Part 1: Ranger – current operations

Conceptual models of contaminant pathways for operational phase of Ranger uranium mine

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Background

Conceptual models of contaminant pathways associated with uranium mining in the Alligator Rivers Region (ARR) have been developed as part of the evolving ecological risk assessment framework being developed by the Supervising Scientist since the early 1980s (eg Supervising Scientist 1982, van Dam et al 2004). In response to recommendations by the World Heritage Commission Independent Scientific Panel and the Alligator Rivers Region Technical Committee (ARRTC), a specific project was initiated in the early 2000s to produce an up-to-date comprehensive conceptual model of contaminant pathways associated with the operational phase of the Ranger uranium mine (RUM).

The conceptual model framework was updated using an internal scientific expert panel approach involving senior ***eriss*** scientific staff to identify the main chemical, physico-chemical, biological and radiological contaminant types that could be potentially transported from the Ranger mine lease into the surrounding environment. For each contaminant class the source/s, potential transport mechanisms off-site, affected environmental compartments, receptor organisms, routes of exposure, types of effect (where known) and measures of effect (where available) were detailed. The conceptual model identified six main types of stressors and nine transport mechanisms associated with the operational phase of mining at Ranger (van Dam et al 2004; Table 1).

Table 1 Potential stressors and transport mechanisms associated with ranger uranium mine operational phase (from van Dam et al 2004)

Potential stressors	**Inorganic toxicants** (eg uranium; magnesium; sulfate; manganese; ammonia)
	Organic toxicants (eg chlorinated aliphatic hydrocarbons, monocyclic aromatic hydrocarbons, polycyclic aromatic hydrocarbons, total petroleum hydrocarbons, organic sulfur compounds, volatile organic compounds)
	Radionuclides (eg Uranium – 238, 234, 235; Thorium-230; Radium-226; Lead-210; Polonium-210)
	Radon-222 and its progeny (eg Polonium-218, Lead-214, Bismuth-214, Polonium-214)
	Weed propagules (terrestrial and aquatic)
	Suspended sediments (<63 µm diameter)
Transport mechanisms[1]	Release from minesite waterbodies direct to Magela and Gulungul Creeks
	Seepage from minesite waterbodies to groundwater and possible discharge to surface water systems
	Land application of mine water followed by (i) infiltration to groundwater and discharge to surface water and/or (ii) direct runoff to surface water
	Stormwater runoff from non-mine areas of lease
	Airborne dust and other particulates from minesite
	Airborne emissions from mill stacks and vehicles from mine lease
	Exhalation from mine lease
	Bioaccumulation and trophic transfer to mobile species visiting minesite waterbodies
	Human and non-human vectors (including vehicles)

1 Not all transport mechanisms are relevant to all stressors

A diagram of the conceptual model elements was completed and validated by workshopping with external technical stakeholders in 2006 (van Dam & Bayliss 2006). A sub-model diagram for the transport of inorganic toxicants via the surface water to surface water pathway was also completed to demonstrate the methodology that was being used. However, sub-model diagrams and narratives for the other potential contaminant pathways (up to 30) identified in the conceptual model were not developed at this time. Finalisation of the remaining contaminant pathway sub-models was identified as a priority by ARRTC during its most recent revision of the Key Knowledge Needs (KKN). Consequently, resources were allocated to progress the project in the second half of 2009.

Progress

The incomplete conceptual model framework from van Dam & Bayliss (2006) was progressed using existing data/reports and the combined expert knowledge of senior ***eriss*** and ERA/EWLS scientific/technical staff. A comprehensive review of the status of scientific knowledge regarding the various contaminants and pathways was undertaken and the content and structure of the conceptual model elements were revised as required. Draft sub-model diagrams for each of the potential contaminant pathways showing linkages between various model pathway elements (source, transport mechanisms, environmental compartments) and relevant measurement and assessment endpoints were also developed. The draft sub-models were revised following a technical workshop involving ***eriss*** Program Leaders and other senior scientific staff in September 2009. A report on progress was provided to ARRTC in November 2009. An example of the structure and content of the revised sub-models can be seen in the sub-model for the transport of inorganic toxicants via the surface water to surface water contaminant pathway (Figure 1).

Supporting narratives have been drafted for each of the sub-models. They provide explanatory information on the various pathway components, including spatial or temporal characteristics, the level of scientific knowledge and scientific certainty and any knowledge gaps. The narratives were refined with input from senior ***eriss*** scientific staff in early 2010.

The overall project approach and draft outputs were considered and endorsed by ARRTC in April 2010. Following this, it was decided that the importance of the contaminant pathways should be assessed in terms of their inherent potential to adversely impact on the environment within the ARR. In this context it should be noted that inherent potential does not equate to actual potential in the event of various management strategies (eg impounding of runoff followed by water treatment) being in place to provide mitigation.

A technical workshop involving senior ***eriss*** scientific staff was held in June 2010 in which each of the contaminant pathways were assessed based on the nature and size or generating capacity of the contaminant source, and the volume (and rate) of contaminants able to be transported off the mine lease via the pathway transport mechanisms. Project outcomes will be made available in a SSD Internal Report, in the 2009–10 Annual ***eriss*** Research Summary and, eventually, a Supervising Scientist Report, in 2011.

The content, design and functionality of various communication products arising from the project will be determined based on consultation with ARRTC members, traditional owners and other relevant stakeholders. This project will also contribute towards the future development of a risk-based framework, identified as a knowledge need by ARRTC, to support ***eriss*** research activities and scientific knowledge management.

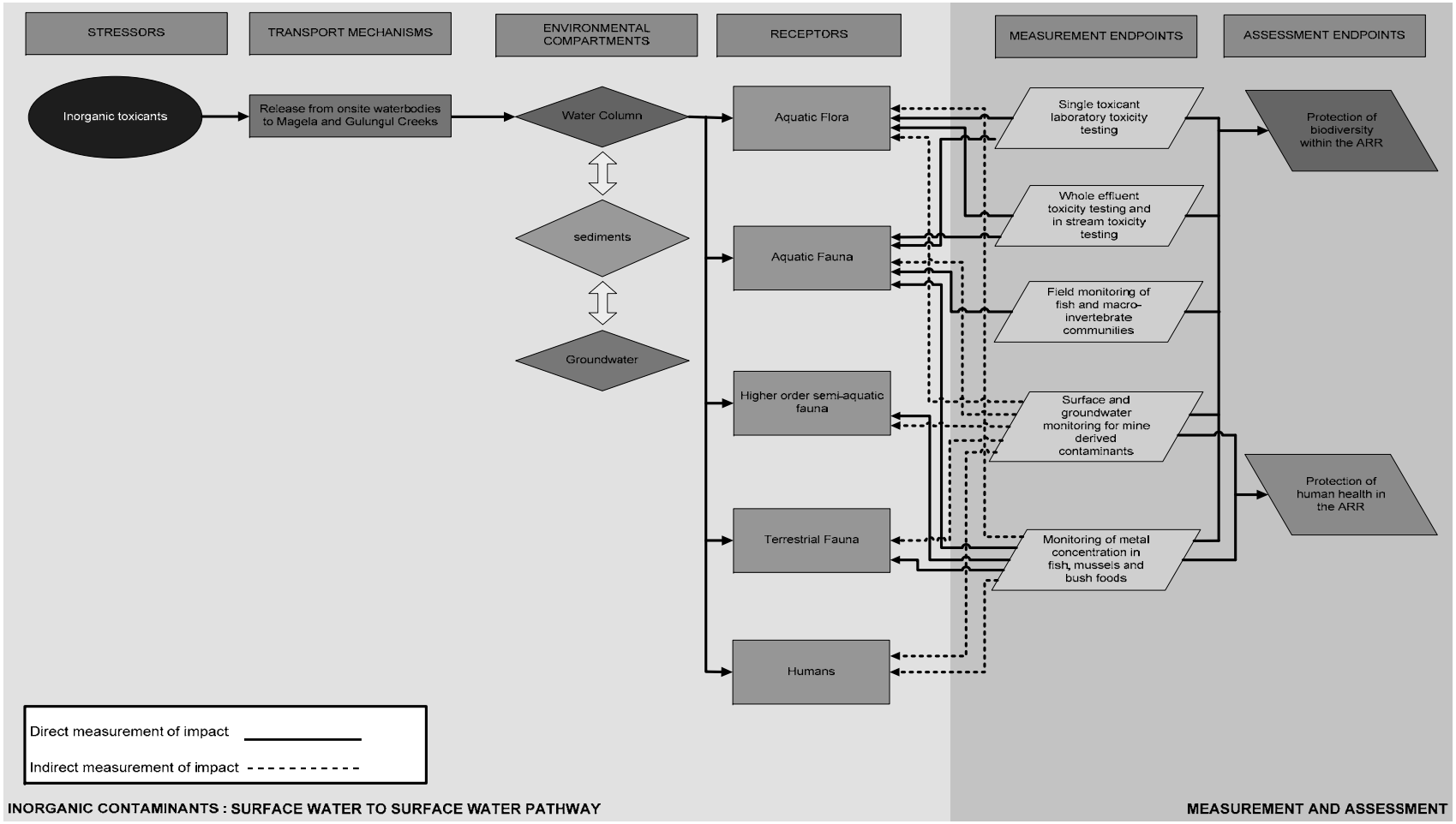

Figure 1 Conceptual model diagram for transport of inorganic toxicants from Ranger uranium mine via surface water to surface water pathway

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Radiological characterisation of Ranger mine land application areas

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Introduction

Water management is a major issue at Ranger uranium mine, given its location in the wet-dry tropics where up to 2 m of rain can fall within a single wet season. Release of water from the site into the downstream environment is minimised by the use of retention ponds (RP1-RP2). RP1 is defined as being part of the sediment control system on the minesite. The water is of relatively good quality, and freely discharges into Magela Creek during most wet seasons. Since 1985, water stored in RP2 during the wet season has been disposed of on site using land application methods. RP2 receives runoff from the low grade ore and waste stockpiles and other areas on the minesite.

The history of development of the land application areas (LAAs) on the Ranger site is summarised in Table 1. The Magela Land Application Area (LAA) was the first to be established using the spray irrigation method. Additional LAAs were developed as the amount of water to be disposed of rose through time as a result of the increasing area occupied by waste and low grade ore stockpiles. Starting in 1995, the RP1 and Djalkmara wetland filters were used to polish RP2 water before it was applied to the RP1 and Djalkmara East and West LAAs. In this context, and in contrast to the other LAAs, it should be noted that the Magela LAA has received untreated RP2 water throughout its entire operational life. It is therefore likely to contain the highest concentrations of metals and radionuclides.

From 2006 onwards increasing volumes of pond water have been treated by MF/RO water treatment during the wet season, with the clean permeate being discharged along the Corridor Creek catchment line. The introduction of active pond water treatment during the wet season has progressively reduced the volume needed to be disposed of by land application during the dry season.

Table 1 Sources of water for land application areas at ERA's Ranger uranium mine

Land Application Area	Source of applied water	Total area (ha)	Year commissioned
Magela A (MALAA)	RP2 water	33	1985
Magela B (MBLAA)	RP2 water	20	1994
RP1	polished RP2 water	46	1995
Djalkmara East (E. Dj)	(un)polished RP2 water	18	1997
Djalkmara West (W. Dj)	(un)polished RP2 water	20	1999
Jabiru East (JELAA)	(un)polished RP2 water	52	2006
RP1 Extension (RP1 ext)	RP2 water	8	2006
Corridor Creek (CCLAA)	RP2 water	141	2007

[1] SafeRadiation, Brisbane (www.saferadiation.com)

[2] Environmental Strategy Department, Energy Resources of Australia, Darwin

The use of land application as a water treatment method relies on the fact that radionuclides and most metals have a tendency to bind to the organic rich surface horizons of soil profiles (Davis 1983, Akber & Marten 1992, Willett et al 1993, Hollingsworth et al 2005). These bound metals and radionuclides have a low leachability and will therefore be unlikely to impact the aquatic environment downstream of Ranger. However, there has been ongoing stakeholder concern about the radiological status of the Ranger LAAs, in particular the Magela LAAs and their capacity to continue to adsorb radionuclides at the current rate of application. The concentration of radionuclides adsorbed in the soil could potentially require the area to be rehabilitated at closure, based on the 1 mSv public dose limit recommended by the ICRP (2007).

The Environmental Strategy Department within ERA, in collaboration with *SafeRadiation*, Brisbane, and the Environmental Research Institute of the Supervising Scientist (***eriss***), have initiated a project to identify and quantify current radiological issues associated with the LAAs, resulting from their use to dispose pond water. The aims of this project are to characterise the magnitude and extent of radiological contamination at each of the Ranger LAAs and to suggest options for their rehabilitation. The nature of these options will strongly depend on the estimated post rehabilitation radiation doses to people provided by data produced by this current project.

Methods

Soil samples were collected at various distances (0–15 m) from the sprinkler heads at all LAAs and also included samples not influenced by irrigation. Soil samples were taken to a depth of 10 cm. In addition, ten soil cores were collected and sampled at a resolution of 5 cm down to 20 cm depth. Whole soil samples were dried and crushed, and prepared for radionuclide analysis via gamma spectrometry at ***eriss***. The methods for gamma spectrometry are described in Murray et al (1987), Marten (1992) and Esparon and Pfitzner (2010). Leaf litter samples were also taken at various distances from the sprinklers. This material was ashed and homogenised and analysed by gamma spectrometry. The radionuclide activity concentration results were used to determine vertical and horizontal depositional patterns and to calculate the total load of radionuclides retained in LAA soils. These loads (in $kBq \cdot m^{-2}$) were then compared with loads calculated from the known volumes and water quality data provided by ERA for the water applied at the various LAAs over the years.

Radon (^{222}Rn) exhalation was determined at various distances from the sprinkler heads using conventional charcoal cups (Spehr & Johnston 1983). Surveys were conducted in the dry season 2008 and in March 2009 (wet season). There was no irrigation of mine waters during and immediately prior to charcoal cup exposure. Charcoal cups were then analysed using the ***eriss*** NaI gamma detector. Results from the radon survey have been reported in a previous Research Summary (Bollhöfer et al 2010).

Results

Soil and leaf litter radionuclide activity concentration

The maximum ^{238}U soil activity concentration measured was 28 000 $Bq \cdot kg^{-1}$ (2270 $mg \cdot kg^{-1}$ uranium) and the average was ~1700 $Bq \cdot kg^{-1}$ (137 $mg \cdot kg^{-1}$).

In contrast, the maximum measured ^{226}Ra soil activity concentration was only a little above 1000 $Bq \cdot kg^{-1}$, with an average of approximately 190 $Bq \cdot kg^{-1}$. A large number of ^{226}Ra activity concentration values are in the range 100–500 $Bq \cdot kg^{-1}$. Most samples exhibit an activity concentration trend of $^{238}U >> {}^{226}Ra > {}^{210}Pb$, which reflects the signature of RP2 water applied

to the soils. This is important for the external gamma pathway, as uranium is only a weak gamma emitter, and the majority of the terrestrial gamma dose rate measured in air originates from ^{226}Ra decay products (^{214}Bi & ^{214}Pb) rather than uranium (Saito & Jacob 1995).

Although the activity concentration in surface leaf litter (Bq·kg^{-1} dry weight) is ~10 times higher than that measured in the underlying soil, only a small fraction of the total load of applied radionuclides appears in the leaf litter. It was found that approximately 90% of the applied radionuclides have been retained in the top 10 cm of the soils. This is in agreement with earlier studies conducted in the Magela LAAs (Akber & Marten 1992).

To put the radiation source term of the Magela LAA into context it should be noted that the concentration of uranium in waste rock has been determined and is typically around 100 mg·kg^{-1} (Lawrence 2006), which translates to 1200 Bq·kg^{-1} of ^{226}Ra in radioactive equilibrium with ^{238}U. The combined exposure to the external gamma radiation and radon progeny inhalation pathways is a function of both the magnitude of ^{226}Ra activity concentration in the soil and its depth of occurrence. The typical diffusion path length for radon in soil is 1–2 m. Thus the 10 cm effective depth of elevated ^{226}Ra (average value of 190 Bq·kg^{-1}.) in the soil of the LAA needs to be compared with the potentially many metres of depth of waste rock containing about 1200 Bq·kg^{-1}. Consequently, annual doses via those two pathways will be less significant over the footprints of the LAAs, assuming that no specific remedial works are undertaken of these areas, compared with the areas that will contain substantial depths of waste rock after remediation of the site.

^{238}U and ^{226}Ra soil activity concentrations in the top 10 cm decrease with distance from the sprinkler heads. This decrease can be approximated mathematically using an exponential equation, and this approach has been used to estimate radionuclide activity loads deposited within the sprinkler wetting zone. The results derived from the direct measurement of soil activities, and subsequent integration over the LAA areas, compare well with the applied loads calculated from historical radionuclide inventories in RP2 water and irrigation rates provided by ERA.

Figure 1 shows the average applied ^{226}Ra and ^{238}U aerial activity densities, respectively, for the various LAAs, calculated from the volumes and the respective radionuclide activity concentrations of applied water. In Figure 2 calculated average aerial activity densities are compared with the modelled results (based on an exponential decrease of aerial activity density with distance from the sprinkler heads, which has been fitted to the radionuclide activity densities measured on ground) for two cases, a relatively old land application area (Magela B) and a relatively recent one (Jabiru East). In all cases, experimentally derived and calculated aerial activity densities agree within one order of magnitude or better.

The activity ratio of ^{226}Ra/^{210}Pb has been used to distinguish areas affected by application of mine waters from areas that may have naturally higher soil radionuclide activity concentrations. Some parts of the LAAs are located within areas that exhibited higher natural backgrounds before mining started (Bollhöfer et al 2010). This is in particular obvious in samples from the Djalkmara East LAA, to the northwest of pit 3, and also for some samples from the Corridor Creek LAA. This finding is important in the context of post irrigation dose assessment, as a proportion of the determined radiation doses will be due to existing natural radiation anomalies at these areas.

Figure 1 ^{238}U (yellow – white in printed copy) and ^{226}Ra (green – grey in printed copy) aerial activity densities (kBq·m^{-2}) calculated from historical radionuclide inventories in RP2 water and irrigation rates provided by ERA (from Akber et al 2010)

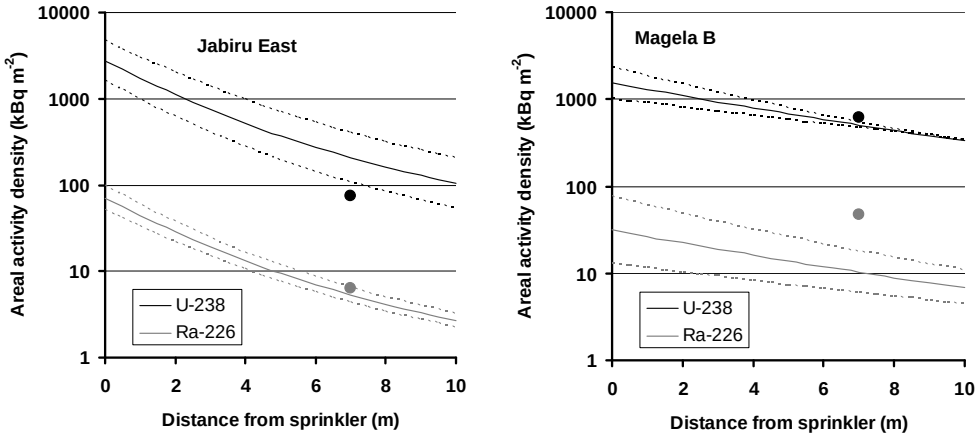

Figure 2 Modelled aerial activity densities for ^{238}U (black) and ^{226}Ra (grey) and comparison with the applied loads (dots) calculated from historical radionuclide inventories in RP2 water and irrigation rates provided by ERA

Preliminary dose estimate

Moroney (1992) has developed a model for the LAAs that allows to calculate a committed effective dose rate equivalent (Sv·yr^{-1}) for all exposure pathways from the aerial activity densities (kBq·m^2) of applied ^{238}U and ^{226}Ra, respectively. This model has been used here,

taking into account changes in applicable dose conversion factors, for a preliminary dose estimate, or a trend investigation, of radiation doses from the various pathways. The aerial activity densities shown above, and results from our investigation, have been used for the pathway analysis. A refined model is currently under development.

As an example the average activity concentrations and aerial activity densities from the Magela A and B LAAs have been used to estimate doses via the various pathways and results are shown in Table 2. In calculating the doses received via the various pathways, results have been adjusted to reflect changes in inhalation and ingestion dose conversion factors (ICRP 1996). In addition, an equivalent soil concentration of $(8.9 \pm 4.3)\cdot10^{-7}$ Bq·m^{-3} in air per Bq·kg^{-1} in soil has been used and, to be conservative, dust class S (lung adsorption type: slow) was assumed for ^{226}Ra and uranium, respectively (ICRP 1996). Average fruit/soil concentration factors ($Bq/kg_{wet}/Bq/kg_{dry}$) were applied for ^{226}Ra (0.0179) and uranium (0.0016), determined from data in Ryan et al (2005). These concentration factors are almost one order of magnitude lower than those used by Moroney (1992) in his model. A conversion factor for radon has been determined in our study and is similar to the one used by Moroney (1992). Furthermore, it has been assumed that the LAAs are accessed for four hours every day during daytime for food gathering purposes, and that 5 kg of fruit are gathered and ingested by an individual per year.

