criterion - Cbackground (1 - fintermittent)] / fintermittent, where 30-d Criterion is the lentic or lotic criterion of 1.3 and 3.1 µg/L, Cbackground is the average background concentration, and fintermittent is the fraction of any 30-d period during which elevated selenium concentrations occur.

## 2.2.1 AUSTRALIA AND NEW ZEALAND GUIDELINE VALUES

Australia and New Zealand have default guideline values for Se that were last updated in 2000 (ANZG, 2018). These guidelines range from 5 to 34 µg/L, depending on the targeted level of protection (Table 2-1). The guidelines are based on percentiles of the species sensitivity distribution (SSD), with the lower percentiles driven by a no-observed-effect concentration (NOEC) for a crustacean. The SSD was based on water-only exposures. ANZG notes that bioaccumulation and secondary poisoning effects should be noted for Se, and that the reliability of the default guideline values for Se is low (ANZG, 2018).

## 2.2.2 INTERNATIONAL GUIDELINE VALUES

### CANADA

The British Columbia Ministry of Environment (BCMOE) finalised water quality guidelines for Se in 2014 (BCMOE, 2014). Guidelines were developed for (1) surface water; (2) fish diets; (3) fish eggs/ovaries; (4) fish muscle and whole-body; and (5) bird eggs (Table 2-1). Various approaches were used in developing each of these. The water Se guideline of 2 µg/L was identified as a concentration that would protect most waters based on a weight-of-evidence evaluation from several field and laboratory studies. The fish egg/ovary Se guideline of 11 mg/kg dw was based on the mean Se EC10 (10% effect concentration) for west slope cutthroat trout and rainbow trout, which was divided by an uncertainty factor of 2. The diet-borne and fish muscle and whole-body Se guidelines of 4 mg/kg dw are based on a weight-of-evidence evaluation of EC10s and no-effect concentrations for several species. Lastly, the bird egg Se guideline of 6 mg/kg dw was derived from an EC10 of 12.5 mg/kg dw for reduced hatchability in mallard and application of an uncertainty factor of 2.

### UNITED STATES

The U.S. Environmental Protection Agency (USEPA) recently finalised new ambient water quality criteria for Se (USEPA, 2016). The new criteria include the following elements: (1) a fish egg/ovary criterion; (2) a fish muscle criterion; (3) a whole-body fish criterion; and (4) surface water criteria for lentic and lotic waters (Table 2-1). The egg/ovary criterion supersedes the muscle and whole-body criteria, which in turn supersede the water Se criteria.

The egg/ovary criterion of 15.1 mg/kg dw is based on Se toxicity tests from which relationships between toxicity to larvae were linked to egg/ovary Se concentrations. Toxicity data were available for nine fish species and eight genera and expressed as EC10s. The criterion of 15.1 mg/kg dw was the 5th percentile of the genus-mean EC10s. The muscle and whole-body Se criteria of 11.3 and 8.5 mg/kg dw, respectively, were derived from the same tests used to develop the egg/ovary Se criterion, with muscle and whole-body Se concentrations estimated using tissue conversion factors. Lastly, the water Se criteria of 1.3 and 3.1 µg/L for lentic and lotic water bodies, respectively, were back-calculated from the fish tissue-based criteria based on Se bioaccumulation models for a large number of lentic and lotic water bodies. The water criteria are based on conservative percentiles of the back-calculated water Se concentrations. As provided in Table 2-1, the USEPA also provides an intermittent selenium criterion equation that accounts for short-term exceedances of the lentic and lotic criteria.

## 2.2.3 SITE-SPECIFIC GUIDELINE VALUES

Given the low concentrations of Se detected within the Finnis River system monitoring data (see below) and low reliability of the current ANZG guidelines, it is recommended that the most conservative and appropriate international approach be adopted as the site-specific guideline value, as discussed in Section 4.2.