Table 2 Preliminary pathway analysis for the Magela A and B land application areas, assuming 4 hours occupancy per day (daytime) and 5 kg of fruit are ingested per year

Pathway	^{226}Ra			^{U}nat		
	(Sv·yr^{-1}) per (kBq·m^{-2})	kBq/m^2	mSv·yr^{-1}	(Sv·yr^{-1}) per (kBq·m^{-2})	kBq/m^2	mSv·yr^{-1}
External γ MALAA	$2.5\cdot10^{-6}$	64	0.16	$4.0\cdot10^{-8}$	970	0.04
MBLAA		47	0.12		610	0.03
Ingestion MALAA	$1.7\cdot10^{-7}$	64	0.01	$5.1\cdot10^{-9}$	970	0.005
MBLAA		47	0.01		610	0.003
RDP MALAA	$2.6\cdot10^{-7}$	64	0.02		970	
MBLAA		47	0.01		610	
Dust MALAA	$2.9\cdot10^{-7}$	64	0.02	$5.5\cdot10^{-7}$	970	0.53
MBLAA		47	0.01		610	0.33
Total MALAA			0.21			0.57
MBLAA			0.15			0.37

Total estimated dose received at the Magela B and A LAAs from applied ^{226}Ra and uranium amounts to 0.5–0.8 mSv per year, assuming 4 hours occupancy per day. It is important to note that further refinements of the model are currently implemented. In particular, the investigation has shown that the dust inhalation pathway in the LAAs may become increasingly important and efforts currently focus on determining the resuspension factors for the area to more reliably quantify this pathway.

Conclusions and future work

This investigation has shown an increase of radionuclide activity concentration in soils at the Magela, Djalkmara and RP1 LAAs due to irrigation of RP2 water. However, this accumulation of radionuclides is restricted to the top 10 cm of the soil profile where most of the applied load

is captured. There is good agreement between measured radionuclide loads in the LAAs, and loads inferred from water quality data and irrigation rates over the past 25 years.

A preliminary dose assessment has shown that annual doses when accessing the Magela LAAs for four hours every day are ~0.6 mSv per year due to the radionuclides in the applied water. The dust pathway is associated with the highest uncertainty, and further research is required to quantify this pathway. A refined model is currently being developed.

There are several rehabilitation options that could be used to reduce exposure of people potentially accessing the footprint of the LAAs, in the event that it was determined that such a reduction was needed. These options include removal of the surface 10 cm of contaminated soil and placing it into the pit, tilling of the soil, or a mixture of both. A rehabilitation trial has been started at the Magela B LAA in order to investigate whether the predicted reductions in dose rates can be achieved.

The extent of above background doses that will be permitted by current standards at Ranger post-remediation depends on both the pre-mining radiological conditions and the nature of future use of the area by indigenous people and the general public. The current status of a project to determine pre-mining radiological conditions using Ranger Anomaly 2 as an analogue is described in another paper in this volume (KKN 2.2.5 Pre-mining radiological conditions at Ranger mine). An agreed position by stakeholders on future land use activities and likely occupancy of the area is required as a pre-requisite to being able to predict applicable doses to humans post-remediation, and to inform the possible need to carry out specific rehabilitation of the LAAs. The land use and occupancy factors will be used to further refine the radiation dose model to be produced for the land application areas.

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Influence of dissolved organic carbon on the toxicity of aluminium to tropical freshwater biota

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Background

This work is part of a PhD project studying the influence of dissolved organic carbon (DOC) on metal toxicity to freshwater organisms. The first part of the project assessed the effects of DOC on uranium toxicity and the results were presented in the 2008–09 Annual Report.

Aluminium (Al) is a metal of general ecotoxicological concern for the mining industry. Inputs of Al to surface waters can occur through acidic seepage or discharge of acidic mine waters from legacy, closed and operating minesites. Examples of such sites in the Northern Territory include the legacy Rum Jungle and Rockhole Creek uranium mines and metal minesites throughout the Pine Creek Geosyncline metal province. The outcomes of the assessment done by the Supervising Scientist Division for the Rockhole Mine Creek site located in the Alligator Rivers region have been previously documented (Supervising Scientist 2009).

The classic acid drainage conditions occurring at these sites provide an environment in which the bioavailability and toxicity of Al to biota are potentially much increased. In the case of fish, Al binds to the gills where it leads to respiratory dysfunction (Rosseland et al 1990). Al has also been found to bioaccumulate in filter feeding invertebrates, in particular those feeding on benthic detritus (Gensemer & Playle 1999). There are few toxicity data for Al in freshwater, particularly at acidic pH. The only water quality guideline available for Al in freshwater at low pH is a *low reliability* trigger value of 0.8 µg/L Al (ANZECC & ARMCANZ 2000). This guideline also does not incorporate the influence of DOC, which can form strong complexes with Al and potentially influence its bioavailability and toxicity (Tipping 2002).

The objective of this study was to quantify the influence of DOC on the toxicity of Al to three tropical freshwater species at low pH (5.0) and alkalinity (2–14 mg/L as $CaCO_3$). The selected tropical species, green hydra (*Hydra viridissima*), green alga (*Chlorella* sp), and the cladoceran (*Moinodaphnia macleayi*) were chosen to cover a range of trophic levels.

Methods

The influence of DOC was assessed using two sources of DOC: (i) the international standard Suwannee River fulvic acid (SRFA) and (ii) a local DOC present in water sourced from Sandy Billabong located adjacent to Magela Creek upstream of Ranger mine in Kakadu National Park. Four concentrations (1, 2, 5 and 10 mg/L) of SRFA and local DOC were used and test species were exposed to up to 5 mg/L total Al. For the SRFA, toxicity testing was conducted using diluted (25% dilution with Milli-Q water) Magela Creek water (DMCW), containing a natural DOC concentration of <1 mg/L, as the test medium. DMCW, rather than

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synthetic Magela Creek water (SMCW), was used as the diluent because its low concentrations of background DOC (~1 mg/L) and alkalinity were required to provide buffering capacity to maintain the low test pH of 5.0 (SMCW, which lacks DOC, was not able to hold pH at a pH lower than pH 6).

For the local DOC, Sandy Billabong water (SBW), naturally containing 10 mg/L DOC, was diluted to the required DOC concentrations (1, 2, 5, 10 mg/L) using SMCW containing a similar inorganic composition to SBW but lacking in DOC. For the *Chlorella* test, nitrate and phosphate were added as nutrients (3.28 mg/L nitrogen and 0.046 mg/L phosphorus).

Test systems were static, with 24 h renewal of test solutions for *H. viridissima* only (there was no renewal for the *Chlorella* sp or *M. macleayi* tests). Test temperatures were maintained at 27 ± 1ºC for *M. macleayi* and *H. viridissima* and 28 ± 1ºC for *Chlorella* sp. For each species, four tests were conducted for SRFA and three tests for the SBW DOC, in order to fully characterise the concentration-response relationships.

Test durations and endpoints were as follows: *H. viridissima* – 96-h population growth rate; *Chlorella* sp – 72-h growth rate; *M. macleayi* – 24-h neonate survival. For all tests, general water parameters (pH, DO and EC) were monitored daily. At the beginning of each test, water samples were taken for analyses of DOC, alkalinity, hardness and a standard suite of metals and major ions. For each species, response data from the tests were pooled, and concentration-response relationships were determined using non-linear regression analyses.

Progress

Concentration-response relationships and associated linear regressions of toxicity (expressed as IC_{50} – the concentration that results in a 50% inhibition of the test response relative to the control response) against fulvic acid concentration are shown in Figures 1 and 2, respectively, while the toxicity summary data are shown in Table 1. Al toxicity was reduced in the presence of both DOC sources. For *H. viridissima*, SRFA was ~5 times more effective (based on the increased slope of the IC_{50} versus DOC plot) at reducing Al toxicity than the local SBW DOC. For *Chlorella* sp, SRFA was only ~2 times more effective at reducing Al toxicity than the local DOC. For *M. macleayi*, Al toxicity was reduced by a similar factor in the presence of both DOC sources.

Physicochemical variables were input into the WHAM (Windermere Humic Aqueous Model, CEH 2002) chemical speciation computer model to estimate the effect of DOC on Al speciation, which was related back to Al toxicity. For both DOC sources, the decrease in Al toxicity with increasing DOC can be attributed to a reduction in the free (Al^{3+}) and monomeric hydroxy ($Al(OH)_2^+$) ion concentrations (the two most toxic species), due to Al being bound by DOC. These results and those of additional speciation modelling used to investigate finer aspects of the observed responses to Al will be presented in more detail in subsequent publications.

Extending the number of species tested to 5 or 6 would enable a high reliability trigger value to be derived for Magela Creek (and similar composition) waters. However, to do this would be technically very challenging. For a species to be suitable for this testing it would need to be able to tolerate water at pH 5 and exhibit effects within the solubility limits of Al (which for water at pH 5 is around 400–500 µg/L).

Based on the responses of the three test species to Al in the presence of 1 mg/L DOC (IC_{50}s ranging from 50–950 µg/L Al), it appears that the current *low reliability* trigger value of 0.8 µg/L Al, which does not account for the influence of DOC, is likely to be overly protective for natural waters containing this level, or greater, of DOC.

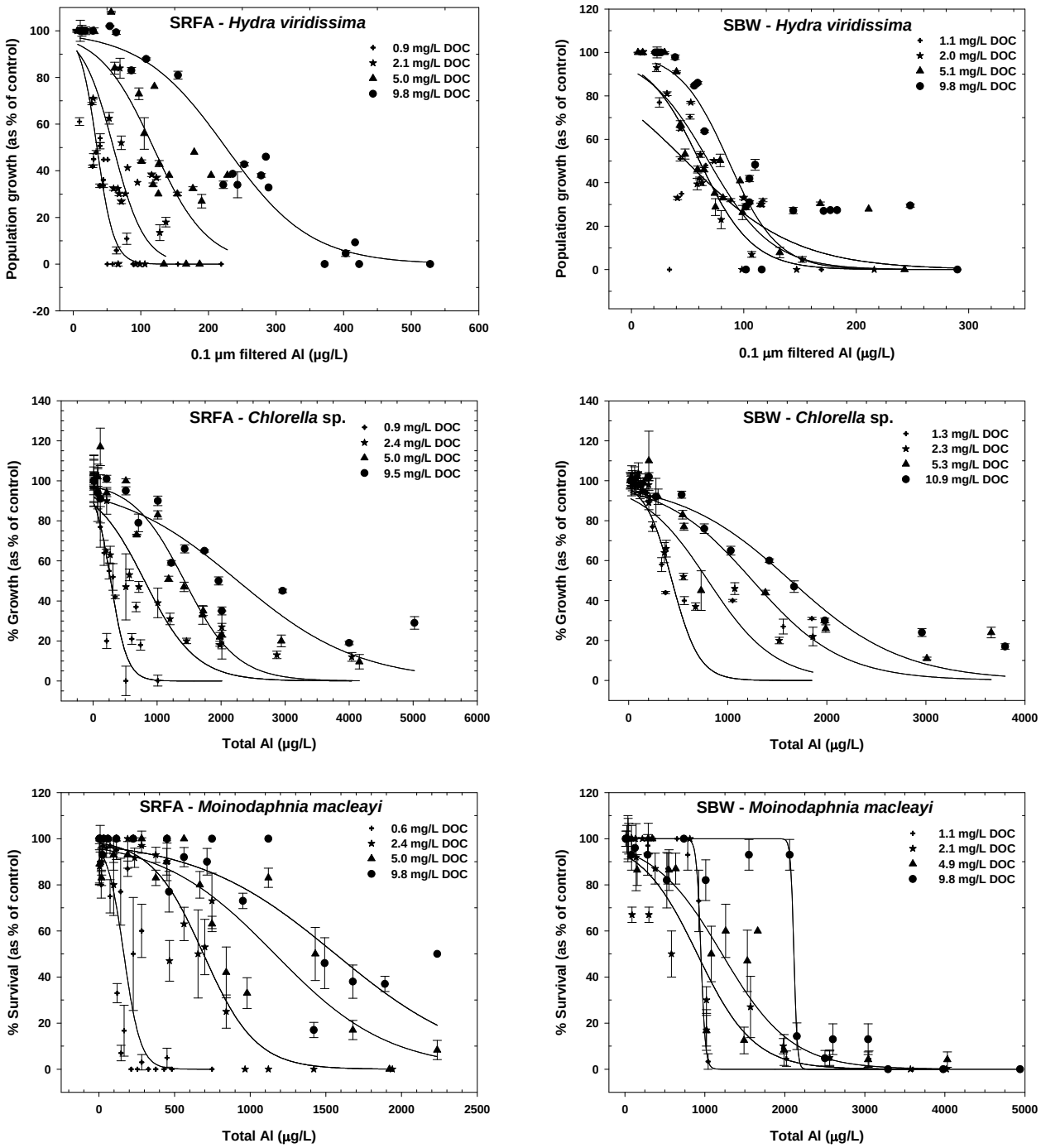

Figure 1 Concentration-response plots for Al exposures. Left: using Suwannee River fulvic acid (SRFA) in dilute Magela Creek water, 4 pooled tests for each species. Right: Sandy Billabong Water (SBW) diluted in synthetic Magela Creek water, 3 pooled tests for each species. Data points represent the mean of 3 replicates ± SE for *Chlorella* sp and *M. macleayi*, and 2 replicates ± SE for *H. viridissima*.

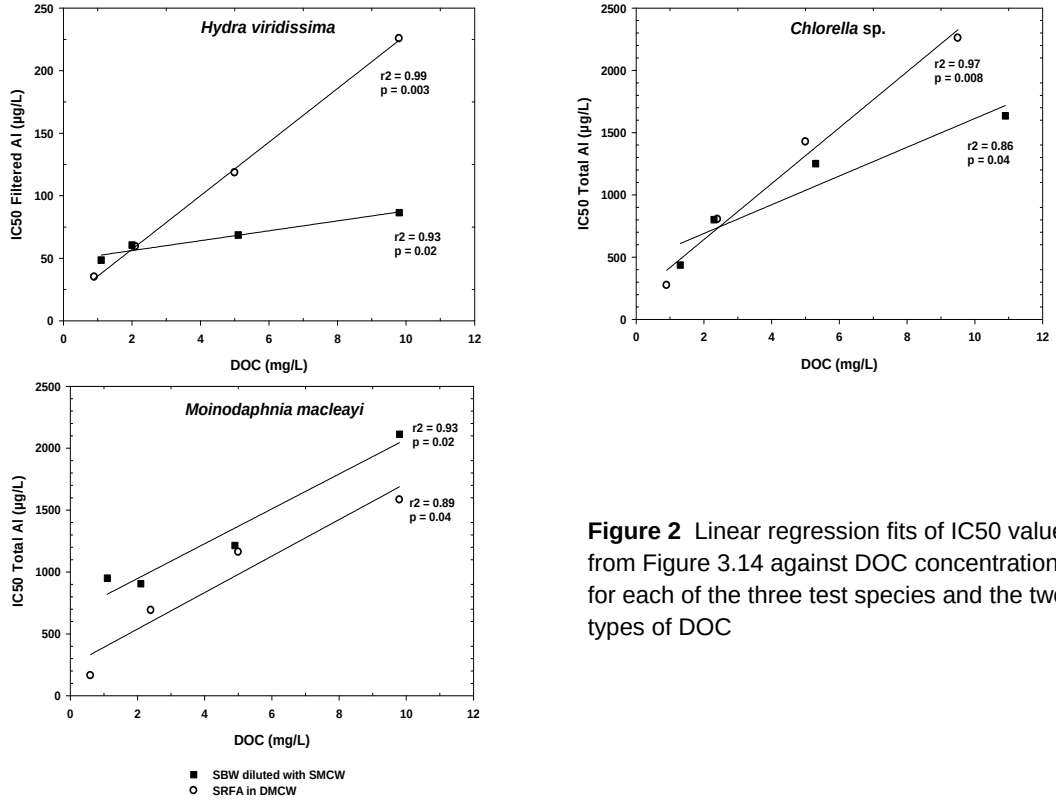

Figure 2 Linear regression fits of IC50 values from Figure 3.14 against DOC concentrations for each of the three test species and the two types of DOC

Table 1 Effect of two different forms of dissolved organic carbon (DOC): (i) Suwannee River fulvic acid standard I, and (ii) DOC in sandy billabong water, on the toxicity of aluminium to three local freshwater species

Species	DOC[a] (mg/L)	IC_{50}[b] (95% CL)[c]		Extent of amelioration of Al toxicity (µg Al mg/L DOC^{-1})[d]	
		DMCW+SRFA[e]	SBW diluted with SMCW[f]	DMCW +SRFA	SBW
Hydra viridissima	1	35 (29–39)	49 (NC–149)		
(green hydra)	2	59 (40–71)	61 (48–72)	21	4.0
	5	119 (91–138)	69 (54–81)		
	10	226 (204–242)	87 (65–101)		
Chlorella sp	1	275 (189–384)	437 (315–679)		
(unicellular alga)	2	805 (560–1032)	801 (560–1134)	225	115
	5	1427 (1242–1582)	1251 (870–1724)		
	10	2260 (1830–2867)	1635 (1410–1895)		
Moinodaphnia macleayi	1	164 (123–206)	950 (939–983)		
(cladoceran)[g]	2	691 (610–767)	905 (608–1293)	147	141
	5	1162 (972–1390)	1214 (868–1510)		
	10	1584 (1277–1930)	2113 (2083–2140)		

a DOC: dissolved organic carbon, b IC_{50}: the concentration that results in a 50% inhibition of the test response relative to the control response; c 95% confidence limits; d extent of amelioration is the slope of the regression between IC50 and the concentration of DOC (Figure 3.15). e SRFA made up in dilute Magela Creek water (25%) ; f SBW diluted with SMCWr; g For *M. macleayi*, toxicity estimates relate to concentrations that affect percentage survival (as a % of control survival), compared to sub-lethal endpoints, such as growth and reproduction, for the other species.

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Development of a reference toxicity testing program for routine toxicity test species

K Cheng, R van Dam, A Hogan, A Harford, C Costello & M Trenfield

Background

Over the past six years, in response to recommendations by van Dam (2004) and Dr Jenny Stauber at ARRTC's 14th meeting (September 2004), the ***eriss*** ecotoxicology laboratory has been progressively implementing a program of reference toxicant testing, using uranium, for its routine suite of test species. The methods were developed in accordance with formal guidance on reference toxicant testing (Environment Canada 1990). Since 2004–05, reference toxicant control charts have been developed for four of the five routine testing species. The aims for 2009–10 were to:

1 continue with the established reference toxicity testing programs for *Moinodaphnia macleayi*, *Chlorella* sp, *Hydra viridissima* and *Mogurnda mogurnda*;

2 continue to investigate identified difficulties with the *Lemna aequinoctialis* (duckweed) reference toxicity test with the objective of establishing an acceptable and consistent control growth and a consistent concentration-response relationship.

Methods

Descriptions of the testing procedures are provided in 'Ecotoxicological testing protocols for Australian tropical freshwater ecosystems' Supervising Scientist Report 173 (Riethmuller et al 2003).

Progress

In total, 19 reference toxicants tests (*Chlorella* – 4; *Hydra* – 4; *Moinodaphnia* – 4; *Mogurnda* – 3 and *Lemna aequinoctialis* – 4) were completed during 2009–10. Of these tests, 18 provided valid results, as summarised in Table 1. The associated control charts are presented in Figure 1.

A summary of the issues identified during 2009–10 for each component of the reference toxicity test program is provided below.

Chlorella sp

All four *Chlorella* sp tests were valid for the period with control growth rates within the acceptability criterion of 1.4 ± 0.3 doublings/day (Riethmuller et al 2003). Test 1064G had an IC50 of 10 µg L^{-1} U, which was slightly below the lower warning limit of 13 µg L^{-1} U at the time (current warning limit is 5 µg L^{-1} U). A repeat test (1080G) resulted in an IC50 of 15 µg L^{-1} U which was only just above the warning limit. Test conditions for both tests were acceptable (ie water quality, light intensity and temperature) and there were no visible problems with culture health at the time (ie colour/appearance of cells looked normal). Tests 1091G and 1106G reported IC50s of 30 and 32 µg L^{-1} U, respectively, which was closer to the running mean of 40 µg L^{-1} U.