## 2.3 URANIUM

### 2.3.1 URANIUM IN NATURAL FRESHWATER

Uranium (U) is a naturally occurring element but can also be introduced into the environment by anthropogenic activities which could result in elevated levels of U that potentially cause adverse environmental impacts. Uranium generally occurs in natural waters in three oxidation states of uranium IV ( $U^{4+}$ ), uranium V ( $UO_2^+$ ) and uranium VI ( $UO_2^{2+}$ ). In oxidised surface waters, uranium occurs as uranium VI ( $UO_2^{2+}$ ) and forms stable, readily soluble, cationic, anionic and/or neutral complexes which are highly mobile (Langmuir, 1978; Osmond and Ivanovich, 1992). However, in reducing surface waters, uranium normally occurs as uranium IV ( $U^{4+}$ ) and V ( $UO_2^+$ ). Uranium (VI) forms complexes with inorganic and

organic ligands that reduce its concentration and ameliorate the biological uptake and its toxicity of uranium. A study by Langmuir (Langmuir, 1978) observed that  $U^{4+}$  has a strong tendency to precipitate and to remain immobile, whereas  $UO_2^+$  forms soluble, but relatively unstable complexes.

### 2.3.2 EFFECT OF ENVIRONMENTAL PARAMETERS ON URANIUM

Speciation of uranium and its abundance in natural waters is governed by redox potential, pH, suspended particles, organic and inorganic ligands (e.g. phosphates and carbonates) and dissolved organic carbon. The redox and complexation reactions of uranium in surface waters are strongly influenced by hydrolysis because hydrolytic reactions may limit the solubility or influence sorption behaviour (Choppin and Stout, 1989). In oxidised fresh surface waters with pH ranges from 5 to 9, the speciation of uranium is relatively complex (Grenthe *et al.*, 1992; Palmer and Nguyen-Trung, 1995).

Uranium speciation is highly pH-dependent. For example, in waters containing an environmentally relevant concentration of  $10 \mu\text{g L}^{-1}$  dissolved U, at  $\text{pH} \leq 5.0$  the predominant species is free hydrated uranyl ion ( $UO_2^{2+}$ ) but the concentration of this species becomes insignificant when pH increases to greater than 6.0 (Grenthe *et al.*, 1992; Markich *et al.*, 1996). The second most dominant species at pH 5 is  $UO_2OH^+$ , which increases in importance up to pH 6 (Grenthe *et al.*, 1992; Markich *et al.*, 1996). Polymeric uranyl-hydroxide complexes including  $(UO_2)_2(OH)_2^{2+}$ ,  $(UO_2)_3(OH)_5^+$ ,  $(UO_2)_4(OH)_7^+$  and  $(UO_2)_3(OH)_7^-$  formation increase at  $\text{pH} \geq 5.0$ , particularly at higher U concentrations (Grenthe *et al.*, 1992; Markich *et al.*, 1996).

In natural water, it is known that uranyl ions form stable complexes with dissolved organic matter (DOM) (Choppin and Sullivan, 1992). Insoluble uranyl-DOM complexes may reduce the bioavailability and toxicity of U to aquatic organisms by acting as a sink for U (Brown *et al.*, 1994), while soluble uranyl-DOM complexes contribute to the migration of uranyl ions in water (Moulin *et al.*, 1992). In organic-rich freshwaters that have a low hardness and alkalinity ( $\text{pH} 5.7$ ), the uranyl-DOM complexes are considered the dominant species of dissolved U (Markich *et al.*, 1996). However, when the hardness, alkalinity and pH (usually  $\text{pH} > 7.8$ ) of the water increases, there is a shift in speciation where uranyl-carbonate and uranyl-hydroxide-carbonate species become more important than uranyl-DOM complexes (Moulin *et al.*, 1992).

The most important inorganic complexing agent of uranyl ions is carbonate due to the formation of very stable complexes (Greene *et al.*, 1986). In moderate to hard waters (i.e. hardness and alkalinity  $> 60 \text{ mg CaCO}_3 \text{ L}^{-1}$ ) at  $\text{pH} 5.6$ ,  $UO_2CO_3$  is the dominant species, while at  $\text{pH} 6.8$ ,  $UO_2(CO_3)_3^{4-}$  is the dominant species. The complexation of uranyl by chloride, sulfate, nitrate and silicate is considered relatively weak compared with uranyl complexes with carbonate and phosphate in freshwaters (Gascoyne, 1992). Uranyl-phosphate complexes only start to become significant when the concentration of phosphate approaches  $100 \mu\text{g/L}$  (Langmuir, 1978).