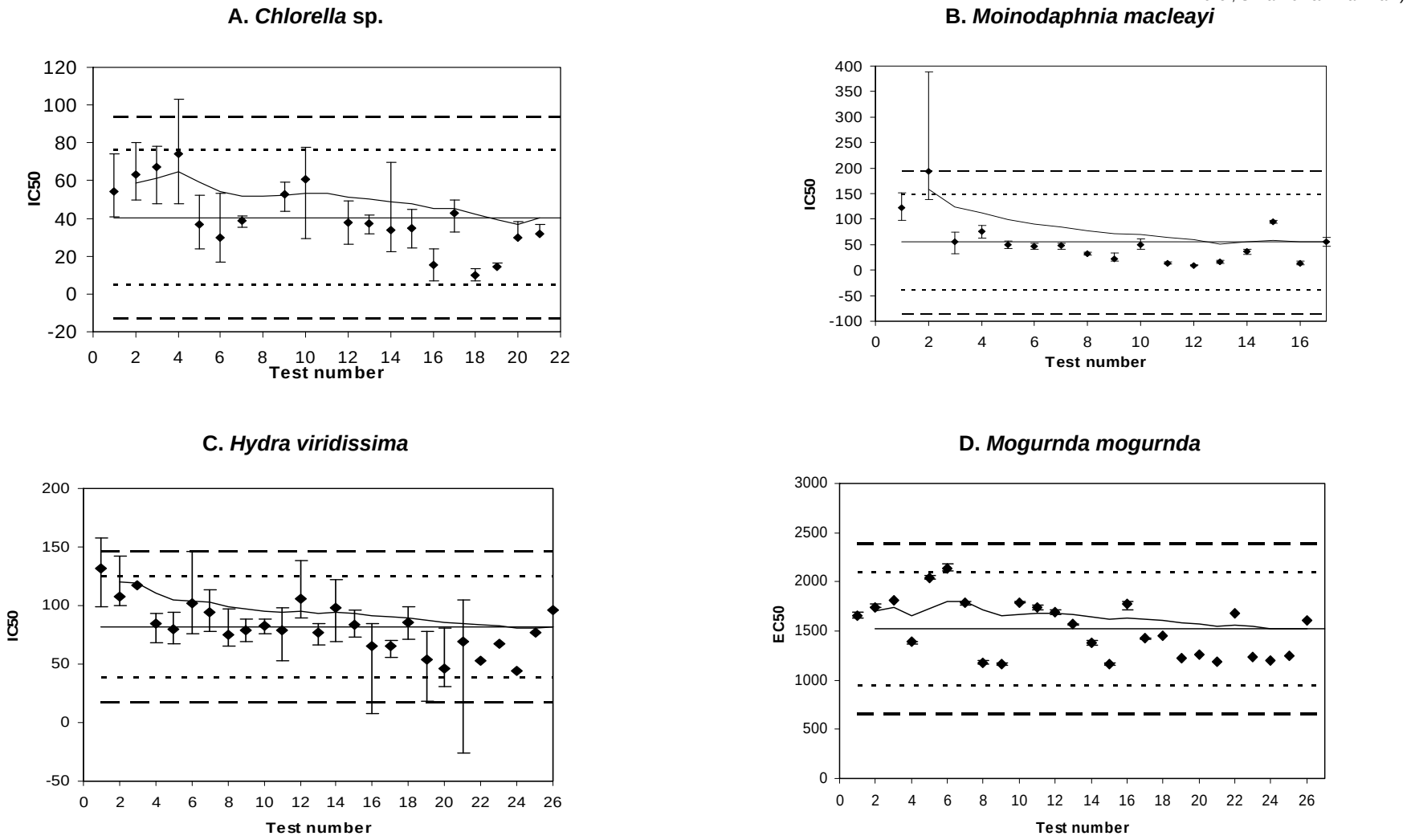

Figure 1 Reference toxicant control charts for A. *Chlorella* sp., B. *M. macleayi*, C. *H. viridissima* and D. *M. mogurnda*, as of September 2010. Data points represent IC_{50} or EC_{50} toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – upper and lower 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – upper and lower warning limits (± 2 standard deviations); unbroken line – running mean.

Table 1 Summary of uranium reference toxicity test results for 2009–10

Species & endpoint	Test Code	IC_{50} (µg L^{-1})	Valid test?	Comments
Chlorella sp (72-h cell division rate)	1064G	10 (7, 13)	Yes	
	1080G	15 (11, 17)	Yes	
	1091G	30 (26, 38)	Yes	
	1106G	32 (26, 37)	Yes	
Hydra viridissima (96-h population growth)	1029B	68 (65, 71)	Yes	
	1077B	45 (32, 59)	Yes	
	1099B	77 (72, 81)[a]	Yes	
	1136B	98 (86, 109)	Yes	
Moinodaphnia macleayi (48-h immobilisation)	1023I	95 (93, 98)	Yes	
	1078I	NC[b]	No	No effect at highest concentration[a]
	1103I	14 (11, 17)	Yes	
	1129I	55 (47, 64)	Yes	
Mogurnda mogurnda (96-h sac fry survival)	1060E	1202 (1043, 1349)	Yes	
	1098E	1252 (1108, 1373)	Yes	
	1123E	1582 (1469, 1689)	Yes	
Lemna aquinoctialis (96-h population growth	1049L	10000 (7800 12000)	Yes	
	1065L	11650 (10300, 12300)	Yes	
	1089L	9480 (9080, 10530)	Yes	
	1093L	10370 (9900, 10900)	Yes	

Values in parentheses represent 95% confidence limits

[a] See text for discussion

[b] Not calculable

In the previous reporting period (2008–09), there were problems with achieving valid control growth, with only one of the four tests being acceptable. Low nutrient (NO_3^- and PO_4^{-3}) concentrations were suspected to have been the cause. Northern Territory Environmental Laboratories (NTEL) advised that stocks should be prepared at more regular intervals rather than retaining bulk stocks for prolonged periods. However, nutrient results provided from NTEL were variable and did not necessarily support this advice. For example, a freshly made nitrate stock sampled in September 2009 measured 2.83 mg L^{-1} N, which is close to the nominal concentration of 3.24 mg L^{-1} N. In March 2010, another freshly made batch measured 1.81 mg L^{-1} N. It is also worth noting that control growth rates in tests using the above two nitrate stocks were 0.91 and 1.64 doublings/day, respectively. This relative response is inconsistent with the reported concentrations of N in the two stocks.

In the four tests completed in 2009–10, results for N and P analyses ranged between 1.14 and 3.01 mg L^{-1} (nominal N: 3.24 mg L^{-1}) and 0.01–0.13 mg L^{-1} (nominal P: 0.045mg L^{-1}), respectively. All four of these tests had good control growth, regardless of the reported range for nutrient concentrations. This issue is still under investigation.

H. viridissima (green hydra)

All four reference toxicity tests for *H. viridissima* were valid. There are no issues associated with this protocol. The running mean IC50 is 83 µg L^{-1} U with all results within the upper and lower warning limits (± 2 standard deviations) of 126 and 40 µg L^{-1} U, respectively.

M. macleayi (water flea)

Of the four reference toxicity tests for *M. macleayi*, three were valid. The second test (1078I) produced an unusual result, whereby there was 100% survival of *M. macleayi* exposed to the highest concentration tested of ~140 µg L^{-1} U. In response to this, the third test (1103I) investigated U concentrations over a broader range (control, 4, 18, 75, 147 and 300 µg L^{-1}) to determine an effect concentration. Mortality was observed at 18 µg L^{-1} U with 100% mortality

to *M. macleayi* exposed to 75, 147 and 300 µg L^{-1} U, resulting in a LC_{50} of 14 µg L^{-1} U. Using the same concentration range as 1103I, the fourth test (1129I) resulted in significant mortality at 80, 160 and 320 µg L^{-1} U, and a LC_{50} of 55 µg L^{-1} U. The running mean LC_{50} (lower, upper warning limits) is 55 (-38.1, 149) µg L^{-1} U.

It was suspected that the fermented food provided to *M. macleayi* during testing (fermented food with vitamins, FFV) was contributing to the variable response observed across cladoceran tests. To further investigate this an additional test (1130I) was run concurrently with Test 1129I using 0.1 µm filtered FFV, instead of the usual unfiltered FFV This change in procedure was used to determine if the filtered or particulate fraction of FFV was more nutritionally important for *M. macleayi*, and whether U toxicity would be affected, noting that such factors might explain the variable toxicity results produced by this test. All other aspects of the test conditions were the same (ie diluent water, added algae, light, temperature etc).

While all the individuals exposed to 320 µg L^{-1} U in Test 1129I died, all of the fleas in Test 1130I (with the filtered FFV) survived (Figure 2). The results suggest that U bound to particulate organic matter may be a more significant source of U to *M. macleayi* than dissolved U, at least under the conditions used here. These tests will be repeated in the near future to determine the reproducibility of the results. In addition, the effects of filtered versus unfiltered FFV on *M. macleayi* reproductive output under control conditions, and on chronic U toxicity will also be assessed.

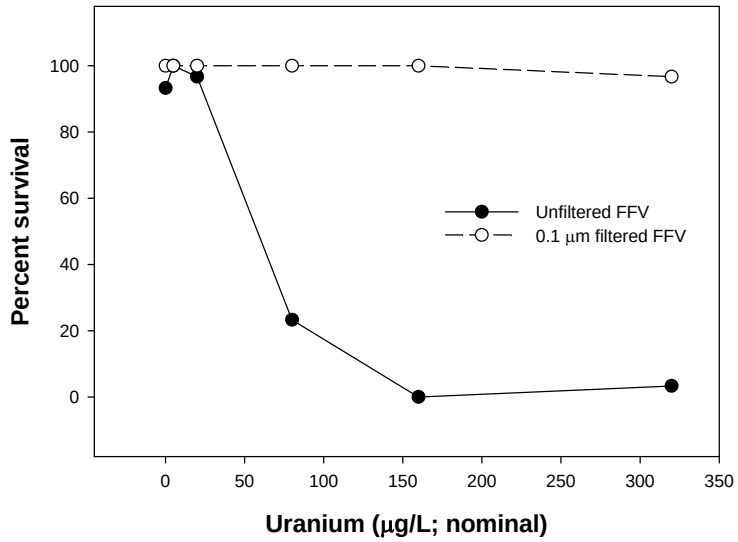

Figure 2 Effect of unfiltered (Test 1129I) and 0.1 µm filtered (Test 1130I) FFV (food source) on the toxicity of uranium to *Moinodaphnia macleayi*

M. mogurnda

All three reference toxicity tests for *M. mogurnda* were valid with all IC_{50} values within the warning limits. There are no problems associated with this protocol. The running mean IC50 is 1544 µg L^{-1} U and all results within the upper and lower warning limits of 2130 and 960 µg L^{-1} U, respectively.

Reference toxicity test development for *L. aequinoctialis*

The reference toxicant test method has been finalised. Previous growth trials using 2.5% CAAC plant growth medium (the medium used to culture this species; see Riethmuller et al 2003) have shown that it supported good growth and generally met the growth criteria. However, due to the very high concentrations of nutrients and essential elements in the CAAC medium, very high reference toxicant (U) concentrations were required to elicit a toxic response. The key challenge has been optimising the test medium so as to enable adequate control growth whilst still enabling a response to be observed at uranium concentrations that are not excessively high.

Five reference toxicant tests were conducted for *L. aequinoctialis* with the first test using 2.5% CAAC (control, 642, 1200, 2520, 5160, 1120, 19600 µg L^{-1} U). There was no effect at any of these concentrations. Subsequently, four tests were conducted using 1% CAAC, with control growth in all tests above the protocol's minimum acceptable growth rate of 0.35 (ie four-fold increase in frond numbers after 96-h). The average (range) IC50 for the four tests was 10 400 (9480 –11 650) µg L^{-1} U, and a preliminary reference toxicity control chart has been generated, with more points to be added as future results are obtained.

The possibility of using a test endpoint based on frond surface area was investigated in Test 1093L. Surface area (mm^2), measured from micro-photographs using the image analysis package, ImageJ, was based on the greenness of leaves using Hue (pure colour), saturation (intensity of colour) and brightness (amount of grey). When comparing the data (ie growth rate based on frond number versus growth rate based on surface area), surface area appears to be a more sensitive endpoint (Figure 3). This is because it measures 'greenness' (compared to the control), whereas counts of frond numbers include all fronds whether they are healthy or pale/ patchy (ie dead or near-dead), giving a slightly less sensitive result. These initial results suggest that surface area represents a suitable and measurable endpoint, although additional testing will be undertaken to confirm reproducibility.

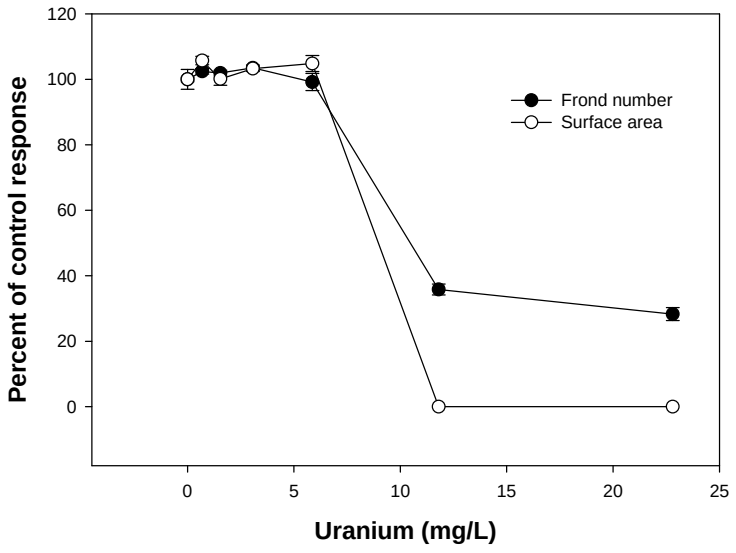

Figure 3 Comparison of growth rate based on frond number and surface area as endpoints to assess the toxicity of U to *Lemna aequinoctialis*

Planned testing in 2010–11

The reference toxicity testing programs for all five species will continue in 2010–11, with the aim of completing at least four tests per species. For *M. macleayi*, additional tests will focus on the role of FFV in influencing reproductive output as well as chronic U toxicity. In addition, further endpoint development will be undertaken for *L.aequinoctialis*. The reference toxicant test and frond surface area measurement methods will be documented in an Internal Report.

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Effects of magnesium pulse exposures on aquatic organisms

A Hogan, R van Dam, A Harford, K Cheng & C Costello

Background

Continuous monitoring of electrical conductivity (EC) in Magela Creek enables equivalent magnesium (Mg) concentrations to be derived since there is a very strong relationship between EC and Mg. These monitoring data have shown that peak Mg concentrations associated with pulse events at times exceed the provisional site-specific Limit for Mg (3 mg/L) in Magela Creek, and have, on one occasion, reached a maximum value of approximately 16 mg/L. However, the majority of these pulses occur over timescales of only minutes to hours. In contrast, the ecotoxicity data upon which the Mg provisional limit was derived are based on continuous exposures over three to six days (depending on the test species). Consequently, it was unknown if these shorter duration exceedances cause adverse effects on aquatic biota. To address this issue, an assessment of the toxicity of Mg under a pulse exposure regime has been ongoing since late 2008.

The approach taken to address this issue, and the results of 14 tests assessing the effects of pulse durations of 4 and 24 h on five local species, were previously described in detail by Hogan et al (2010). Generally, it was found that pulse exposures of ≤ 24 h were substantially less toxic than continuous exposures of four to six days. Additional work with the cladoceran, *Moinodaphnia macleayi*, showed that the life stage at which a pulse exposure occurred strongly influenced the sensitivity of the response.

This summary provides an update on the 4 and 24 h pulse duration tests conducted during 2009–10, along with data on an additional intermediate test duration of 8 h.

Methods

Four local test species (duckweed, hydra, cladoceran and fish) were exposed to a single Mg pulse of 4, 8 and 24 h duration. In each test the organisms were exposed to a range of Mg concentrations except for the fish, which, due to its relative insensitivity, was only exposed to one very high concentration (4 g/L Mg). The pulse was administered at the beginning of the test, after which time the organisms were returned to natural Magela Creek water for the remainder of the standard test period (four to six days).

Additional tests conducted for *M. macleayi* involved administering a 4 h pulse around the time of the onset of reproductive maturity (27 h) to further investigate the importance of life-stage on the response of the cladoceran to the Mg pulse.

The results from all tests were compared with those from tests where the organisms were continuously exposed to Mg throughout the standard test period.

Results

The responses of the duckweed (*Lemna aequinoctialis*), the green hydra (*Hydra viridissima*) and *M. macleayi* to pulse durations of 4, 8 and 24 h are presented in Figure 1. The response of

each organism to a continuous exposure of Mg (solid line) is also included for comparative purposes.

For all three species, toxicity decreased with a reduction in exposure period. However, the degree to which toxicity was reduced was found to be species dependent. The toxicity estimates for each species and duration combination will be reported separately. Since each of these species is from different taxonomic groups the differences in their anatomy and physiology would likely result in markedly different uptake, assimilation and depuration rates for Mg. Each of these factors would influence the magnitude of response of each organism over a particular exposure period (eg Diamond et al 2006).

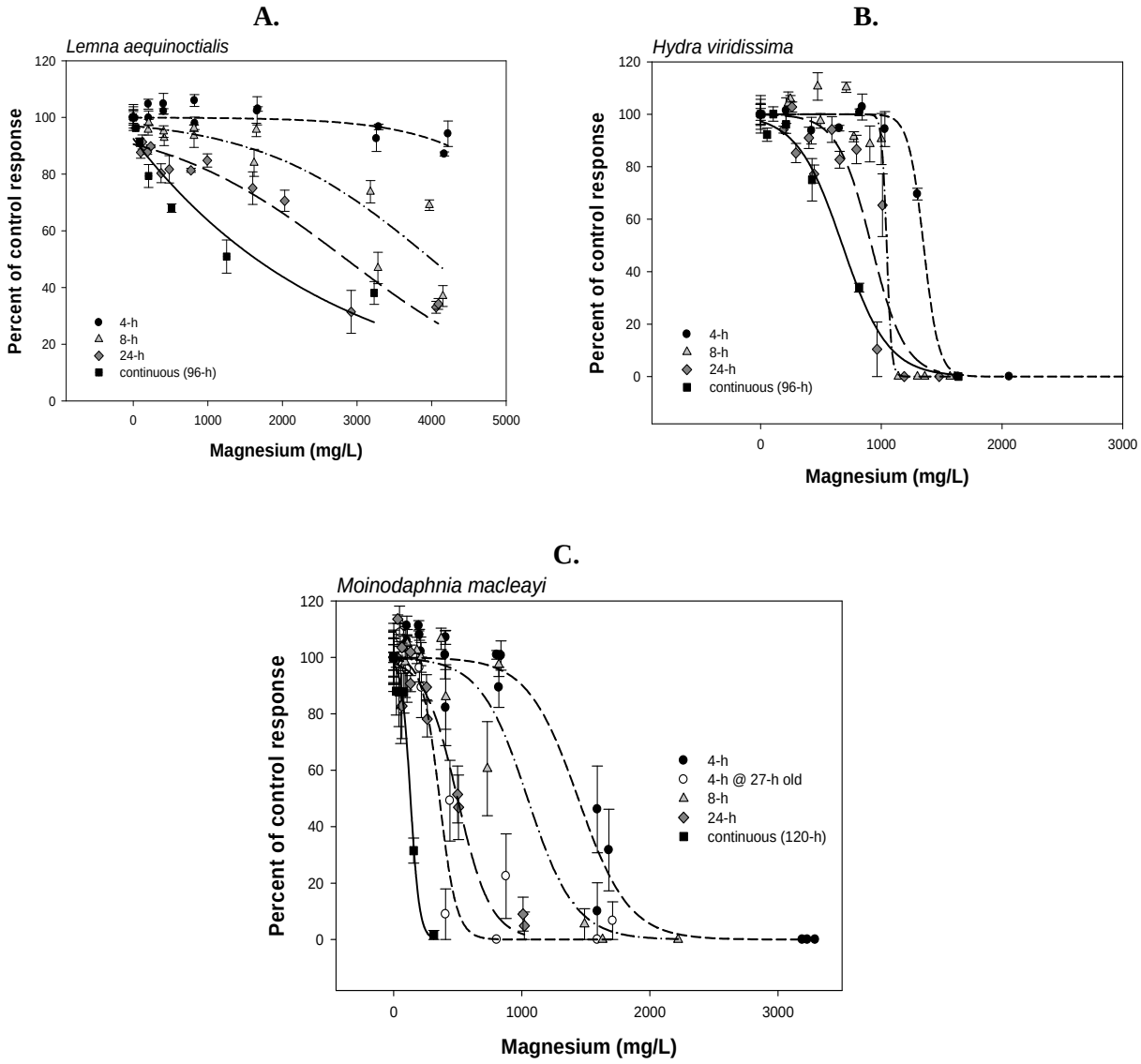

Figure 1 Toxicity of magnesium to A. duckweed, *Lemna aequinoctialis*, B. green hydra, *Hydra viridissima*, C. cladoceran, *Moinodaphnia macleayi*. Data from continuous exposure experiments are represented by a solid line, while 4 h pulse data are represented by short dashed lines, 8 h pulse data by dashed-dotted lines and 24 h pulse data by long dashed lines.

Data have not been presented graphically for the fish (*Mogurnda mogurnda*) because this species was not affected at all by 4, 8 or 24 h exposures of up to 4 g/L Mg. This result was not surprising given that *M. mogurnda* was found to be relatively insensitive to Mg for a chronic exposure regime (concentration that was lethal to 50% of the test organisms, LC50 = 4054 mg/L Mg; van Dam et al 2010).

As noted by Hogan et al (2010), the response of *M. macleayi* appears to be dependent on the life-stage at which this species is exposed to Mg. A 4 h exposure of Mg administered when the cladocerans reach reproductive maturity (27 h of age) was more toxic than when the cladocerans were exposed at ~3 h of age (Figure 1C, open circles versus solid circles). The appearance of eggs (indicating reproductive maturity) has been noted to coincide with a moult that allows for the swelling of the brood pouch. It is yet to be determined if the increase in sensitivity around 27 h of age is due to the occurrence of the moult, and subsequent increase in permeability of the cladocerans carapace, or a change in physiology as they reach reproductive maturity. In any case, it is the data from tests incorporating the sensitive life stage in the exposure period that will be used for trigger value derivation.

For the species tested thus far, the concentrations that exhibited toxic effects were much greater than the maximum concentration of Mg (16 mg/L) that has been reported in Magela Creek downstream of the mine. Even in the most sensitive test where *M. macleayi* was exposed for four hours at the onset of reproductive maturity, the concentration of 136 mg/L Mg that caused a 10% inhibition of the test endpoint (IC10; generally considered an 'acceptable' level of effect), was still 8 times higher than the reported maximum Mg concentration in Magela Creek. A final assessment of the risks posed by pulses of Mg to aquatic biota will be made once Mg pulse trigger values are derived and a trigger value versus exposure duration relationship is established (see below).

Conclusions

Work in 2009–10 has supported the earlier conclusions that Mg pulse exposures are substantially less toxic than for equivalent concentrations under a continuous exposure regime. It was also observed that the degree to which toxicity is reduced by a shorter duration exposure is both species and life-stage specific.

Negative response effects observed during toxicity testing for the species thus far tested have occurred at concentrations much higher than those that have been measured to date in Magela Creek. However, it is the trigger value that is ultimately derived from the multispecies data set, rather than the test results to date from individual species, that needs to be used to provide a more robust estimate of the risks posed to aquatic ecosystems by a given magnitude and duration of exposure to Mg.

Steps for completion

In total, at least two 4, 8 and 24 h pulse experiments need to be completed for each of the six species routinely tested at ***eriss*** to provide a high confidence interpretative framework for assessing the magnitude of risk posed by a given pulse exposure scenario. Two further 8 h pulse tests need to be completed for *M. macleayi* whereby the cladocerans are exposed to the pulse during their more sensitive life-stage. Some technical challenges remain to be overcome to provide reliable pulse test results for the unicellular alga *Chlorella* sp. Specifically an effective non-disruptive method for washing the algae is required. Separating the algal cells from the $MgSO_4$ solution using dialysis instead of centrifugation is currently being

investigated. Testing the effects of the three pulse durations to the gastropod, *Amerianna cumingi*, will be done in early 2011.

Once data are available for all six test species for each pulse duration, trigger values will be derived for each exposure duration using the species sensitivity distribution approach. From these data, a relationship between TV and exposure duration will be established that will underpin the implementation of an exceedance monitoring framework for Mg/electrical conductivity in Magela Creek.

The next phase of the research may involve testing multiple pulses, given that pulse frequency and the length of the recovery period between pulses are also important. Further research into toxicokinetic-toxicodynamic modelling may also be warranted, considering that pulsing scenarios in Magela Creek can be quite complex when multiple exceedances of a trigger value occur over a relatively short timeframe. As these scenarios may not always be matched to those tested in the laboratory, there may be a future need for the predictive ability that could be achieved through such modelling approaches (see Ashauer et al 2006, Diamond et al 2006).

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Toxicity testing of Ranger process water permeate

R van Dam, A Hogan, A Harford, K Cheng & C Costello

Background

Active treatment of process water at Ranger was implemented in late 2009 to accelerate reduction of the process water inventory. Untreated process water typically has a pH of ~4, an electrical conductivity > 25 000 μS/cm and contains highly elevated concentrations of sulfate – >30 000 mg/L, magnesium – >5000 mg/L, total ammonium ~900 mg/L, uranium (U) – >25 mg/L, aluminium – >400 mg/L and manganese – >2000 mg/L). The treatment of process water comprises lime and carbon dioxide softening, followed by microfiltration/ultrafiltration, and finally reverse osmosis (Topp et al 2003). The water treatment plant was designed to produce water to a standard such that the treated water, after an additional passive wetland polishing treatment, would be suitable for release to the off-site aquatic environment with no measurable biological impact. The final wetland step was specifically intended to remove residual ammonia (present in solution as ammonium ion) given that it was anticipated that the reverse osmosis treated water (permeate) from the water treatment plant could contain up to 20 mg/L of this species. Ammonia is both a toxicant and a nutrient so it is important that its concentration is reduced to environmentally acceptable levels prior to release of the final treated water.

A key question to be addressed from both an operational and environmental perspective, notwithstanding the wetland biopolishing step, was the extent to which the permeate contained residual toxicity, and whether this toxicity could be accounted for by the ammonium present. Toxicity testing in 2001 of the permeate produced from a pilot water treatment plant indicated low toxicity to three aquatic species, with IC/LC50 ratios ranging from 44% to >100% permeate (Camilleri et al 2002). The aims of the present study are to (i) assess the toxicity of permeate from the full scale treatment plant commissioned at Ranger mine and, if residual effects were observed, to (ii) identify the cause/s of the effects.

Methods

Commissioning of the process water treatment plant at Ranger was completed in October 2009. On 26 October 2009, following advice from ERA that the permeate being produced was representative of typical outputs, SSD staff collected a sample for toxicity testing in the SSD Darwin laboratories. Separate samples of the permeate were collected for analysis of chemical constituents. The toxicity of the permeate was assessed using the following five local species: unicellular green alga (*Chlorella* sp); macrophyte (duckweed; *Lemna aequinoctialis*); cnidarian (*Hydra viridissima*); cladoceran (water flea; *Moinodaphnia macleayi*); and a fish species (*Mogurnda mogurnda*). The test species were exposed to concentrations of 6.25%, 12.5%, 25%, 50% and 100% permeate (diluted in Magela Creek water), and a Magela Creek water control. The toxicity testing methods are detailed by Riethmuller et al (2003).

Results

The chemical composition of permeate is compared with process water and Magela Creek water in Table 1. The treatment process was highly effective in removing major ions and metals from process water, including U. Analytes present in the permeate at concentrations substantially above those of natural Magela Creek water included ammonia (6.7 mg/L, as total ammonia-N), boron (236 μg/L), bromine (49 μg/L), rubidium (4 μg/L) and rhenium (10 μg/L). The ammonia concentration, although greatly reduced by the treatment process , was still at least seven times higher than the Australian and New Zealand water quality trigger value of 0.9 mg/L applying at the pH of the permeate (pH 8) (ANZECC/ARMCANZ 2000). Existing toxicity data suggest that the other analytes listed above were unlikely to be a concern.

Table 1 Water quality of Magela Creek water and untreated and treated process water from Ranger uranium mine

Variable	Magela Creek water	Process water	
		Untreated [a]	Treated
pH	6.2	3.9	8.3
Electrical conductivity (μS/cm)	18	28 200	91
Dissolved organic carbon (mg/L)	2.6	NM[b]	1.5
NO_3_N (mg/L)	<0.005	1.77[c]	0.005
NH_3_N (mg/L)	<0.005	1040[c]	6.8
Ca (mg/L)	0.2	602[c]	<0.1
Mg (mg/L)	1.1	6390[c]	<0.1
Na (mg/L)	1.3	97[c]	4.9
SO_4 (mg/L)	0.3	38 600[c]	2.4
Al (μg/L)	5.5	491 000	3.1
B (μg/L)	12	NM	236
Br (μg/L)	10	NM	49
Cu (μg/L)	0.23	12 600	0.2
Fe (μg/L)	40	10 300	<20
Mn (μg/L)	2	2 520 000	<0.01
Pb (μg/L)	0.22	4480	0.05
Rb (μg/L)	0.5	NM	4
Re (μg/L)	<0.05	NM	10
U (μg/L)	0.005	32 300	0.074
Zn (μg/L)	0.5	6130	<0.1

a Unless otherwise stated, values for untreated process water represent measurements from a sample collected on 2 November 2009, one week after the permeate sample for toxicity testing was collected. Data supplied by Energy Resources of Australia Ltd (ERA).

b NM: Not measured

c Values supplied by ERA from a sample collected on 30 November 2009

Although the concentration of U (0.07 μg/L) in the permeate was an order of magnitude greater than background concentrations in Magela Creek water, it was two orders of magnitude lower than the derived site-specific Limit for uranium in Magela Creek of 6 μg/L (Hogan et al 2005). Hence U is not a toxicant of concern in the permeate sample submitted for testing.

Significant effects of permeate were observed for all species above a concentration of 12.5%, with the responses ranging from growth stimulation to moderate toxicity (Figure 1, Table 2).

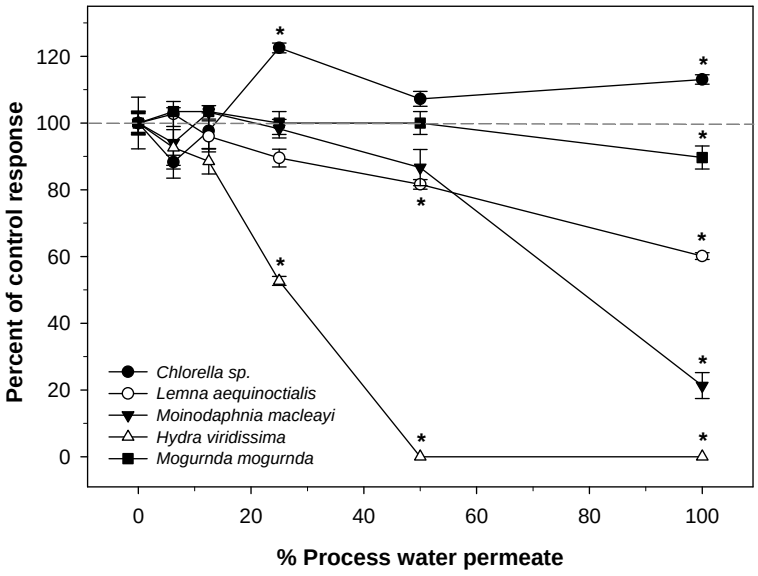

Figure 1 Responses of five tropical freshwater species to treated process water from Ranger Uranium Mine, expressed as percentages of the control response (see Table 2 for control response data). Data points represent the mean ± standard error of three replicates (10 replicates for *Moinodaphnia macleayi*). Asterisks denote treatments that are significantly different ($P \leq 0.05$) from the control response.

Table 2 Toxicity estimates for treated process water from Ranger uranium mine

Species	Control response (mean ± standard error)	Toxicity (% process water permeate)	
		IC10[a] (95% CL)[b]	IC50[a] (95% CL)
Chlorella sp	Doublings per day = 1.6±0.04	NC[c]	NC
Lemna aequinoctialis	Growth rate = 0.43±0.01	22 (0–45)	NC
Moinodaphnia macleayi	Offspring per adult = 35.2±2.7	43 (5–54)	78 (69–83)
Hydra viridissima	Growth rate = 0.31±0.01	10 (0–18)	26 (23–29)
Mogurnda mogurnda	Percent survival = 97±3	67[d] (0–100)	NC

a IC10 and IC50: concentrations that result in a 10% and 50% inhibition of response compared to the control (ie unexposed) response, respectively. Estimates were derived using linear interpolation (ToxCalc V5.0.23).

b 95% CL: 95% confidence limits

c NC: Not able to be calculated since there was insufficient response across the dilution gradient

d Value represents an LC05 (ie concentration resulting in 5% mortality of larval *M. mogurnda*; derived using non-linear interpolation; ToxCalc V5.0.23). A lower effect level than 10% was selected given the test is an acute test.

Chlorella sp growth rate was significantly enhanced at permeate concentrations of 25% (22% enhancement compared with the control) and 100% (13% enhancement). Exposure of *L. aequinoctialis*, *M. macleayi*, *H. viridissima* and *M. mogurnda* to 100% permeate resulted in significant reductions in responses of 40%, 80%, 100% and 10%, respectively. *Hydra viridissima* exhibited the strongest response of all the species, with a full response at 50% permeate and 47% reduction in growth rate at 25%. Based on the extent of response (negative or positive) at 100% permeate, the order of sensitivity of the species (from highest to lowest) was: *H. viridissima* > *M. macleayi* > *L. aequinoctialis* > *Chlorella* sp ≈ *M. mogurnda*.

The process water treatment process is clearly effective at removing the majority of contaminants and hence reducing or eliminating toxicity, compared with the composition of the untreated process water.

The effects of the reverse osmosis permeate, including the stimulatory response by *Chlorella* sp, are hypothesised to be primarily due to residual ammonia (present largely as ammonium ion). Alternatively, or in addition, the adverse responses of some of the species could be due to the very low concentrations of nutrients (other than N) or essential trace elements in permeate preventing normal growth, development and/or survival. This was previously shown to be the case for treated pond water permeate from Ranger (Hogan et al 2009).

Steps for completion

Additional work is being undertaken to confirm if the effects of permeate are largely caused by the residual concentration of ammonium ion. This will involve the selective removal of ammonia (as ammonium) from the permeate followed by toxicity testing of the residual solution.

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The toxicity of uranium (U) to sediment biota of Magela Creek backflow billabong environments

A Harford, R van Dam, C Humphrey, D Jones, S Simpson[1], J Stauber[1], K Gibb[2] & C Streten-Joyce[2]

Background

There is currently a paucity of data concerning the impact of uranium (U) contaminated sediments on benthic biota. Internationally there have been very few studies that have focused on this issue and the toxicity estimates reported have varied by at least three orders of magnitude (Dias et al 2008, Lagauzère et al 2009). In the only Ranger-related site-specific sediment toxicity study that has been conducted to date, Peck et al (2002) reported that the local chironomid (midge), *Chironomus crassiforceps*, was not affected by sediment U concentrations up to 5000 mg/kg dry weight. Conversely, studies using other species have reported significant effects at concentrations as low as 3 mg/kg (Dias et al 2008). Good quality sediment U toxicity data are required to help determine if observed differences in populations of benthic biota in billabongs adjacent to the Ranger Uranium Mine are due to U in sediments or to other mining or non-mining related factors. For mine closure, sediment quality criteria will also be required for on-site sentinel wetlands, which will serve to capture and 'polish' seepage and runoff waters from the rehabilitated minesite, as well as for downstream receptor wetlands. Thus, an ***eriss*** research project is underway, in collaboration with CSIRO Centre for Environmental Contaminants Research (CECR) and Charles Darwin University (CDU), to address the knowledge gaps concerning this issue and to develop a site-specific sediment quality guideline for U in billabongs and creeks. Further background and context for this project has been given in van Dam et al (2010).

During 2009, a field sediment U toxicity study commenced in Gulungul Billabong, which is a largely undisturbed billabong at the confluence of Gulungul and Magela Creeks. An initial chemical and biological characterisation of the study site was undertaken in April 2009, focusing on sediment chemistry, microbes, microzoobenthos and macroinvertebrates. The results of this 'baseline' sampling were used to optimise sub-sampling methods and other aspects of experimental design. Chemical analyses of the sediments showed that the background concentrations of U were ~5 mg/kg dry weight. The results of the biological analyses were summarised in van Dam et al (2010).

During the 2009–2010 wet season, a pilot-scale experiment was undertaken at the study site. The study aimed to determine the appropriate methods, U concentration range and replication required for a full scale experiment. The pilot study involved the deployment at the study site of U-spiked sediments in retrievable containers over the duration of a wet season. At the end of the exposure period, the extent of colonisation of macroinvertebrate, microinvertebrate, biofilm and microbial communities was measured in the control and test replicates.

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Methods

Moist sediment (~150 kg) was collected to a depth of 5–10 cm from the exposed littoral zone at the study site in August 2009. The sediment was transported to ***eriss*** where it was prepared for the pilot study. Preparation involved freezing for one week (to sterilise the samples, acknowledging, however, that a number of microinvertebrate forms may survive freezing; Dr Russ Shiel, University of Adelaide, pers comm), and wet sieving through a 2 mm mesh size with deionised water in order to create a slurry (1:1.4 sediment:water) for spiking and mixing. The slurry was split into four 30-L batches for the following treatments: control; 5400 mg/kg sulfate control (as $Na_2SO_4.10H_2O$); 400 mg/kg U; and 4000 mg/kg U (as $UO_2SO_4.3H_2O$). Each sediment batch was placed into three 20-L plastic buckets each with 10 L of slurry. The buckets were sealed and were mixed in a cement mixer for 1 h once every two days until each bucket had been mixed 7 times (14 days).

Following mixing, the sediments for each treatment were recombined in a 50-L Nally® bin, mixed thoroughly, and placed in a cool room in the dark at 4°C for an initial equilibration period to allow the adsorption of spiked metal (or ion) to the sediment. After 21 days, the sediments were removed from the cool room and placed outdoors (still in the Nally® bins) at ambient temperature (24–35°C) for 10 days to dry to a point where they could be transferred to the experimental containers. Sub-samples for sediment and porewater chemistry were collected on days 0, 7, 14, 21 and 28 post-mixing.

Sub-samples of the bulk sediments were then transferred to the experimental containers (~20 × 20 × 15-cm plastic containers with ~5 mm mesh size sides and base). For each treatment, approximately 2 L (or 2000 cm^3 – 20 × 20 × 5-cm) of sediment was placed into each of nine replicate containers. The test containers were then placed in holding containers, covered, and left in a cool room in the dark at 4°C prior to their deployment in the field.

A total of 45 test containers were deployed in the field on 15 December 2009, which was approximately 2 weeks before the site was inundated. Nine replicates of a field control were also produced (ie Gulungul control) and consisted of sediment from the site, which was wetted on site and placed in a test-container. Treatment groups were arranged in a stratified random sequence to allow for slight environmental differences across the study site.

The containers were retrieved at the end of the wet-season on 30 March 2010. Containers were recovered by a person with a snorkel and mask, and immediately placed in a bucket of Gulungul Billabong water until they could be processed at the Jabiru Field Station. Sub-samples of each replicate were taken for chemical analysis at CSIRO CECR, and bacterial analysis by eco-genomics at CDU. The remaining sediment was sieved through 500, 125 and 63 µm mesh sizes with copious water. The three size fractions were then placed in absolute ethanol for macroinvertebrate (ie 500 µm) analysis at ***eriss*** and microinvertebrate (125 and 63 µm) analysis at the University of Adelaide.

Results

Table 1 reports chemistry data for the field control and laboratory treated sediment samples, both pre-deployment and post-retrieval. Chemical analyses of sub-samples taken during the spiking process showed that U binding to the sediment was rapid and complete. Additionally, the only measured difference between the treatments was the U content of the sediments. There was very little loss of U over the course of the wet-season and the U was evenly distributed through the depth of the test containers. As a result, chemical analyses of natural billabong sediment samples taken 10 cm adjacent to a high U treatment showed very little elevation in U concentrations.

Table 1 Key metals (total extractable) in the field-collected and laboratory-treated sediments

Element (mg/kg)	Field control	Pre-deployment [a]				Post-retrieval [b]				
		Control	5400 mg/kg sulfate	400 mg/kg uranium	4000 mg/kg uranium	Control	5400 mg/kg sulfate	400 mg/kg uranium	4000 mg/kg uranium	10 cm adjacent to 4000 mg/kg uranium
U	6	6	6	535	4220	6	6	532	4160	14
Al	57100	41100	53100	45300	50300	39200	43700	44400	42600	48200
Co	12	12	12	11	12	15	17	22	14	14
Cr	38	33	37	35	34	33	35	37	34	36
Cu	27	26	26	26	28	24	22	24	24	24
Fe	12600	11200	12550	11900	11500	10800	11000	11500	11000	12966
Mn	56	57	61	61	57	59	61	65	59	129
Ni	22	15	20	17	20	18	18	19	18	20
Pb	11	13	12	13	15	12	12	16	13	12
Zn	14	11	13	11	12	<60	<60	<60	<60	19

[a] The results are from one measurement made of the batch treatments sampled on 2 October 2009 before distribution to individual test containers.

[b] The results are from one individual replicate.

Biological analyses of the sediments from the study were potentially confounded by the inadvertent creation of highly compacted, fine grain-sized sediment that became 'mud bricks' that were not representative of the natural sediment. These appeared to be non-conducive to macroinvertebrate (and possibly other faunal) colonisation. The 'mud bricks' were created as a result of the large amount of sediment manipulation – specifically, the removal of coarser organic matter and an extended period of homogenisation of the U-spiked sediments, followed by 'sun-baking' of the fine-grained material in the field prior to the start of the first rains.

Preliminary macroinvertebrate assessment found generally low abundance and species richness with no apparent effect of U on either endpoint (although further sample analysis is required to confirm this). In contrast, preliminary (multivariate analysis) data for the microbial communities showed an apparent effect at 4000 mg U/kg, but not at 400 mg U/kg, compared with the control (Figure 1). The microbial results also confirmed a difference between the spiking control and Gulungul control sediments. The sulfate control also appeared to be different from the Gulungul control, although there was some variability among replicates, and the data have not yet been statistically analysed for significant differences (Figure 1). The microzoobenthos samples were not assessed as originally planned, due to the difficulties in sorting very small and often cryptic organisms from the fine sediment. Instead, microzoobenthos samples will be analysed (by CSIRO CECR) using a similar metagenomic approach as used for the microbes at CDU. The use of an artificial substrate for algal biofilm colonisation was also trialled. Specifically, a plastic mesh was secured to the surface of the sediments and during the retrieval process this was removed, rinsed and placed in Lugol's solution for species identification by traditional taxonomy at CSIRO. There was little difference between controls and U treatments for the biofilm communities. However, these results are unreliable as it was observed during both sediment deployment and retrieval that there was limited contact of the mesh with the sediment. Future studies will quantify sediment algal biofilm communities through the ecogenomic approach described above for microzoobenthos, as this includes all eukaryotes (ie algae and microzoobenthos but not bacteria).

To date, the pilot study results have shown that biological effects of U in sediments can be measured, but that the sediment spiking method needs to be further refined to minimise physical disruption of the sediment structure. The only way to overcome this problem is to minimise (i) the amount of sediment manipulation, in particular, the sieving and the extent of mixing, and (ii) aerial exposure prior to wet season inundation. However, at the same time, it will be necessary to ensure a reasonably/acceptably homogeneous distribution of U throughout the sediment layer.

Steps for completion

During the 2010–11 wet season, a second pilot scale field study will be undertaken to assess the suitability of a modified sediment preparation method on:

(i) the sediment physical characteristics,

(ii) U distribution and

(iii) faunal colonisation.

Consequently, the main experiment to quantify the effects of U on sediment biota of Magela Creek backflow billabong environments has been delayed until the 2011–12 wet season.