Sorption of U to clay minerals below pH 5, and iron and aluminium (oxy)hydroxides, silica and micro-organisms at higher pH, reduces the mobility of uranium in oxic waters (Prikryl *et al.*, 1994; Waite *et al.*, 1994; Kohler *et al.*, 1996; Turner *et al.*, 1996). In this process uranium absorbed to insoluble organic matter, or organic matter attached to particles also reduces the mobility of uranium (Pompe *et al.*, 1996). It is generally established that sorption of uranium to particles increases with increasing pH until a threshold point is reached around pH 6–8 (Dzombak and Morel, 1990; Choppin and Sullivan, 1992; Willett and Bond, 1995).

### 2.3.3 URANIUM TOXICITY

Uranium is a non-essential element for biological processes and is generally toxic at elevated concentrations (Berlin and Rudell, 1979). Uranium is highly soluble and mobile in most natural waters (see above and Morse and Choppin, 1991) therefore, contaminated waters from local uranium activities



are a potential hazard to aquatic biota. A study by Markich *et al.* (1996) reported that  $\text{UO}_2^{2+}$  (a species of uranium VI) is primarily responsible for eliciting a toxic response in aquatic organisms. Markich *et al.* (2000) also provided evidence suggest that  $\text{UO}_2^{2+}$  and, to a lesser extent,  $\text{UO}_2\text{OH}^+$  are the U species that contribute most to the toxic response observed in aquatic biota, where  $\text{UO}_2^{2+}$  has approximately twice the effect of  $\text{UO}_2\text{OH}^+$ .

The mechanisms of U toxicity in water is similar to that of other metals. Aquatic organisms integrate their excretion and storage processes to manage metal uptake. Some organisms are able to regulate the levels of a particular metal in their bodies independently of environmental concentrations, while others accumulate the metal in their bodies, detoxifying when necessary (Hyne *et al.*, 1993). Uranium accumulates in nematocysts of hydra and inhibits the replacement of discharged nematocysts, resulting in feeding dysfunction and reduced population growth (Hyne *et al.*, 1993). The walls of the nematocyst capsules are collagenous in nature and may have an affinity for U (Blanquet and Lenhoff, 1966; Anselme *et al.*, 1990). The mechanism of U toxicity in higher animals may be attributed to changes in cellular membrane permeability due to the binding of uranyl ions to phosphate ligands and to the inhibition of cellular carbohydrate metabolism (Ellender *et al.*, 1992). The principal effect is the inactivation of phosphate-containing molecules and biological ligands such as ATPase (Ellender *et al.*, 1992).

A number of studies have established that uranium toxicity is inversely related to water hardness and alkalinity (Tarzwell and Henderson, 1960; Parkhurst *et al.*, 1984; Poston *et al.*, 1984). Parkhurst *et al.* (1984) reported that the 96-hour LC50 for brook trout (*Salvelinus fontinalis*) was 5.5  $\mu\text{g/L}$  in soft water (hardness, 35 mg/L as  $\text{CaCO}_3$ ; alkalinity, 11 mg/L as  $\text{CaCO}_3$ ; pH, 6.7). In contrast it was 23  $\mu\text{g/L}$  in hard water (hardness, 208 mg/L as  $\text{CaCO}_3$ ; alkalinity, 53 mg/L as  $\text{CaCO}_3$ ; pH, 7.5).

Probably the most important complexing agent for uranium in oxidised freshwaters is carbonate (Clark *et al.*, 1995). Markich *et al.* (1996) showed that the toxicity of uranium to a freshwater bivalve (*Velesunio angasi*) was inversely proportional to alkalinity, where both pH and water hardness were held constant. Complexes of uranium with carbonate are less toxic than  $\text{UO}_2^{2+}$  (Nakajima *et al.*, 1979; Poston *et al.*, 1984; Greene *et al.*, 1986). Phosphate is an important complexing agent when its concentration approaches 75  $\mu\text{g/L}$  (Langmuir, 1978).

Several studies have shown that the uptake and toxicity of uranium is inversely related to pH, over the range 2 to 7, where both water hardness and alkalinity were held constant (Nakajima *et al.*, 1979; Greene *et al.*, 1986; Markich *et al.*, 1996). Markich *et al.* (1996) showed that the sublethal toxicity of uranium to the bivalve *Velesunio angasi* in a synthetic water was about five times greater at pH 5 (48-hour EC50 = 117  $\mu\text{g/L}$ ) than at pH 6 (48-hour EC50 = 634  $\mu\text{g/L}$ ). They concluded that changes in uranium speciation were responsible for the changes in toxicity of uranium.

## 2.4 URANIUM GUIDELINE VALUES

River and stream surface waters in tropical Australia, particularly the Northern Territory, typically contain less than 1  $\mu\text{g L}^{-1}$  U (Hart *et al.*, 1987, Markich, 1998). The toxicity of U to freshwater biota has been studied using tropical Australian freshwater organisms from several phyla including Chlorophyta (Trenfield *et al.*, 2011; van Dam *et al.*, 2012), Chordata (Holdway, 1992; Cheng *et al.*, 2010), Cnidaria (Hyne *et al.*, 1992; Markich and Camilleri, 1997; N Riethmuller *et al.*, 2000; Riethmuller *et al.*, 2001; Trenfield *et al.*, 2011; van Dam *et al.*, 2012), Crustacea (Hyne *et al.*, 1993; Semaan *et al.*, 2001; van Dam *et al.*, 2012) and Mollusca (Hogan *et al.*, 2010).

### 2.4.1 AUSTRALIA AND NEW ZEALAND GUIDELINE VALUES

The Australian and New Zealand Guidelines for Fresh and Marine Water Quality provides default guideline values (DGVs) as a generic starting point for assessing water quality to protect aquatic ecosystems for a range of water types (ANZG, 2018). It also emphasises that site-specific guideline



values for physico-chemical and toxicant indicators should be derived and used in preference to DGVs. A current interim GV for uranium of 0.5 µg/L with unknown reliability and level of species protection is recommended as a starting point for assessing water quality to protect aquatic ecosystems. Site-specific GVs will incorporate data relevant to local conditions and organisms, provide a higher level of confidence that the GV will protect the aquatic ecosystem at a site compared to DGVs.

## 2.4.2 SITE-SPECIFIC GUIDELINE VALUES

The site-specific GVs can be based on local biological effects data relevant to the site of interest and can be acquired from either laboratory based single-species toxicity tests, laboratory or field micro- or mesocosm experiments, or field biological data (van Dam *et al.*, 2014). This derivation method involves constructing a cumulative probability distribution of chronic toxicity data of a species sensitivity distribution (SSD) and selecting a specific centile as the GVs.

In 2005, the GV of 6 µg/L for U for the Magela Creek system was derived from a SSD based on the chronic toxicity data from a minimum data requirement of 5 local species from 4 taxonomic groups, represented the 1<sup>st</sup> percentile of the SSD or the concentration predicted to protect at least 99% of species a high ecological value aquatic ecosystem (Hogan *et al.*, 2005). Since then, new toxicity data and increased knowledge have been acquired, including U toxicity data for 3 new local species, *Lemna aequinoctialis* (Hogan *et al.*, 2010), *Ceratophyllum demersum* (Markich, 2013), and *Amerianna cumingi* (Mooney *et al.*, 2016). Additional U toxicity data for 3 species, *Chlorella* sp., *Hydra viridissima*, and *Mogurnda mogurnda* (Cheng *et al.*, 2010; Trenfield *et al.*, 2011; van Dam *et al.*, 2012) were already available.

In 2012, a study by van Dam *et al.* investigated 46 existing U chronic toxicity datasets and found that DOC was the strongest determinant of IC10 concentrations. The study also developed an algorithm to enable the adjustment of U toxicity values and U hazard estimates based on local species data to modify U GVs according to environmental DOC concentrations. It has been recognised that GVs derived from SSDs using data from a small dataset, can only ever be classified as low to moderate reliability, and the generation of data for more species is encouraged (Warne *et al.* 2015). When high confidence in GVs is a requirement, consideration of site-specific conditions is needed.

A recent study by van Dam *et al.* (2017) provided a site-specific (SS) GV of 2.8 µg/L U to protect 99% freshwater species for Magela and Gulungul creeks. The study also provided an ability to adjust the GVs on the basis of environmental concentrations of DOC from 2 to 20 mg/L with DOC-modified U GVs (van Dam *et al.*, 2017). The revised GV has been adopted as the statutory limit, with the regulatory framework structured so the GV requires adjustment based on DOC concentration only when an exceedance occurs.