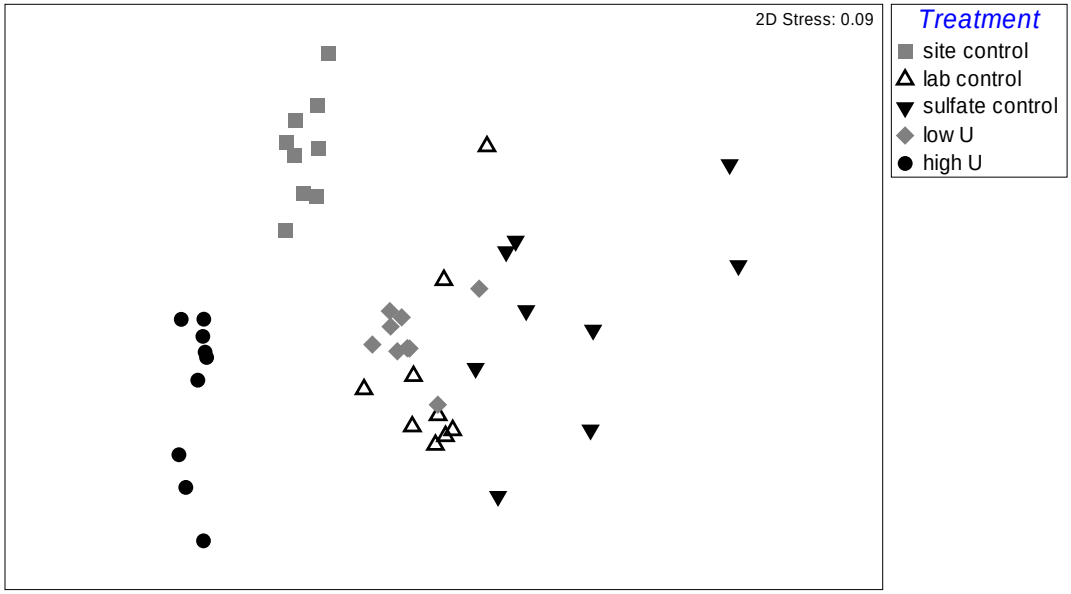

Figure 1 MDS (Multi Dimensional Scaling) ordination of microbial communities measured in treatment replicates through ecogenomics for the site (Gulungul) control and sediments treated with U and pure water (lab control). Low uranium = 400 mg/kg and high uranium = 4000 mg/kg.

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Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer, R Cahill, J Pfitzner & J Matthews

Introduction

The inhalation pathway has previously been identified as the main contributor (albeit a very small proportion of the total dose) to public radiation dose from the Ranger minesite for an adult living in Jabiru and working in Jabiru East during the operational phase (Martin 2000). Both Energy Resources of Australia Ltd and SSD monitor the inhalation exposure pathways in the region. The two potential pathways are radioactivity trapped in or on dust (or long lived alpha activity, LLAA) and radon decay products (RDP). Of these two airborne pathways, RDP accounts for most of the dose received by the public (Supervising Scientist 2007, ERA 2009).

The dose limit to the public for exposure from a planned exposure situation (such as an operating uranium mine) recommended by the International Commission on Radiation Protection is 1 milli Sievert (mSv) per year. This limit applies to the sum of all pathways and relevant practices to which people could potentially be exposed. Furthermore, the ICRP (2007) recommends that in order to optimise radiation protection for planned exposure situations a public dose constraint should be selected that is 'less than 1 mSv and that a value of no more than about 0.3 mSv would be appropriate'. Consequently, a dose constraint of 0.3 mSv has been applied when assessing radiological monitoring data for the Ranger mine.

Since the main areas of habitation in the vicinity of Ranger and Jabiluka are Jabiru, Mudginberri and Jabiru East, the SSD monitoring program focuses on those three population centres. RDP and LLAA concentrations in the air are measured and the quarterly-averaged results are compared with those from ERA's atmospheric radiological monitoring program (Supervising Scientist 2010).

Results

Radon pathway

Figure 1 shows the quarterly averaged RDP concentrations from Jabiru Water Tower, Jabiru East and the Mudginberri Four Gates Road Radon Station measured by ***eriss*** from January 2005 to June 2010. Environmental Radon Decay Product Monitors (ERDM; *Radiation Detection Systems, Adelaide*) were acquired in 2008 to replace the Alphaprisms (*alphaNuclea*r, Saskatoon, Canada) that were used previously. Since 2009, three ERDMs have been used for RDP monitoring. During the reporting period data were acquired continuously at Mudginberri Four Gates Road Radon Station and for periods of up to one month at a time at Jabiru and Jabiru East.

A two sample t-test shows there is no statistically significant difference ($p = 0.49$) between the quarterly RDP concentrations measured at Jabiru Water Tower and Mudginberri Four Gates Road Radon Station, the latter of which is considered a background site. The average RDP concentration measured from July 2003 to June 2010 at the Mudginberri and Jabiru Water Tower sites is 0.044 $\mu J/m^3$. The Jabiru East values are significantly higher ($p < 0.05$)

with the average being 0.069 μJ/m^3. RDP concentrations at Jabiru East also show more variation due to the closer proximity of Jabiru East to the mine pit and ore stockpiles, the largest localised sources of radon in the area. Annual averages for the three sites over the past three years are shown in Table 1.

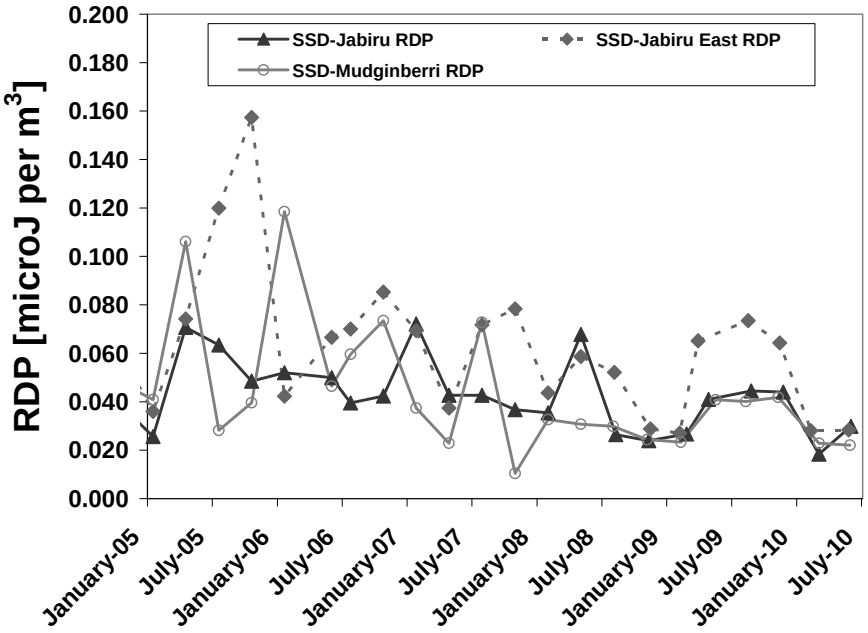

Figure 1 Radon decay product concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

Jabiru is sufficiently distant from the mine that most of the mine origin radon has dispersed to very low levels. Variations in measured radon concentrations are mainly caused by diurnal meteorological effects and the annual cycle of wet and dry seasons. Airborne radon concentrations are generally lower during the wet season, as radon exhalation from the soil decreases with increasing soil moisture content. The influence of other factors such as soil ^{226}Ra activity concentration, soil morphology, and vegetation cover have been investigated and reported previously (Lawrence et al 2009).

Table 1 Average radon decay product concentrations (ERA 2010 in brackets) at Jabiru, Jabiru East and Mudginberri, and associated total and mine derived annual doses received at Jabiru, between 2006 and 2009. Differences in RDP concentrations are due to differences in time of sampling and different sampling sites for *eriss* and ERA at Jabiru East.

		2007	2008	2009
RDP concentration [μJ/m^3]	Jabiru East	0.064 (0.059)	0.046 (0.033)	0.057 (0.095)
	Jabiru	0.049 (0.038)	0.038 (0.037)	0.039 (0.062)
	Mudginberri	0.036	0.029	0.037
Total annual dose [mSv] Jabiru		0.47 (0.37)	0.37 (0.36)	0.38 (0.61)
Mine derived dose [mSv] at Jabiru[a]		0	0.01	0.03

[a] predicted from wind field model and reported by ERA

ERA estimates the mine origin RDP using a wind correlation model and calculates the mine derived dose from the inhalation of RDP. Table 1 shows the annual averages at Jabiru and Jabiru East for the total RDP concentrations measured by ***eriss***, and reported by ERA, together with the calculated total and mine derived annual doses from RDP inhalation. This

dose calculation assumes a full time occupancy of 8760 hrs (1 year) and a dose conversion factor for the public of 0.0011 mSv per μJ/hr/m^3. The RDP concentration for the mine-related dose calculated for 2009 is about 10% of the public dose constraint.

Continuous data from Mudginberri Four Gates Road Radon Station

Since early 2009 an ERDM has been deployed permanently at Mudginberri Four Gates Road Radon Station. As noted above this is considered a background site, similar to the Djarr Djarr site in the Alligator Rivers Region (Martin et al 2004). The ERDM logs data continuously, and is downloaded and serviced monthly. Figure 2 shows the results for July – December 2009. The data gap between 9 November and 8 December was due to the instrument being out of service for repair.

The plot highlights the daily variations in RDP concentrations and the large differences that exist between day and nighttime RDP concentrations. This diurnal pattern is due to atmospheric inversion layers that form in the early morning hours and hence prevent mixing of the air and dispersion of radon. This means that radon exhaled from the earth's surface is effectively trapped, with RDP reaching concentrations up to 10 times the annual average (see Figure 2). This radon and its decay products are usually dissipated by atmospheric mixing produced by the onset of wind later in the day. However, the inversions can sometimes persist for extended periods (for example around 8 October 2009) and RDP concentrations may not reach their normal daytime minimum. The inversions can result in up to a factor of 3 increases in average 24hr RDP concentrations between consecutive days. .

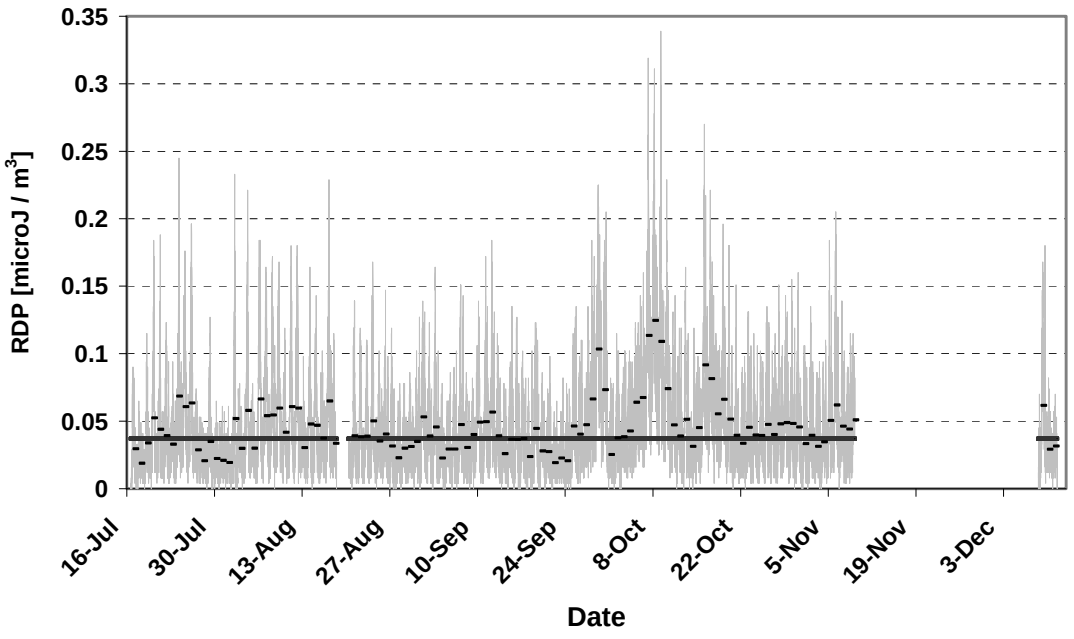

Figure 2 Continuous radon decay product concentrations (grey) measured at Mudginberri Radon Station between 16 July and 11 December 2009. The annual average (solid black line) and daily averages (black dashes) are overlaid for comparison.

Continuously recording ERDM units have now been deployed at Jabiru and Jabiru East. It is envisaged that data will be logged continuously (apart from gaps caused by instrument maintenance and repair) at these two sites in the future. These continuous data will allow a more robust and reliable average at these sites to be produced for future reporting years.

Dust pathway

Atmospheric dust activity concentrations are routinely monitored by both ***eriss*** (Jabiru, Jabiru East and Mudginberri radon station) and ERA (Jabiru and Jabiru East). Figure 2 shows the LLAA at Jabiru, Jabiru East and Mudginberri measured by ***eriss*** from January 2005 to July 2009. In 2007, permanent dust samplers were installed at Jabiru and the Mudginberri Four Gates Road Radon Stations to facilitate uninterrupted sampling of LLAA at these locations.

Similar to the atmospheric radon concentration, the dust concentration is lower during the wet season due to the higher soil moisture content that suppresses dust generation. Generally, the LLAA concentration is higher at Jabiru East due to its closer proximity to the mine. The averages measured from July 2003 to June 2010 at Jabiru, Jabiru East and Mudginberri are 0.00020, 0.00031 and 0.00017 $Bq \cdot m^{-3}$, respectively. There is no statistically significant difference between the Mudginberri and Jabiru sites ($p = 0.289$).

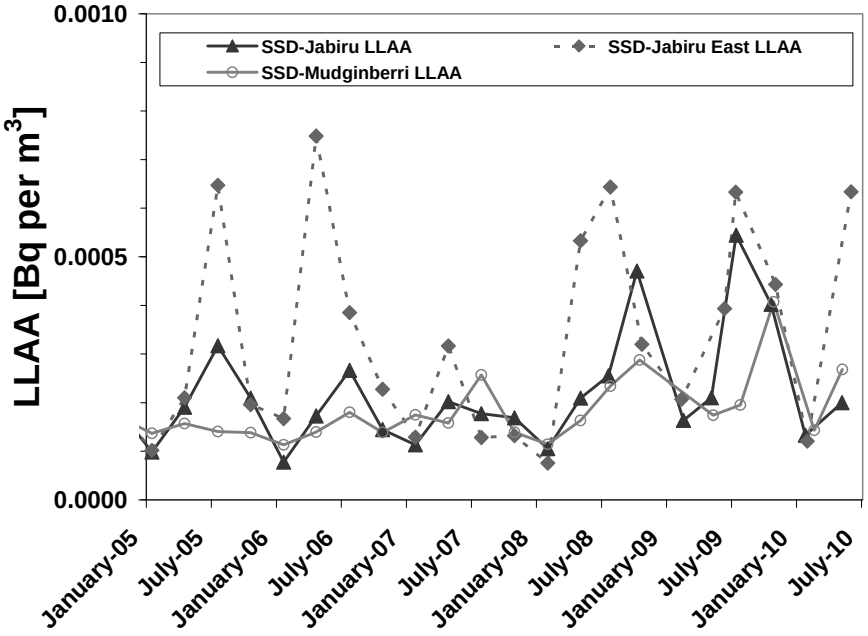

Figure 2 Long lived alpha activity concentration measured at Jabiru, Jabiru East and Mudginberri

The total annual dose from inhalation of dust is calculated using a dose conversion factor of 0.0057 mSv per alpha decay per second (Zapantis 2001) and a breathing rate of 7300 m^3 per year for adults (UNSCEAR 2000). The total dose in Jabiru for 2009 was 14 μSv per annum , with only a small fraction (ie ~2 μSv for a person working in Jabiru East and living in Jabiru) of that dose being be mine-related (Bollhöfer et al 2006).

Conclusion

The routine monitoring of dust and radon progeny will be maintained at Jabiru, Mudginberri Four Gates Road Radon Station and Jabiru East. Permanent dust samplers have now been installed and continuous RDP monitors have been acquired and tested at the three monitoring sites. The continuous RDP data will be imported into SSD's Hydstra® database, and future reporting and data analysis will be conducted using the Hydstra system.

Monitoring of the radon and dust exposure pathways over the past 9 years has shown that the main contributor to radiological exposure of the public at Jabiru via inhalation is the inhalation of radon decay products (RDP). RDP are now monitored continuously at Mudginberri Four Gates Road Radon Station, and continuous RDP data will be available in 2010–11 for Jabiru and Jabiru East. Although the contribution from the minesite has been shown consistently to be much less than the public dose constraint of 0.3 mSv per year at Jabiru and is of no concern according to current best practice standards, atmospheric monitoring will be continued to provide independent assurance to the public that the risk from inhalation of mine derived radionuclides remains very low.

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Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

P Medley & A Bollhöfer

Introduction

Surface water samples in the vicinity of the Ranger project area are routinely collected and measured for their radium-226 (^{226}Ra) activity concentrations to check for any significant increase in ^{226}Ra levels downstream of the impacted areas. This is due to the potential risk of increased exposure to radiation via the biophysical pathway due to mining activities. Mussels, in particular, bioaccumulate ^{226}Ra, which may then be incorporated into the human body upon consumption (Martin et al 1998, Bollhöfer et al 2010). Water samples are collected weekly in Magela Creek (Ranger) from both upstream and downstream sites. Water samples have also been collected monthly from the Ngarradj Creek (Jabiluka) downstream site, but sampling has now ceased with the last samples collected in the 2008–09 wet season. Jabiluka has been in long-term care and maintenance since 2003 and sufficient baseline data have been accrued. Samples are not collected during periods of no contiguous surface water flow (ie during the dry season).

Measuring the activity concentrations of ^{226}Ra does not by itself identify the source of radium in the environment. However, the activity concentration ratio of ^{226}Ra and ^{228}Ra can potentially be used as a signature to pinpoint the source of radium (Bollhöfer & Martin 2003, Medley et al 2010). This is the case since ^{226}Ra is a member of the ^{238}U decay series while ^{228}Ra comes from the decay of thorium-232 (^{232}Th). Therefore increases in the ^{226}Ra/^{228}Ra activity concentration ratios can be used to infer if the measured radium is derived from a primarily uraniferous source (eg a uranium mine). The differences in ^{226}Ra/^{228}Ra activity concentration ratios between upstream and downstream sites in Magela Ck can be used to indicate the proportion of ^{226}Ra at the downstream site that is coming from the Ranger mine. This will be especially important in the event that elevated downstream levels are detected

Samples are analysed for total ^{226}Ra (ie dissolved plus particulate phase) via alpha spectrometry in the ***eriss*** environmental radioactivity laboratory using a method described in Medley et al (2005). Alpha spectrometry is also used for ^{228}Ra determination after allowing for ingrowth of the ^{228}Th daughter (Medley 2010). In low-level samples it can take several years for sufficient ^{228}Th activity to accumulate for a reliable determination of ^{228}Ra activity concentration. ^{228}Ra activity concentrations can be determined retrospectively in samples prepared for analysis for ^{226}Ra.

Prior to the 2006–07 wet season, weekly samples obtained from Magela Creek were combined to provide monthly averages. Since 2007, the weekly samples collected from Magela Creek have been increased in size from 1 L to 5 L. This was done to improve the detection limit and to enable measurement of ^{228}Ra on combined weekly samples to give a monthly average. Initial results from ^{228}Ra determination in Magela Creek samples collected during March to May 2007, including ^{226}Ra/^{228}Ra activity concentration ratios, are presented here. Further analysis is being undertaken on remaining samples.

Results

The ^{226}Ra activity concentration data in Magela Creek for the 2009–10 wet season are compared with the previous wet seasons in Figure 1. In addition the wet season median values for each location and the wet season median differences between locations are reported in Table 1. Each wet season, the difference value is calculated by subtracting the upstream median from the downstream median (Sauerland et al 2005). This difference is called the wet season median difference (shown by the solid black lines in Figure 1) and should not be more than the limit of 10 mBq/L. The data for the nine sampling seasons indicate that ^{226}Ra levels in Magela Creek are due to the natural occurrence of radium in the environment (upstream dataset) and that ^{226}Ra activity concentrations in Magela Creek water are not elevated (‘*All years*’ column, Table 1) downstream of Ranger uranium mine.

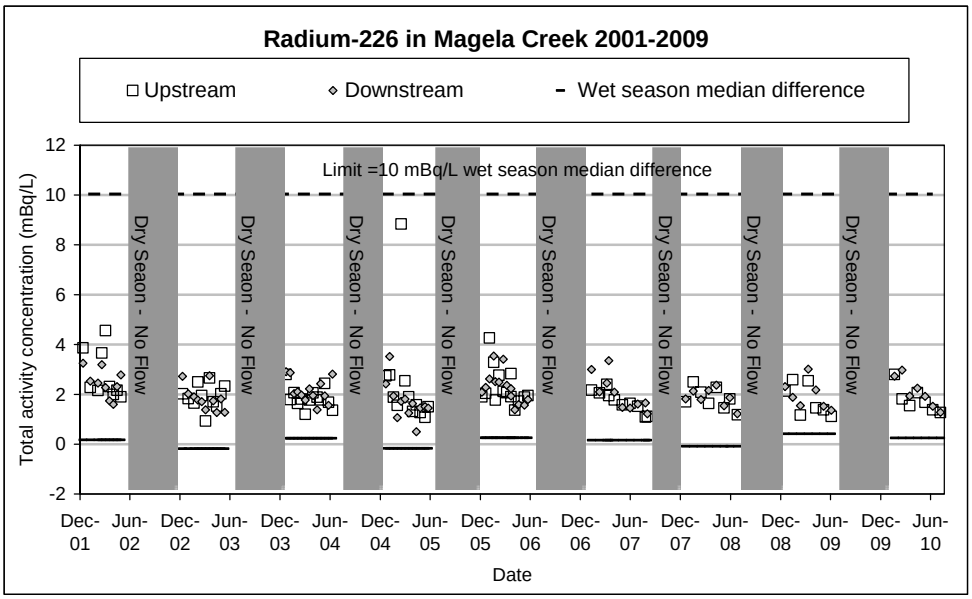

Figure 1 Total Radium-226 in Magela Creek for the 2001–2010 wet seasons

Table 1 Median and standard deviations of the ^{226}Ra activity concentration in Magela Creek for individual wet seasons (2001–10)

Wet season	Median and standard deviation		Median difference
	Upstream	Downstream	
2001–2002	1.9 (± 1.0)	1.6 (± 0.6)	-0.3
2002–2003	2.0 (± 0.5)	1.8 (± 0.5)	-0.2
2003–2004	1.8 (± 0.4)	2.0 (± 0.5)	0.2
2004–2005	1.7 (± 2.1)	1.6 (± 0.7)	-0.2
2005–2006	2.0 (± 0.8)	2.3 (± 0.7)	0.3
2006–2007	1.7 (± 0.4)	1.9 (± 0.7)	0.2
2007–2008	2.0 (± 0.5)	1.9 (± 0.4)	-0.1
2008–2009	1.5 (± 0.6)	1.9 (± 0.6)	0.4
2009–2010	1.7 (± 0.5)	1.9 (± 0.6)	0.3
All years	1.9 (± 1.0)	1.9 (± 0.6)	0.1

A limit of 10 mBq/L increase above natural (upstream) background in total ^{226}Ra concentration in surface waters downstream of Ranger has been defined for the purpose of human radiological protection (Klessa 2001), This value was inferred from the potential dose received from the ingestion of ^{226}Ra in the freshwater mussel *Velesunio angasi* (Martin et al 1998), taking into account the uptake factor for Ra from the water column.