# 3. METHODS TO REFINE THE LDWQOS

## 3.1 DATA REVIEW AND TEMPORAL TREND ANALYSIS

Temporal water quality data from 2015 onwards were analysed to identify any increasing or decreasing trends in measured parameters for which data were available. Only data from the wet season were selected for this analysis. This was to avoid seasonal bias, in particular expected increased elevations during the dry season and first flush due to evapoconcentration. A Spearman rank correlation was done to identify any substantial monotonic changes. However, the results were inconclusive due to the small sample sizes available. Any potential trends indicated by the statistical analyses were confirmed or otherwise using graphical representation of the data.

## 3.2 DERIVATION OF LDWQOs FOR SELENIUM AT RUM JUNGLE

No new site-specific WQOs could be derived for Se using the available data. Refer to Section 4.2.

## 3.3 DERIVATION OF LDWQOs FOR URANIUM AT RUM JUNGLE

The site-specific WQO derivation process for U at Rum Jungle was based on the site-specific water quality characteristics and ecotoxicity data relevant to the protection of the ecosystems of Magela and

Gulungul creeks (van Dam *et al.*, 2017) and incorporated DOC correction using the following equation (van Dam *et al.*, 2017, modified from van Dam *et al.*, 2012):

$$WQO_{U(x)} = \frac{GV_{U(2)}}{(1 + 2 \times \text{chronic factor})} \times (1 + x \times \text{chronic factor})$$

where

- $WQO_{U(x)}$  is the newly derived water quality objective for uranium in surface water corrected to a site-specific DOC concentration of  $x$  mg/L;
- $GV_{U(2)}$  is the default GV value of 2.8  $\mu\text{g(U)}/\text{L}$  at 2 mg/L DOC (corresponding to 99% species protection);
- $x$  is the site-specific DOC concentration in mg/L; and
- *chronic factor* represents the generic slope for the effect of DOC on chronic toxicity, defaulted to 0.090 (van Dam *et al.*, 2012).

As the ecosystems of the Finnis River feature considerable taxonomic overlap with Magela and Gulungul Creeks, the use of the above approach was assessed to be appropriate after adjustment for the water quality characteristics of the Finnis River system. The water quality monitoring program at Rum Jungle has investigated DOC concentrations at relevant sites on few occasions to date. Based on the limited DOC dataset provided,  $x$  was set at the 10<sup>th</sup> percentile of data reflecting a conservative estimate of typical DOC concentrations observed for each zone (Table 4-1). For DOC concentrations below the laboratory limits of detection of 1 mg/L, half of that concentration (0.5 mg/L) was used for calculations of the percentile.

Table 3-1 Summary of uranium LDWQOs derived for Ranger mine creeks (van Dam *et al.*, 2017)

Level of protection (% species)	Guideline value Uranium concentration ( $\mu\text{g}/\text{L}$ )	Confidence interval [lower 95% CI; upper 95% CI] ( $\mu\text{g}/\text{L}$ )
99%	2.8	[1.1 – 18]
95%	8.3	[4.6 – 28]
90%	13	[8.5 – 40]
80%	23	[15 – 71]
75%*	27	[17 – 74]
70%*	32	[20 – 81]

Notes: Data extracted from the supplementary report 1 of van Dam *et al.* (van Dam *et al.*, 2017); \*the 75 and 70% levels of protection were not calculated in the original publication, they were derived for this report by entering raw data from van Dam *et al.* (van Dam *et al.*, 2017) in Burrlioz 2.0 program, as per the methods used in van Dam *et al.* 2017 (i.e. log logistic model).

# 4. DERIVATION OF LDWQOS

## 4.1 TEMPORAL TRENDS

The temporal trend analysis was limited by a lack of comparable water quality data (wet season only) at many of the sites in the period investigated between 2015 and 2019. The compounds investigated included aluminium (Al), arsenic (As), cadmium (Cd), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), manganese (Mn), nickel (Ni), selenium (Se), uranium (U) and zinc (Zn), where possible. Sites for which sufficient data were available were FC@LB and EB@LB in Zone 1 (2015 and 2017-2019), EB@G Dys in Zone 2 (2017 to 2019) and TC@LB in Zone 3 (2017 to 2019).