Data for Magela Creek show that not only are the levels of ^{226}Ra very low, both upstream as well as downstream of the Ranger mine, but there is also no statistically significant difference between average ^{226}Ra activity concentrations at the upstream and downstream sites in the 2009-10 wet season (two sample t-test; p = 0.36). In addition, ANOVA (using a general linear model) was performed on the measured upstream-downstream differences in ^{226}Ra activity concentrations between the 2009-2010 wet season and previous wet seasons. There is no statistical difference between the individual wet season (p = 0.17) and between this wet season and the previous years (p = 0.46), respectively.

^{228}Ra and ^{226}Ra:^{228}Ra activity concentration ratios in Magela Creek – first results

^{228}Ra analyses were started in 2009–10 using alpha spectrometric measurement after allowing a suitable time for ingrowth of the ^{228}Th daughter in the samples prepared for ^{226}Ra analyses (Medley 2010). The dataset is still incomplete, but some preliminary results are shown here.

Table 2 Preliminary results of ^{228}Ra activity concentration and ^{226}Ra/^{228}Ra activity concentration ratios for composite samples collected in March and April 2007. Data for both the dissolved and particulate fractions are shown.

Site	Collection dates	^{228}Ra (mBq/L)	^{226}Ra: ^{228}Ra
Magela Creek Upstream (Filtrate)	22/03/2007–04/04/2007	0.63 ± 0.10	1.6 ± 0.2
Field duplicate		0.51 ± 0.06	1.7 ± 0.1
Magela Creek Downstream (Filtrate)	22/03/2007–04/04/2007	0.49 ± 0.01	2.0 ± 0.1
Field duplicate		0.61 ± 0.08	1.7 ± 0.1
Magela Creek Upstream (Particulate)	12/04/2007–03/05/2007	0.37 ± 0.05	1.9 ± 0.1
Magela Creek Downstream (Particulate)	12/04/2007–03/05/2007	0.56 ± 0.04	1.8 ± 0.1

Associated uncertainties given are one standard deviation based on counting statistics only.

The data presented for filtered ^{228}Ra (Table 2) consistently show a lower activity concentration than for ^{226}Ra. The difference between the field duplicate samples is higher than that between the upstream and downstream sites. Although the ^{226}Ra/^{228}Ra activity ratios for the filtrate samples appear higher downstream, this difference is not statistically significant (p = 0.43). There is also no difference in ^{226}Ra/^{228}Ra activity ratios measured in suspended sediment (particulate phase) upstream and downstream of Ranger for the two particulate samples that were analysed.

Further data are required to complete an investigation of the usefulness of ^{226}Ra/^{228}Ra activity concentration ratio measurements to pinpoint sources of radium in Magela Creek water. Sufficient samples were collected to produce a baseline dataset for ^{226}Ra/^{228}Ra activity concentration ratios in Magela Creek over the past two wet seasons. However, the relatively high uncertainty associated with the measurement of the very low levels of ^{228}Ra may preclude the ratio method from being able to be used to reliably detect any conditions other than a major downstream excursion.

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Results from the routine stream monitoring program in Magela Creek catchment, 2009–10

Introduction

C Humphrey, A Bollhöfer & D Jones

Progress under this KKN for the stream monitoring program in the Magela Creek catchment is reported by way of (i) results of the routine monitoring program conducted for the 2009–10 period, and (ii) monitoring support tasks for the same period, including research and development, reviews and reporting. The latter tasks are reported separately in 'Ranger stream monitoring: Research and development', pp 77–81, this volume.

Since 2001, routine water quality monitoring and ecotoxicity programs have been deployed by SSD for environmental assessment of aquatic ecosystems in the ARR. The objective of this work has been to provide independent assurance that the aquatic environment remains protected from current and past mining-associated activities in the region. The monitoring program incorporates chemical, physical and biological components.

The techniques and 'indicators' used in the monitoring program satisfy two important needs for environmental protection: (i) the early detection of potential significant effects to avoid ecologically important impacts; and (ii) information on the ecological importance of any likely impact (biodiversity assessment). Monitoring techniques adopted by SSD that meet these requirements are:

(i) Early detection of short or longer-term changes

- *Water physico-chemistry*:
  - Grab samples for water quality measurement: includes pH, electrical conductivity (EC), suspended solids, uranium, magnesium, calcium, manganese and sulfate (weekly sampling during the wet season) and radium (samples collected weekly but combined to make monthly composites),
  - Continuous monitoring: use of multi-probe loggers for continuous measurement of pH, EC, turbidity and temperature in Magela Creek, and EC and turbidity in Gulungul Creek;
- *Toxicity monitoring* of reproduction in freshwater snails (four-day tests conducted in situ, at fortnightly intervals);
- *Bioaccumulation* – concentrations of chemicals (including radionuclides) in the tissues of freshwater mussels in Mudginberri Billabong to detect far-field effects including those arising from any potential accumulation of mine-derived contaminants in sediments (mussels sampled every late-dry season).

(ii) Assessment of changes in biodiversity

- *Benthic macroinvertebrate communities* at stream sites (sampled at end of each wet season);
- *Fish communities in billabongs* (sampled at the end of each wet season).

In accordance with the concepts of best practice and optimisation, the routine monitoring program has evolved through time as technologies (eg continuous physicochemical monitoring using datasondes and telemetry) have evolved, and improved methodologies for biological assessment (eg in situ monitoring using snails) and statistical analysis of data have been developed under the SSD research program.

The results from the stream chemical and biological monitoring program for 2009–10 are summarised below. ^{226}Radium activity concentrations in Magela Creek for 2009–10 are reported separately (see 'Surface water radiological monitoring in the vicinity of Ranger and Jabiluka, pp 42–45, in this volume).

Water chemistry monitoring program

A Frostick, K Turner, K Tayler & D Jones

Background

Continuous monitoring of surface waters around Ranger mine is conducted by both SSD (at Magela Creek upstream and downstream sites, MCUGT and MCDW, respectively) and ERA (at RP1W, GC2, Georgetown Billabong and 8 locations along Magela Creek). These data are used for the assessment of potential impacts arising from activities carried out on the minesite (Supervising Scientist 2007, 2008, Turner et al 2008a,b, Turner 2009, Turner & Jones 2009).

A critical attribute of SSD's continuous monitoring network is the ability to remotely monitor in real time (via 3G telemetry) events in the creek system. Telemetry provides a means for early warning of increases in inputs of sediment (turbidity) or solutes (electrical conductivity) from the minesite. The continuous monitoring data are also used to quantify annual loads of solutes and sediment, with the aim of tracking overall performance of the mine's water management system from year to year (Turner & Jones 2009). By comparing the total mass of solutes measured downstream of the mine in Magela Creek with the mass of solutes from point and diffuse sources on the minesite and loads upstream of the mine in Magela Creek, a dynamic assessment of the intra- and inter-seasonal fluxes of salts in the system can be made.

Solute loads were not calculated for the 2009–10 wet season as ERA advised that:

a) the rating table for the RP1 weir needed to be corrected following physical changes to the weir made after the 2008 flood;

b) estimates of discharge from GC2 were inaccurate owing to flow bypassing the control structure.

ERA has committed to deriving a new rating table for the RP1 weir, and the construction of an improved weir at GC2 in the Corridor Creek system to enable a more accurate determination of discharge via Corridor Creek. Once these improvements have been implemented a more accurate calculation of past and current solute loads leaving the minesite will be possible. This will involve recalculating the previously derived and reported (Supervising Scientist 2009, Turner & Jones 2009, 2010) solute loads from the 2005/06 to the 2008/09 wet seasons.

Monitoring program for 2009–10 wet season

The methods used for continuous monitoring have been reported previously in the 2008–2009 Supervising Scientist Annual Report, Section 3.1, and included:

- Monitoring stations recording turbidity, electrical conductivity and stage height located in Magela and Gulungul Creeks above and below Ranger Mine, in mid-Gulungul Creek and at downstream Ngarradj Creek below the Jabiluka project area.
- Automatic samplers located at the downstream Magela and Gulungul sites programmed to collect:
  - a weekly sample at the upstream and downstream stations on the same day as the routine weekly grab sampling program; and

  - event-based samples at the upstream and downstream stations according to pre-set criteria that statistically define significant changes in stream turbidity and EC.

- Weekly grab samples collected alongside the continuous monitoring stations, allowing direct comparison between grab and continuous data.

Results

The flows in the minesite tributaries and in Magela Creek depend on rainfall occurring both in the upper Magela catchment and in the vicinity of the minesite. Annual total rainfall measured at Jabiru airport (by the Bureau of Meteorology) and cumulative annual discharge volumes for Magela Creek (as measured at GS210009, adjacent to the 009C compliance site) since 2006 are shown in Table 1. These data show the variability in annual rainfall and resultant discharge.

Table 1 Jabiru rainfall and Magela creek wet season flow conditions since 2006

Wet Season	Annual cumulative rainfall (mm)	Annual cumulative discharge (GL)
2006–07	2540	845.2
2007–08	1673	416.6
2008–09	1186	235.2
2009–10	1596	369.6

Magela Creek

As with the 2008–09 wet season there was close integration of the routine water chemistry weekly grab sampling monitoring program with continuous water quality monitoring and in situ toxicity monitoring programs. The weekly grab samples, as for previous seasons, were measured for key minesite signature analytes, including physicochemical parameters.

Flow was first recorded for the 2009–10 wet season on 24 December 2009 at the Magela Creek upstream monitoring station. At the downstream monitoring station flow started on 27 December 2009.

The first water chemistry grab samples for the Supervising Scientist's 2009–10 wet season surface water monitoring program were collected from Magela Creek on 30 December 2009. Weekly sampling continued throughout the wet season until cease to flow was agreed on 27 July 2010. The continuous monitoring of EC and turbidity was maintained at both the downstream and upstream sites throughout the wet season.

Rainfall in the Magela Creek catchment in late December 2009 resulted in increased flow, with consequent decreased manganese concentration, electrical conductivity and pH, and increased turbidity at both the upstream and downstream sites. This behavior is typical of first flush conditions.

The series of minor electrical conductivity events (Figure 1) seen in late January is likely to be associated with the release of mine-derived solutes from Retention Pond 1 (RP1) to Coonjimba Billabong. These EC events lasted between 9 and 13 hours. During two of these events the EC remained above the EC guideline value of 43 µS/cm for periods of 2.25 and 0.83 hours.

Water levels within Magela Creek remained relatively low during mid-February. High rainfall in late-February resulted in high creek levels from 26 February – 3 March 2010. Below

average rainfall during March resulted in very low creek levels and increased values for electrical conductivity and pH, and higher magnesium and sulfate concentrations.

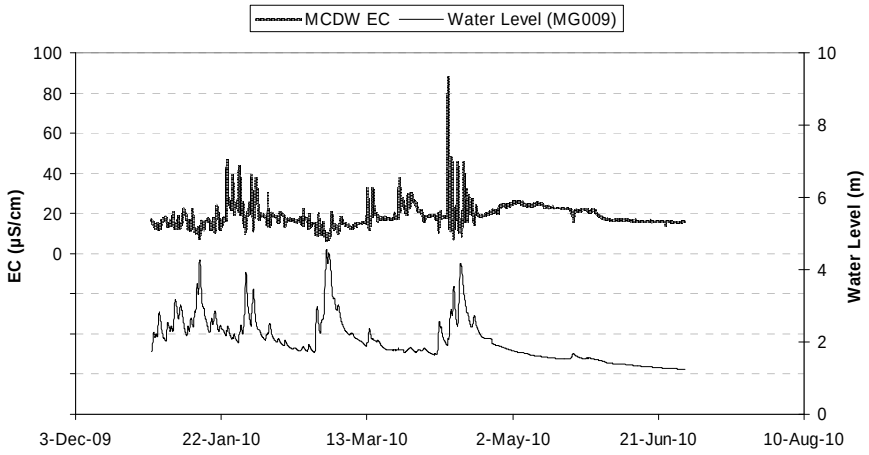

Figure 1 Electrical conductivity and discharge measurements in Magela Creek between December 2009 and July 2010 – continuous monitoring data

Heavy rainfall during mid-April resulted in seasonally low solute concentrations and increased turbidity due to high water flows. The continuous monitoring data (Figure 1) show several EC events during this period of high creek levels. These events coincided with increased discharge of water from RP1, with values of EC exceeding the EC guideline of 43 µS/cm for durations ranging from 2.75 to 8.5 hours, with maximum EC values ranging from 48 to 90 µS/cm.

These pulses of higher EC water are likely to have originated from RP1 (via Coonjimba Billabong). It is probable that an increase in water level in Magela Creek had initially restricted flow from Coonjimba Billabong, by virtue of a hydraulic damming effect. When flow in Magela Creek declined, the hydraulic head dissipated and the water held back in Coonjimba Billabong flowed out.

Overall, the data from the continuous monitoring and grab sample monitoring programs indicate that water quality in Magela Creek was comparable with previous seasons for the west channel (Figure 2).

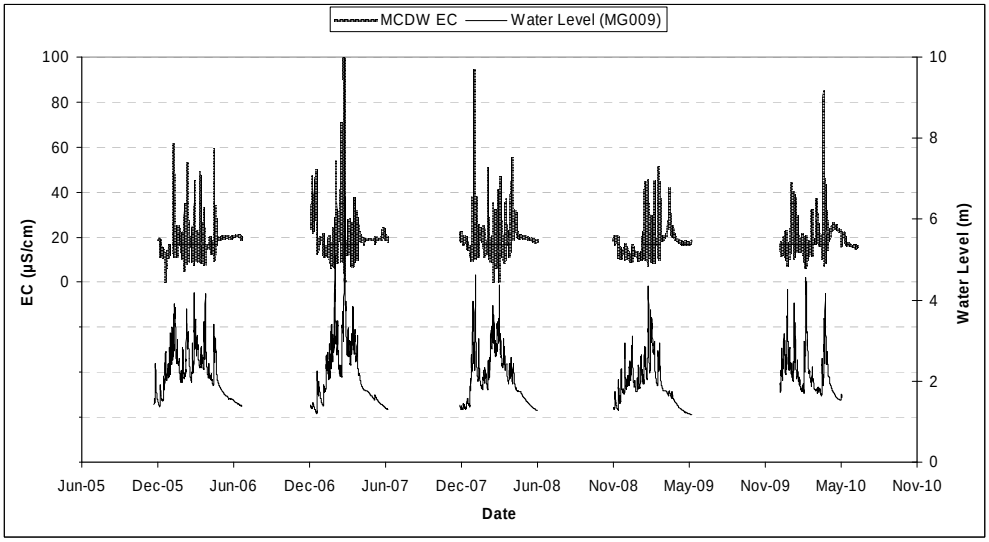

Figure 2 Electrical conductivity measurements and water level (lower trace) in Magela Creek (SSD data) between December 2005 and July 2010 – continuous monitoring data

Figure 3 shows that uranium concentrations measured during the 2009–2010 wet season were comparable with previous seasons for the downstream west channel of Magela Creek and remained well below the statutory limit of 6 µg/L.

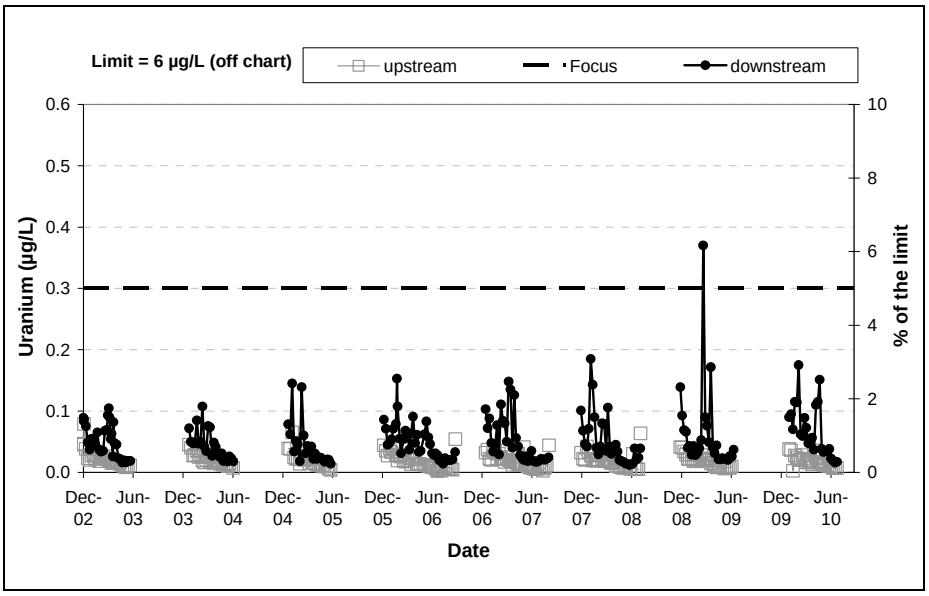

Figure 3 Uranium concentrations in Magela Creek since the 2002–03 wet season – grab sample data

Gulungul Creek

Routine weekly grab sampling for analysis of water chemistry variables was discontinued at the upstream site from the start of the 2008–09 wet season, as this site does not represent a useful reference site (ie water chemistry measured at this site may show upstream (natural) catchment influences that compromise its effectiveness for assessing downstream impacts from the mine). However, during the 2009–10 wet season grab samples were taken at the upstream site during the period of trial deployment of the in situ snail toxicity tests.

Weekly grab sample monitoring was continued at the downstream site. The continuous monitoring of EC and turbidity has been maintained at both the downstream and upstream sites.

The first water chemistry samples for the 2009–10 wet season were collected from Gulungul Creek on 30 December 2009. Weekly sampling from the downstream site continued throughout the season until 24 June when MTC stakeholders agreed that surface flow had ceased in Gulungul Creek.

All weekly grab samples had electrical conductivity measurements (EC) below the Magela Creek guideline value of 43 µS/cm. However, continuous monitoring (Figure 4) showed two exceedances of this guideline value during the peak of EC events occurring on 26 January and 24 March 2010. These events lasted for 14 and 21.5 hours respectively, during which time the EC remained above the guideline value for 3 hours during the January EC event and 1.25 hours during the March event. Uranium concentrations in grab samples were less than 10% of the Magela Creek limit (Figure 5).

Overall, the water quality measured in Gulungul Creek during the 2009–10 wet season indicates that the aquatic environment in the creek has remained protected from mining activities.

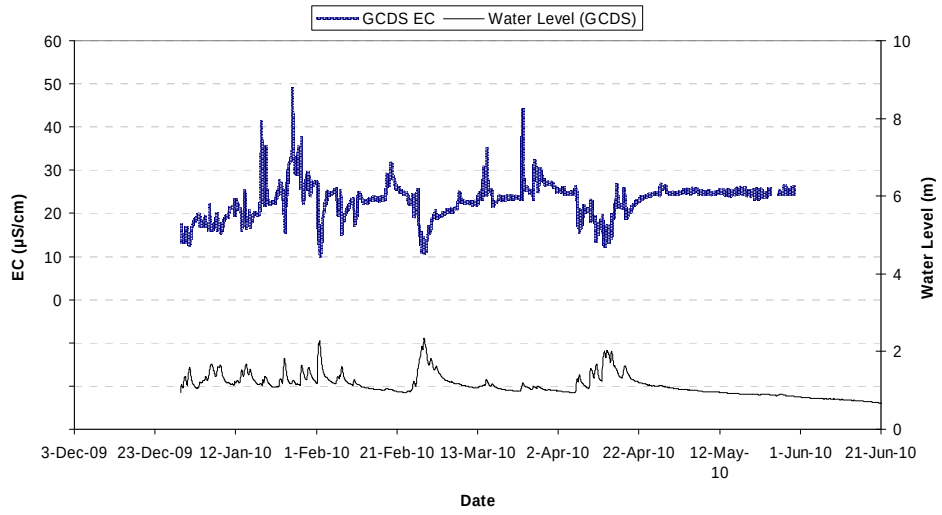

Figure4 Electrical conductivity measurements in Gulungul Creek between December 2009 and June 2010 – continuous monitoring data

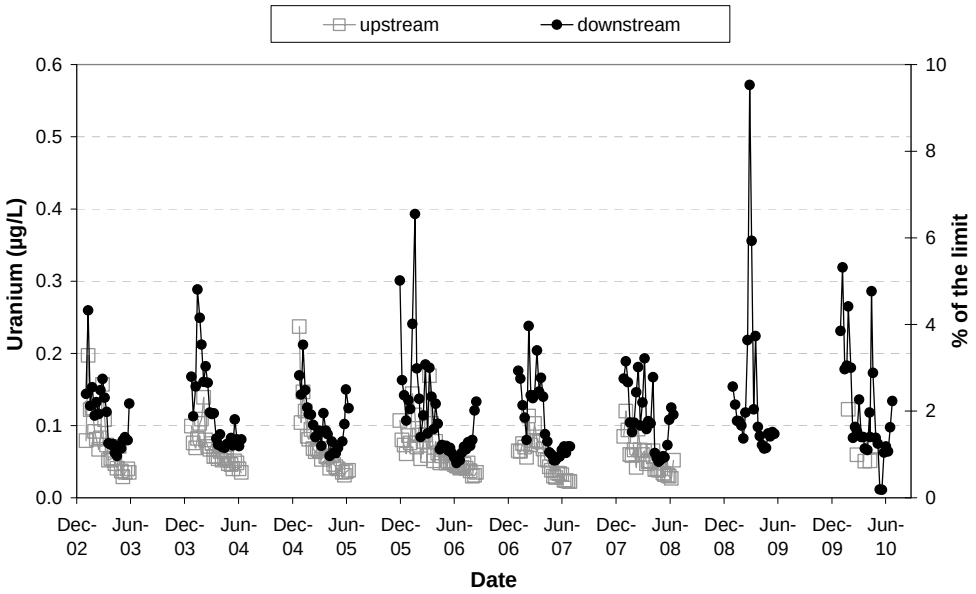

Figure 5 Uranium concentrations measured in Gulungul Creek by SSD between December 2002 and June 2010 – grab sample measurements.