The trends discussed below focus on compounds for which data were available and above laboratory detection limits. It is noted that Cd and Se were frequently reported below laboratory detection limits at the selected sites investigated, however Cd had been detected in water from other sites located in Zone 2 and 3. Se was only analysed from 2017 onwards.

Spearman rank correlation tests did not reveal any substantial temporal trends, although the statistical power of the analyses were limited by the small sample sizes. Graphical exploration of the data revealed:

- decreases in some metals in Zone 1 from 2015 to 2019 (Figure 4-1):
  - zinc (Zn) and manganese (Mn) decreased at site FC@LB

- median Zn decreased at site EB@LB;
- minor increases in aluminium (Al) and Mn at Zone 2 between 2017 and 2019 (Figure 4-2); and
- decreases in cobalt (Co), copper (Cu), nickel (Ni) and Zn in Zone 3, particularly from 2018 to 2019 (Figure 4-4).

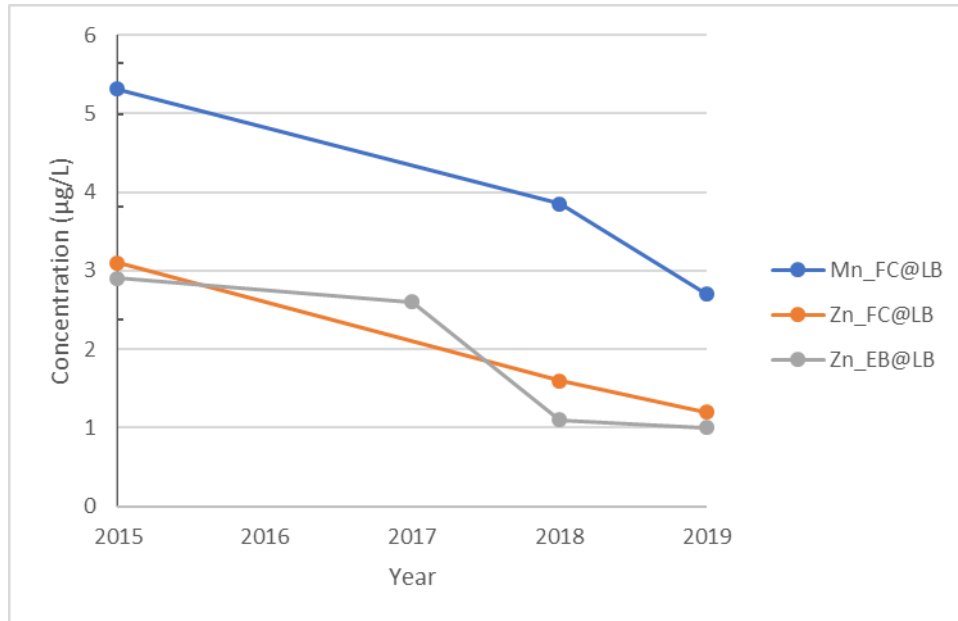


Figure 4-1 Median concentrations of manganese (Mn) and zinc (Zn) at Zone 1 locations, absence of error bars indicates one sample only