Water chemistry monitoring program for 2010–11

A substantial review of the water chemistry monitoring program was undertaken over the 2010 dry season, to identify what was needed to implement continuous monitoring as SSD's primary water quality monitoring tool for the 2010–11 wet season. Posting of continuous water quality data on the internet (weekly in arrears) will be initiated and event triggered auto-sample collection will become the primary collection method (as opposed to weekly grab sample collection) for water samples to be chemically analysed. These changes have required significant upgrades to existing infrastructure, review of relevant procedures and manuals and increased staff resourcing. Each monitoring location will have a dedicated site manager with responsibility for all aspects of infrastructure and instrument maintenance and data quality control.

Monitoring infrastructure

Improvements to monitoring infrastructure completed over the 2010 dry season include:

- Duplicate multi-probe sondes at upstream and downstream Gulungul Creek;
- New custom built monitoring pontoon for Magela Creek downstream site (Figure 6);
- Duplicate Gamet auto-samplers fitted to Magela Creek downstream pontoon;
- Various improvements to Gulungul Creek monitoring stations to facilitate calibration and replacement of sondes under high-flow conditions;
- Improved data download and telemetry system including revised programming for sample triggers and alarms.

Figure 6 Magela downstream monitoring pontoon

Continuous data

Continuous data will be downloaded from the monitoring stations to a Hydstra database on a daily basis, or more frequently upon request. Any events exceeding pre-set triggers will result in a notification via mobile phone to key staff members allowing a rapid response to changes in water chemistry.

Validated continuous EC, turbidity and flow data for Magela Creek (upstream and downstream), Gulungul Creek (upstream and downstream) and Ngarradj (downstream) will be posted to the web weekly in arrears.

QA/QC and radium

Each site manager will visit their sites fortnightly to conduct a QA/QC check for measurement of physicochemical parameters and to collect a water sample for analysis of the standard suite of metals. At this time an additional sample will be collected at the upstream and downstream Magela Creek sites for the analysis of Ra. The fortnightly radium samples will be filtered, then combined to give a monthly composite from each site in line with current SSD procedures.

Automated sample collection and chemical analysis

The downstream monitoring sites in both Magela and Gulungul Creeks have been equipped with Gamet auto-samplers. These samplers are programmed to collect a 1 litre water sample based upon pre-specified EC and/or turbidity triggers and will notify staff of each sample collected via the mobile phone network.

The downstream Magela Creek site has duplicate samplers – one of which is trigged by conductivity, the other by turbidity.

From the 2010–11 wet season onwards, all water samples will be analysed for *total* metal concentrations (dissolved metals plus those weakly bound to suspended particulate matter). This contrasts with the previous weekly grab sample program where all samples were filtered in the field immediately after collection, and analysed for filterable (dissolved) metals only.

The event-based samples will be subject to a variable period of standing (mostly less than 24 hours but dependent upon environmental conditions) before they are able to be retrieved from the auto-sampler and acidified in the laboratory. By analysing the total metal concentration (as distinct from dissolved metal concentration), the proportion of dissolved metals that would typically become 'lost' due to adsorption to the surface of particulate matter during the standing period, will be accounted for.

Over two wet seasons (2008–09 and 2009–10), the SSD analysed event-based water samples collected over a range of EC and turbidity values for both the total (dissolved metals as well as those weakly bound to suspended particulate matter) and filterable (dissolved metals only) metal concentrations. The data are compared in Figure 7.

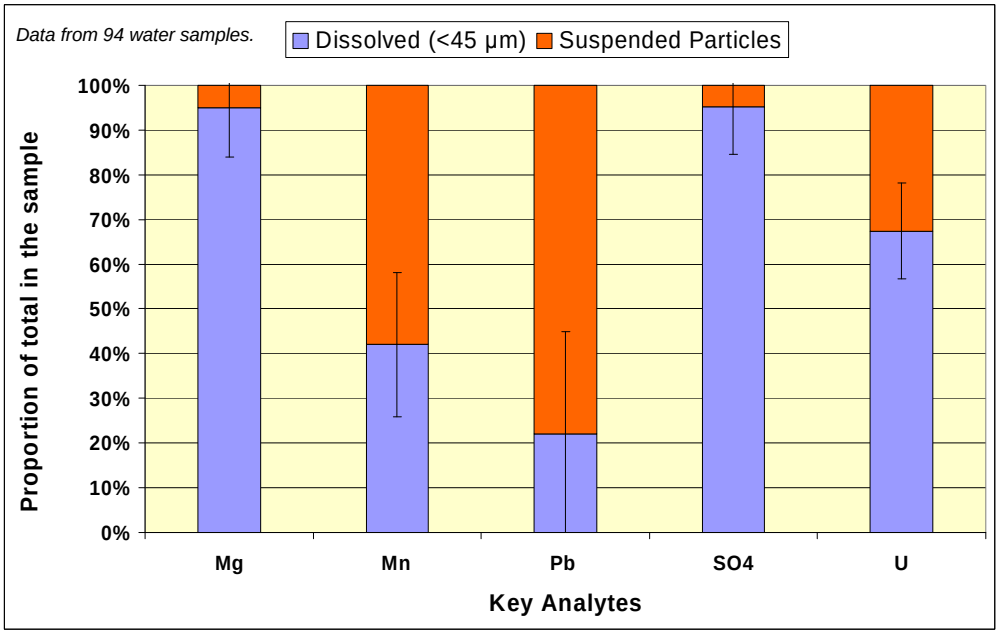

Figure 7 Proportions of the total analyte concentration associated with wither the dissolved (filterable through a 0.45 µm filter) or particulate fractions. Error bars represent the standard deviation.

The results in Figure 7 (summarised in Table 2 below) show that concentrations of magnesium (Mg) and sulfate (SO_4) are dominantly associated with the dissolved fraction. This is consistent with their chemically non-reactive nature. For uranium, approximately 30% is associated with the particulate fraction and approximately 70% with the dissolved fraction. The behaviour of manganese (Mn) was highly variable.

Table 2 Percentage of the total metal concentration in a sample that is expected to be associated with either the dissolved or particulate fractions

	Mg	Mn	SO_4	U
Dissolved (%)	95 (± 11)	42 (± 30)	95 (± 11)	67 (± 14)
Particulate (%)	5 (± 11)	58 (± 30)	5 (± 11)	33 (± 14)

Data from 94 water samples

It is clear that the majority of Mg, SO_4 and U in samples collected from the Magela Creek downstream site is present in the dissolved fraction. The variation (standard deviation) in the proportion of dissolved and particulate concentrations is shown in brackets in Table 2. Using the information in Table 2, the relative proportions (dissolved or particulate) of the total concentration likely to be present in samples measured during the 2010–11 wet season can be estimated. This will enable a comparison to be made with the dissolved data measured in previous season.

The use of totals for future monitoring of water quality will provide a more conservative (ie more protective) assessment of the concentrations of metals present, noting that the guideline values used for compliance assessment in Magela and Ngarradj Creeks are based on the dissolved (more bioavailable) concentrations present.

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Toxicity monitoring in Magela Creek

C Humphrey, C Davies & D Buckle

Background

Biological toxicity monitoring evaluates effects of waters dispersed from the Ranger minesite on receiving waters using responses of aquatic animals exposed in situ to creek waters. The response measured in Magela Creek is reproduction (egg production) by the freshwater snail *Amerianna cumingi*. This species has been shown to be among the most sensitive, to both uranium and magnesium, of SSD's suite of six local species as determined using standardised laboratory toxicity test protocols.

For the 1990–91 to 2007–08 wet seasons, toxicity monitoring was carried out using the 'creekside' methodology. This involved pumping a continuous flow of water from the adjacent Magela Creek through tanks containing test animals located under a shelter on the creek bank. This method was replaced in the 2008–09 wet season by an in situ testing method. The in situ testing was implemented following a rigorous three year period of method development involving side-by-side comparison of creekside and in situ testing to ensure that both methods produced similar results (see Humphrey et al 2009a for rationale and results from this three year trial).

Methods

Nine in situ toxicity tests were conducted at a fortnightly frequency (ie every other week) over the 2009–10 wet season. The first started on 4 January 2010 and the final test started on 3 May 2010. Each test ran over a four-day exposure period. More detail on the methods can be found in Humphrey et al (2009a).

Results for the 2009–10 wet season

Results are plotted in Figures 1a and b with egg production at upstream and downstream sites, and differences in egg production between the sites being displayed. On average, egg numbers at the downstream site are slightly greater than that measured at the upstream control site. Unlike previous wet seasons, snail egg production during the 2009–10 season was *consistently* higher (8 out of 9 tests; Figure 1b) at the downstream site compared with the upstream site. The positive difference was particularly marked in the 3rd test and to a lesser extent in the 4th and 5th tests.

Analysis Of Variance (ANOVA) testing was used to test for differences in the upstream-downstream difference values between test results for the 2009–10 wet season and all previous wet season data (see ANOVA details, Humphrey et al 2009b). For the first time, a significant difference was found between the data for the most recent year and that from previous wet seasons ($p = 0.046$), confirming the generally higher downstream egg production in 2009–10 (Figure 1b). A number of factors have the potential to cause the different behaviour observed for the 2009–10 wet season: methodological or systematic operator problems during the wet season; an unusual suppression in egg number upstream over the wet season; or enhancement of egg number downstream that may be associated with inputs of

water (as measured by EC or turbidity data) from the Ranger site. Each of these potential causative factors was assessed in detail using the extensive available historical grab sampling and continuous water quality monitoring datasets acquired by SSD's stream water chemistry monitoring program.

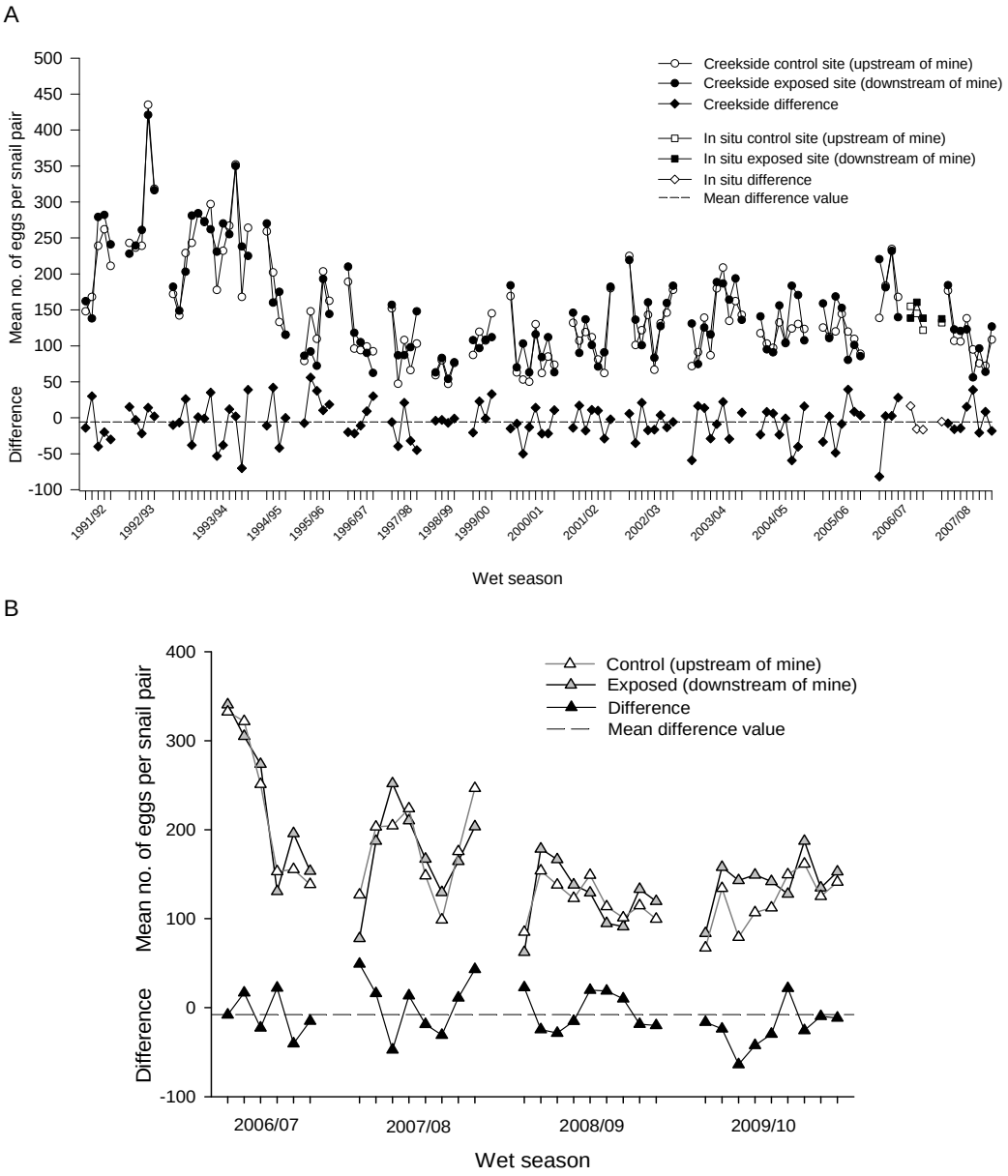

Figure 1 Time-series of snail egg production data from toxicity monitoring tests conducted in Magela Creek using A: (mostly) creekside tests, and B: in situ tests

Assessment of impact

A. Operator/methodological error

An audit of protocol procedures used in the 2009–10 wet season was conducted. The operator conducting and supervising the tests has been the same staff member for the past four wet seasons. While refinements (efficiencies) have occurred in the protocol over this period, mainly as a consequence of moving from creekside to in situ testing, these are very minor and

are not regarded as sufficient to influence the results in 2009–10. Moreover, exactly the same procedures have been followed for the past two wet seasons, yet the pattern of results for the 2009–10 wet season differs from 2008–09 test results (see Figure 1b).

B. Possible anomalies observed at the upstream control site

While the egg number value from the 3rd test conducted in the 2009–10 wet season appears to be unusually low compared with the corresponding downstream value, the same pattern in reproductive response at this site was also observed for the subsequent two tests (4th and 5th tests) (Figure 1b). For the upstream data from the 3rd, 4th and 5th tests, precision among the replicates was similar to that observed at all other times, data from the two independent duplicate containers (each holding 8 pairs of snails) were similar, no outliers were evident in the data, and adult snail mortality over the four day exposure period was well within acceptance limits (data not provided in this report). Based on statistical criteria, there was no reason to consider any of the 2009–10 test data from the upstream site anomalous and 'outlying'.

C. Possible mine-related changes to water quality associated with U or $MgSO_4$

Egg numbers at the downstream 'exposed' site are usually slightly higher than that measured at the upstream control site (Figure 1). Most likely, higher downstream egg production may be attributed to the inputs to Magela Creek of billabong-tributary waters (Georgetown and Coonjimba) at two locations. These tributary ponded-waters have higher temperatures, a higher organic carbon content than main-stem creek waters and, for Coonjimba in particular, elevated concentrations of mine solutes (including $MgSO_4$ and Ca) compared to, low solute Magela creek waters). Higher water temperatures enhance reproductive activity in *Amerianna cumingi* (Jones 1992). Further, both the dissolved salts (providing they are not too high in concentration), increased nutrients and natural organic matter would supplement the food supply and in turn, could enhance egg production, of downstream snails.

Water quality differences between the two Magela monitoring sites are also highly affected by creek hydrology. On a falling hydrograph in Magela Creek, previously-ponded waters from billabongs located between the upstream and downstream sites flow out to the creek, accentuating solute and nutrient differences between the sites (higher concentrations are measured at the downstream site, particularly along the west bank).

While higher concentrations of U are observed in Magela Creek downstream of Ranger during the wet season (but well below the current toxicity-based U guideline value), the dominant mine-derived contaminant entering the creek at this time is $MgSO_4$. Concentrations of this contaminant are conveniently measured by the highly correlated variable, electrical conductivity (EC). The question to be posed is, can this input of $MgSO_4$ lead to enhanced snail egg production at the downstream site?

1. Comparison of field responses with laboratory sensitivities of A. cumingi

Over the concentration ranges tested in the laboratory, there is no evidence of *enhancement* of reproductive responses in *A. cumingi* to either Mg (van Dam et al 2010) or U (Hogan et al 2010), particularly at low concentrations. In the case of U, it must be acknowledged that the lower end of the concentration range tested in the laboratory is well above the concentrations of U measured in the creek. However, for Mg there is an overlap in the lower end of the concentration range tested in the laboratory and concentrations that are actually measured in the field.

The laboratory concentration-response data for Mg indicate no impairment of egg production by *A. cumingi* would be expected below about 1–2 mg/L (van Dam et al 2010) (taking into account the ameliorative effects of Ca present in mine waste waters) which is equivalent to an

EC of about 20–30 µS/cm (Supervising Scientist 2009, Figure 3.3a). The same toxicity data also indicate that no positive effects (ie increased egg production) would be expected due to increases in Mg concentration above the natural background concentrations of Magela Creek.

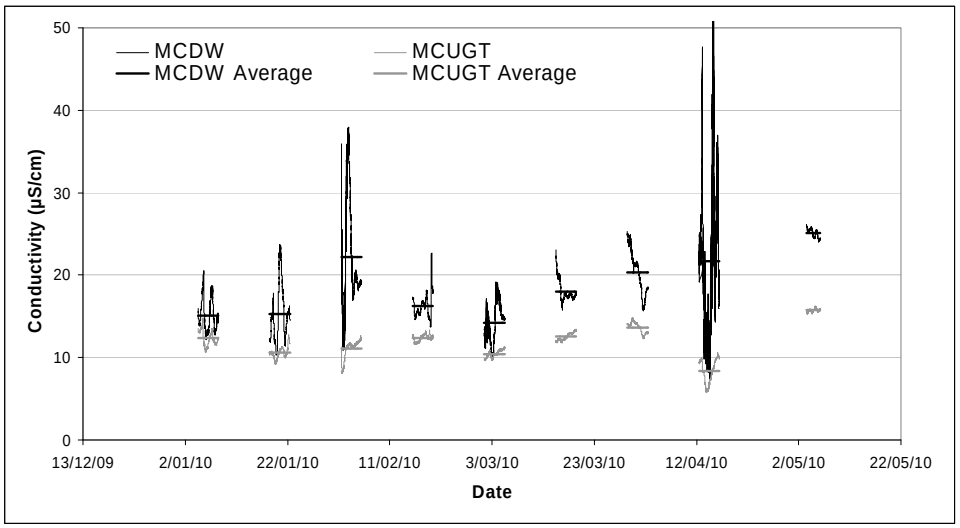

Figure 2 Plots of continuous electrical conductivity (trace) and four-day average values (horizontal line) at monitoring sites in Magela Creek for days in the 2009–10 wet season coinciding with conduct of toxicity monitoring tests

2. Possible gradient in EC observed in previous years between the east and the west side of the creek channel at the downstream site

Since the inception of toxicity monitoring in 1992, there have been duplicate containers holding snails at the upstream and downstream sites. In the period from 1991–92 to 2007–08, the duplicate containers at the downstream site either drew water from, or were located on, the east and west sides of the west channel of the creek. From 2009 onward, the duplicate containers have both been located on the west side of the channel only.

Over three wet seasons in the period 2005–06 to 2007–08, EC and other water quality variables were continuously measured at both east and west locations using datasondes. The EC traces from the sondes highlight the EC gradient between the locations. This is caused by incompletely mixed mine waters (with higher EC signature) flowing closely to the west bank of the channel (Figure 3).