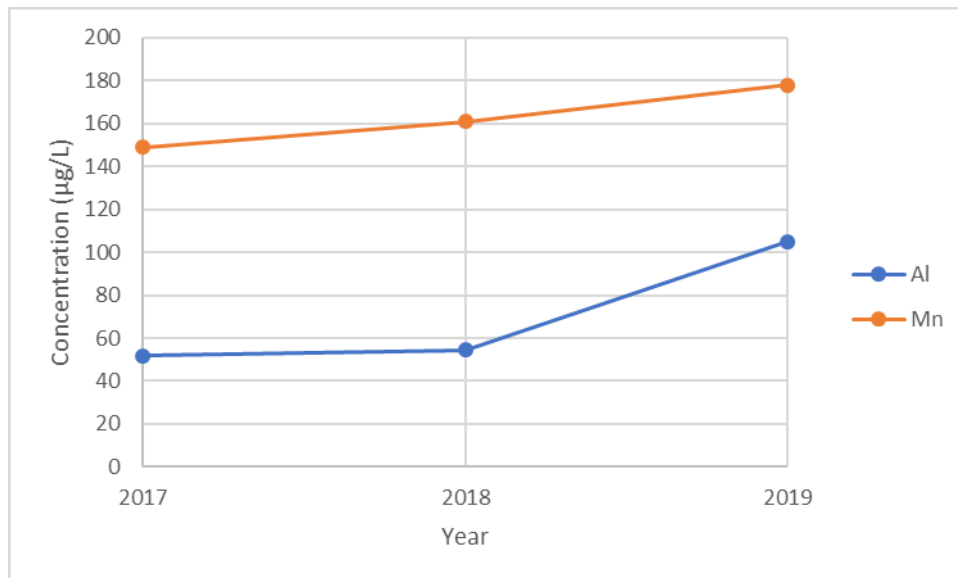


Figure 4-2 Concentrations of manganese (Mn) and zinc (Zn) at site EB@G\_Dys, Zone 2

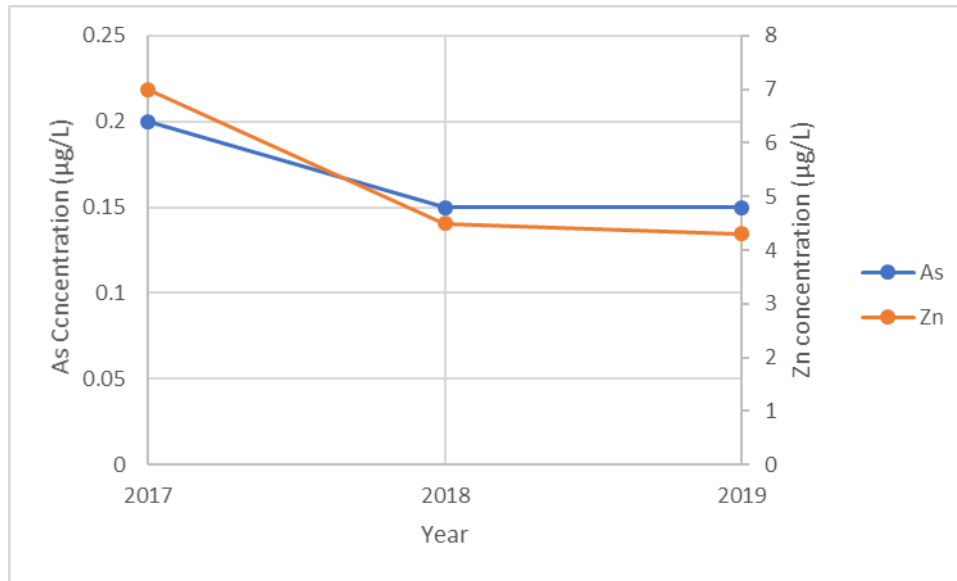


Figure 4-3 Concentrations of arsenic (As) and zinc (Zn) at site EB@G\_Dys, Zone 2

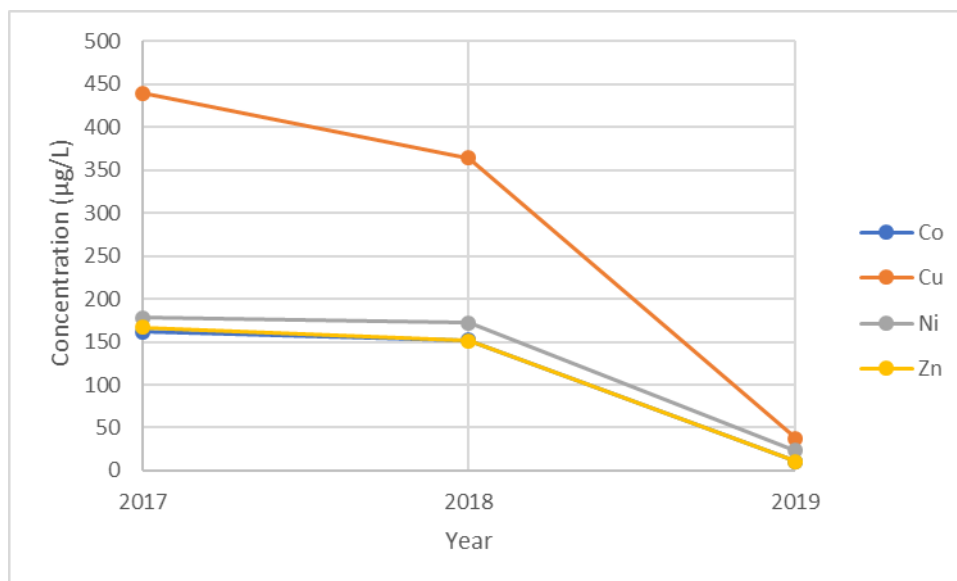


Figure 4-4 Concentrations of cobalt (Co), copper (Cu), nickel (Ni) and zinc (Zn) at site TC@LB, Zone 3

## 4.2 LDWQOS FOR SELENIUM AT RUM JUNGLE

Given that the reliability of ANZG default guidelines for Se are low and the concentrations of Se within the Finnis River system monitoring data are also low, it is recommended that the most conservative international guideline be used as the site specific GV for Se. In this instance, the most conservative and appropriate guideline of 2 µg/L (BCMOE, 2014) is recommended, considering that the USEPA guideline of 1.3 µg/L is for lentic water specifically. As most regulatory Se guidelines are not derived based on the traditional approaches used for other chemicals and have been developed based on concentrations in fish tissue or bird eggs, a minimum requirement of monitoring fish flesh Se

concentration to ensure that this LDWQO is appropriately protective is recommended during and after rehabilitation efforts.

### 4.3 LDWQOS FOR URANIUM AT RUM JUNGLE

New interim LDWQOs were derived for U in each zone. They are summarised in Table 4-1. It appears that the DOC concentrations in all zones investigated were closer to the lower end of the validated GV range of 2-20 mg(DOC)/L, with many values below 2 mg/L. The calculated WQOs are considered interim at this date as limited seasonal DOC data were available. WQOs derived for zones with DOC below 2mg/L are the product of extrapolations, therefore they are considered ‘low reliability’ values.

Table 4-1 Interim LDWQOs for uranium at Rum Jungle

Zone	DOC concentration (mg/L)			Prescribed level of protection based on environmental values (EVs)	Interim U WQO (µg/L)
	Number of values	Range observed	<i>x</i> (10 <sup>th</sup> percentile)		
<b>Zone 1</b>	2	2.2 - 2.5	2.23	SMD (99% species)	2.8
<b>Zone 2</b>	5	1.4 – 2.4	1.60	70% species	31*
<b>Zone 3</b>	8	1.1 – 7.0	1.73	80% species	22.5*
<b>Zone 4</b>	2	2.0 – 4.3	2.23	90% species	13.2
<b>Zone 5</b>	22	<1.0 – 5.0	1.91	SMD (99% species)	2.7*
<b>Zone 6</b>	16	2.2 – 4.3	2.55	SMD (99% species)	2.9
<b>Zone 7</b>	6	1.9 – 4.0	1.95	HCV (99% species)	2.7*

Notes: \* low reliability value *x* outside the DOC range of 2 – 20 mg/L of the default GV; SMD stands for ‘slightly to moderately disturbed’ systems – this value was not defined for U in ANZG (2018) therefore a conservative 99% species protection level has been adopted; HCV stands for ‘high conservation value’.

# 5. CONCLUSION AND RECOMMENDATIONS

Metal concentrations measured in waters downstream from the former Rum Jungle mine only indicated minor increases in Al and Mn concentrations in Zone 2 during the wet season between 2017 and 2019. This result is consistent with Zone 2 being the closest to the mine with potential for direct inputs. The concentrations of all other metals and metalloids investigated decreased or remained unchanged in wet season samples since 2015 in Zones 1 (upstream) and 3 (further downstream from Zone 2).

This report recommended LDWQOs for Se and derived new LDWQOs for U for all surface water monitoring zones upstream and downstream from Rum Jungle. These objectives were based on the most conservative international guidelines or locally relevant toxicity studies, respectively. It is important to note that the U WQOs remain interim due to the limited DOC dataset available to date and required to derive the LDWQOs. It is recommended that these objectives be reviewed once sufficient seasonal DOC data become available. Based on the most recent guidelines from the Queensland Government (DES, 2018b, 2018a), it is recommended that a minimum of 18 data values be collected over 2 years to derive guidelines.



# 6.

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