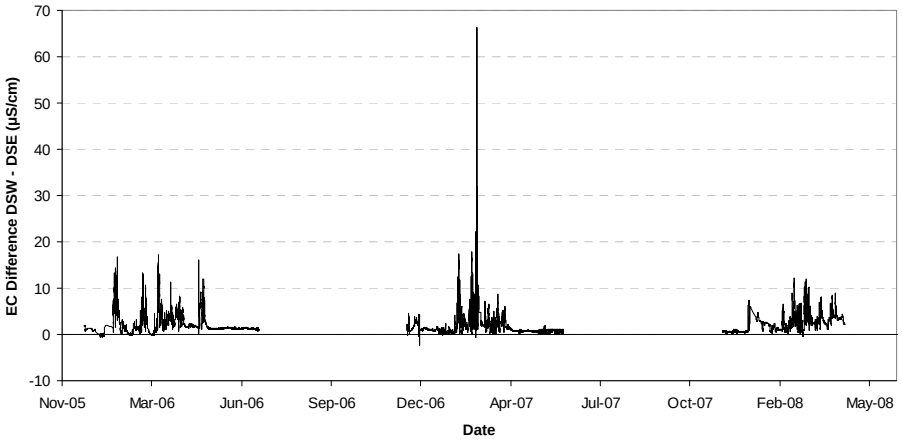

Figure 3 Difference in electrical conductivity (EC) measurements in Magela Creek between west and east sides of the channel downstream of Ranger for the 2005–06, 2006–07 and 2007–08 wet seasons – continuous (hourly) monitoring data (from SSD)

The continuous readings along the west bank are significantly higher (P<0.0001) than corresponding readings taken on the east side of the channel in each of the three years of continuous measurement.

Since 1992 to 2008, the mean egg number representing exposure of snails to both sides of the creek channel has been the same (152 west versus 152 east). For the period 2005–06 to 2007–08 when EC appears to have increased at the downstream site (Supervising Scientist 2010; Figure 2.6), the egg number means have also been similar (128 west versus 131 east, no statistically significant difference). Thus the EC gradient between west and east sides of the creek at the downstream site in this period has not been large enough to result in a statistically different response in snail egg numbers.

3. EC differences between upstream-downstream sites

Cross-correlation of snail egg production data, both mean egg number per site and upstream-downstream differences, for the nine 2009–10 wet season tests with corresponding continuous water chemistry (including EC) and stream water level data for each four-day period, was conducted using the correlation analysis tool of Excel. The population correlation calculation (also equivalent to Pearson's correlation coefficient) returns the covariance of two data sets divided by the product of their standard deviations.

The four day median value of 10 minute readings of continuous data was used in the analysis, with additional metrics (eg minima, maxima, see Table 1) applied to stream water level data measured close to the downstream site (GS8210009) (Table 1). The reason that the four-day median value was used is that this period corresponds to the same deployment time for the in situ toxicity monitoring method. The results of the analyses are shown in Table 1 (for 7 degrees of freedom (9-2 tests), an *r* value ≥0.666 is significant at P<0.05).

(a) *2009–10 wet season*. There was no correlation between any of the EC and egg production measures for the nine tests conducted in the 2009–10 wet season (Table 1; also compare Figures 1b and 2). Indeed, egg numbers actually converged (similar differences) between the sites late in the wet season when greater EC differences were observed (Figure 2).

(b) *EC values observed since 2006.* Related to *1* above, the incidences and magnitude of 'high' EC events in the 2009–10 wet season appear no greater than for the previous 4 wet seasons for which continuous monitoring data are available (Figure 4), yet egg number differences between upstream and downstream sites are not similarly high in previous recent wet seasons (Figure 1b).

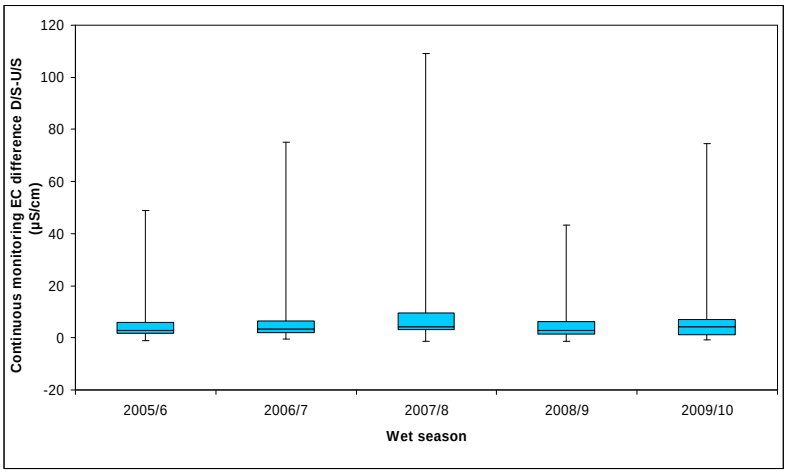

Figure 4 Box plots (median and quartiles) of Electrical conductivity (EC) differences (D/S-U/S) for continuous monitoring sites in Magela Creek

Table 1 Correlations (Pearson *r* values) amongst egg production and continuous water chemistry and stream water level statistical summaries for toxicity monitoring tests conducted in the 2009–10 wet season. Egg number data are means per snail pair from upstream (U), downstream (D) or upstream-downstream differences (Diff), while median data for EC, turbidity and temperature for corresponding sites and four-day test periods were used. Water level (WL) data summarised as maximum, minimum, median, standard deviation, and the maximum fall in water level observed over the four-day period.

	Egg-U	Egg-D	EggDiff
Egg-U	1.000		
Egg-D	0.690	1.000	
EggDiff	0.528	-0.251	1.000
EC-U	0.297	0.248	0.107
EC-D	0.451	0.415	0.116
ECDiff	-0.333	-0.324	-0.066
Turb-U	-0.365	-0.464	0.056
Turb-D	-0.636	-0.638	-0.101
TurbDiff	0.736	0.543	0.347
Temp-U	0.093	0.228	-0.143
Temp-D	0.127	0.257	-0.132
TempDiff	-0.316	-0.307	-0.062
WL-max	-0.527	-0.422	-0.210
WL-min	-0.479	-0.460	-0.101
WL-med	-0.464	-0.506	-0.027
WL-SD	-0.496	-0.387	-0.208
Max fall	-0.483	-0.263	-0.337

Taken together, the above results indicate that $MgSO_4$ is unlikely to be a significant contributor to the greater snail egg number differences observed in the 2009–10 wet season.

D. Other possible mine- or non-mine-related explanations for the 2009–2010 wet season observations

Water temperature, turbidity, organic carbon and stream flow dynamics are other factors that could potentially explain the 2010 results.

1 Water temperature

Water temperature will vary depending upon water levels, cloud cover, riparian vegetation and period of the wet season. Continuous and spot measurements have shown that while downstream water temperatures in Magela Creek are slightly higher than upstream, the differences in 2010 were very similar across all tests and comparable to differences measured over the past several wet seasons (data not provided). Further, there was no correlation between any of the water temperature and egg production measures for the nine tests conducted in the 2009–10 wet season (Table 1), indicating that water temperature is not responsible for the significantly greater downstream egg number differences in 2010.

2 Turbidity

In 2009–10, the mean downstream turbidity measured over the four-day duration of each of the nine toxicity monitoring tests was generally higher relative to corresponding mean upstream turbidity for the same periods (Figures 5).

However, and as shown for EC above, there was no correlation between any of the turbidity and egg production measures for the nine tests conducted in the 2009–10 wet season (Table 1; compare Figures 1 and 5). Upstream turbidities were 'high' during the 8th test (Figure 6) but this corresponded to close upstream-downstream concordance in egg number. Regional (Alligator Rivers Region) and Australian literature suggests that sustained turbidity (ie for at least several days) greater than 20 NTU is required to adversely affect aquatic biota in inland waters (Buckle et al 2010), values which were exceeded but for very short periods only.

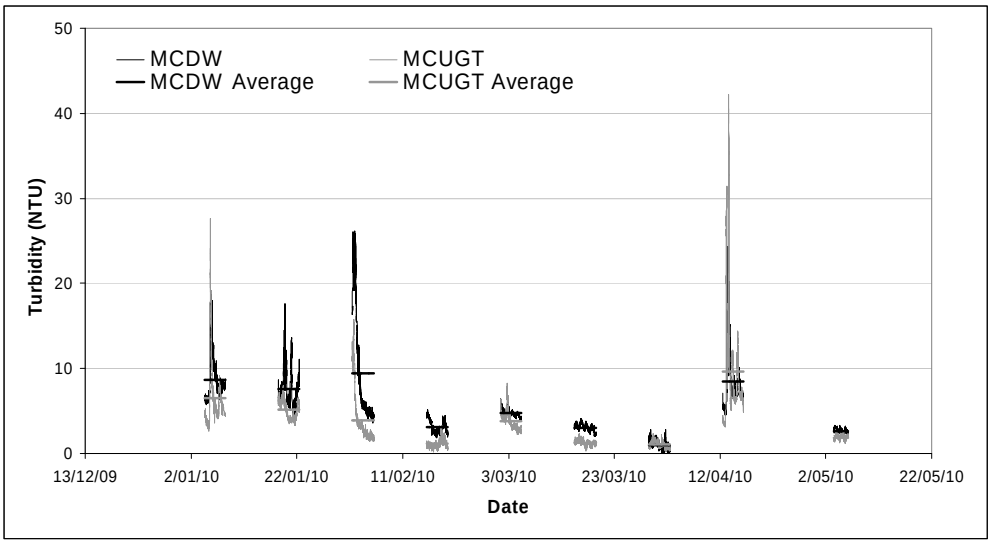

Figure 5 Continuous plots of turbidity (trace) and four-day average values (horizontal line) at monitoring sites in Magela Creek for days in the 2009-10 wet season coinciding with conduct of toxicity monitoring tests. (Note that turbidity traces for test periods 2 and 3 include some error and may be revised in future as ongoing QA/QC is applied to the associated data.)

3 Organic carbon

This variable has been measured sporadically since 1992. However, collation of the full historical data set was not able to be completed by the time this report was prepared. Systematic weekly collection of samples for the measurement of total and dissolved organic carbon (TOC/DOC) commenced with the 5th toxicity monitoring test in 2010 (Figure 6).

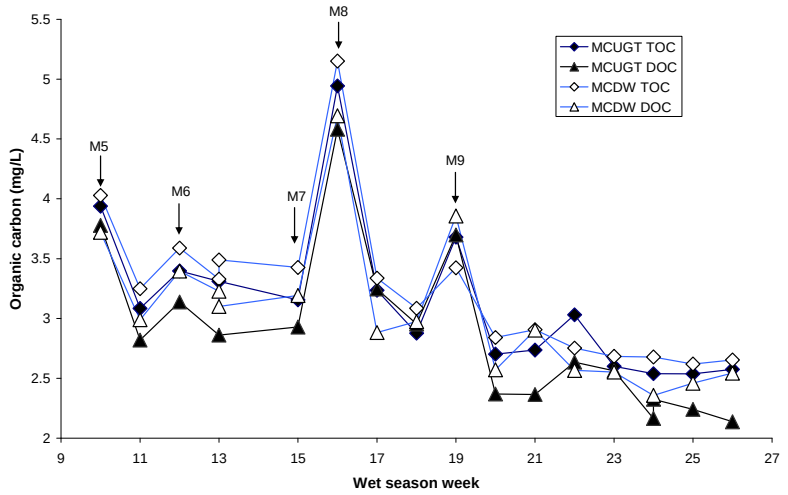

Figure 6 Plots of weekly wet season Total (TOC) and Dissolved (DOC) Organic Carbon in Magela Creek surface waters from grab samples collected from upstream (MCUGT) and downstream (MCDW) sites. Symbols M5 to M9 refer to toxicity monitoring tests 5 to 9 respectively.

Downstream values are usually higher than upstream. However, for the tests for which data were gathered in 2010, the relative differences between the sites were very small. There was no correlation between any of the total organic carbon and egg production measures for the five tests of common data conducted in the 2009–10 wet season (P>0.05).

4 Flow dynamics at the downstream site

As noted above, the greatest influence on water quality at the downstream monitoring site is water draining from billabongs (Georgetown and Coonjimba). A number of water quality variables, including solute concentrations, are enhanced in billabong waters. These inputs could have the potential to enhance snail egg production but, as discussed above, the concentrations of major ion solutes are unlikely to be the cause. The correspondence of toxicity monitoring tests conducted in 2010 with falling stage in Magela Creek was examined. Water levels measured during the toxicity monitoring tests are shown in Figure 7. The falling hydrograph during the third and eighth tests coincided with peaks in downstream EC, as water initially held back in Coonjimba Billabong flowed into Magela Creek (see Figure 2). No correlation was found between any of the water level data and egg production for the nine tests conducted in the 2009–10 wet season (Table 1; compare Figures 1B and 7).

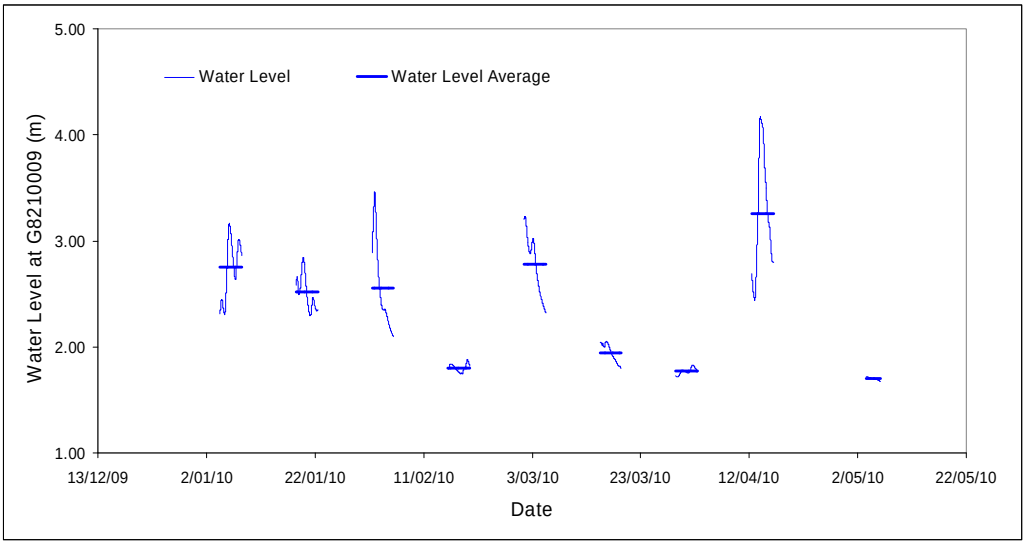

Figure 7 Plots of continuous water levels at Magela Creek downstream (MG009) for days in the 2009–10 wet season coinciding with conduct of toxicity monitoring tests

Monitoring staff have noted, in particular, the deepening of the channel at the downstream monitoring site. This deepening would result in a reduction in water velocity across the stream profile at this location and hence increase potential for settling of the suspended particulate material. Indeed, increased accumulation of organic material, a potential food source for snails, in the toxicity monitoring containers at the downstream site was noted in the 2009–10 wet season compared with previous years. However, no quantitative measurements of the amount of this settled material were made. It is possible that the increase in settled material inside the test containers at the downstream site could have been associated with, or accentuated by, an increase in erosion rates near the minesite as a result of mine exploration activities in recent years. Samples of suspended sediment obtained as part of the event-based sampling regime and collected by the routine grab sampling program are analysed for content of U and other metals. Analysis of these samples may provide additional insight into the source of the material.

Summary and further work

A summary of the conclusions from each of above lines of investigation to determine the possible causative factor(s) for enhanced downstream egg production is presented in Table 2. The main mine contaminants, U and Mg, are discounted as contributing to the 2009–10 observations. Altered flow regime at the downstream site resulting in an increase in settled organic material inside the test containers at this site is a more likely explanation.

Table 2 Multiple lines of evidence to infer the possible cause of relatively higher snail egg production at the downstream site in 2009–10 wet season

Potential causative factor	Potential contributor to enhanced downstream egg production?	
Operator/methodological error	No	• Careful audit of protocol procedures discounted operator errors
Possible anomalies observed at the upstream control site	No	• Comparable precision amongst replicates, as for previous years • Water quality at upstream site not greatly different from that observed in previous years
Uranium	No	• Concentrations measured in Magela Creek well below toxicological thresholds • Enhanced egg production responses at low U concentrations not observed in ecotoxicological studies
Magnesium	Unlikely	• Concentrations measured in Magela Creek at or above toxicological (chronic) thresholds for brief periods but enhanced egg production responses at these (low) Mg concentrations not observed in ecotoxicological studies • Significant cross-channel gradient in Mg concentration at the downstream site not reflected in similar gradient in biological response • No correlation between EC and egg production measures for the nine tests conducted in the 2009–10 wet season • Similar incidences and magnitude of 'high' EC events in previous wet seasons yet pattern of egg number differences between upstream and downstream sites unique to 2009–10 wet season
Water temperature	Unlikely	• Consistent between-site temperature differences amongst nine tests conducted in the 2009–10 wet season • No correlation between any of the water temperature and egg production measures for the nine tests
Turbidity	Unlikely	• No correlation between any of the turbidity and egg production measures for the nine tests conducted in the 2009–10 wet season
Total organic carbon	Unlikely	• No correlation between total organic carbon and egg production measures for the five tests of common data conducted in the 2009–10 wet season
Alteration to flow dynamics at the downstream site	Possible	• Deepening channel at downstream site may have contributed more settled organic material, and hence available food, for snails

There are limitations to observational and correlational approaches to drawing inference because of the potentially concurrent mine and non-mine related factors that could contribute. Laboratory studies to examine the responses of freshwater snails to a limited matrix of water quality variables, including Mg and organic carbon at low concentrations, would be one path to addressing this issue, albeit resource-intensive and with no certainty that subtle responses in this range of concentrations could be discerned within the limits of precision of the snail

test method. Initially it is proposed to implement in 2010–11 a method to quantify the amount (and carbon content) of particulate matter deposited in the in situ test containers, and to assess if there is any positive correlation between the amount (and nature) of deposited material and snail egg production. Though spot measurements of dissolved organic carbon have been made in the past, these data do not address the issue of potential food supply and the quantity available over the four-day test period.

Analysis of biological, water chemistry and creek hydrology data will continue to better determine the water quality constituents contributing to enhanced snail egg production downstream of Ranger, and the extent to which mine inputs and other mine-related alterations to water quality and hydrology of the receiving-water billabongs could be contributing.

Should enhancement in snail egg production be linked to stimulatory mine-related effects, further discussion and consideration would be required to determine whether this in fact constitutes an adverse ecological effect.

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Bioaccumulation of radionuclides in freshwater mussels

A Bollhöfer, B Ryan, C Humphrey & D Buckle

Introduction

Mudginberri Billabong is the first major permanent waterbody downstream (12 km) of the Ranger mine. Local Aboriginal people harvest aquatic food items, in particular mussels, from the billabong and hence it is essential that they are fit for human consumption. Consequently, concentrations of metals and/or radionuclides in the tissues and organs of aquatic biota attributable to inputs to Magela Creek from the mine must remain below acceptable levels. Increased concentrations of metals in tissues of aquatic organisms also provide early warning of the bioaccumulation of these constituents, potentially to harmful levels, in the creek ecosystem. Hence the bioaccumulation monitoring program serves both ecosystem and human health protection roles.

Since 2000, mussels have been collected each year and fish every two years, respectively, from Mudginberri (the potentially contaminated site, sampled from 2000 onwards) and Sandy Billabongs (the control site, sampled from 2002 onwards) (Ryan et al 2005). The radionuclide burdens in mussels from Mudginberri Billabong were found to be generally about twice as high as in mussels from Sandy Billabong. The data for fish accumulated over a period of nine years showed no issues (ie levels were extremely low) with bioaccumulation of radionuclides and metals. Hence the biannual fish sampling program has been discontinued, subject to any significant increases in the concentrations of metals and radionuclides in Magela Creek downstream of the Ranger mine.

In May 2007 a longitudinal study was conducted measuring radium loads in mussels along Magela Creek upstream and downstream of the mine to identify whether the higher radionuclide loads are related to natural or mine inputs, and to determine whether Sandy Billabong was an appropriate control site for mussels in Mudginberri Billabong (Supervising Scientist 2008). It was found that of all sites investigated along the Magela channel, Mudginberri Billabong mussels exhibit the lowest radium loads, age-for-age, and that differences in mussel radionuclide activity loads between Mudginberri and Sandy Billabong mussels are due to natural catchment rather than mine influences (Bollhöfer et al 2010a). A longitudinal study of radium uptake in mussels in Mudginberri Billabong was undertaken in 2008 and showed that the location of sampling in the billabong had no significant effect on the radium loads measured (Bollhöfer et al 2010b). The mussel radionuclide data also showed that the bioconcentration factor for radium uptake in mussels from Mudginberri Billabong has not changed significantly over the past 25 years (Bollhöfer et al 2010a).

Radium uptake is reduced by calcium in the water. Due to inputs of Ca by the weathering of Ca-containing carbonates and carbonaceous rocks along the lowland catchment section of Magela Creek, Ca concentrations gradually increase downstream, resulting in less Ra uptake in mussels from Mudginberri Billabong compared with sites further upstream. In this context it should be noted that water input into Magela Creek from the minesite contains substantial amounts of Ca, produced by weathering of carbonate rock in the waste rock dumps.

Methods

Mussels were collected from Mudginberri Billabong in October 2009 using a suction dredge and placed into acid-washed containers holding water collected from the billabong. Surface water samples for analysis were collected in acid-washed plastic containers at the time of mussel collection. Between 300 and 400 g of sediment in which the mussels were located was also collected, put into zip lock plastic bags and taken back to the laboratory for processing and analysis.

After collection, the mussels were transported to the SSD Darwin laboratories and purged for 6–7 days in billabong water before being measured individually for weight, length and width, and dissected to remove the flesh. Mussel flesh was then combined, and the sample was freeze-dried to determine the dry weight. Sediments were oven dried at 60° C for 3 days.

Both the composite mussel sample and the sediment samples were cast in epoxy resin for determination of radioisotopes of radium (^{226}Ra & ^{228}Ra), lead (^{210}Pb), and thorium (^{228}Th) by gamma spectrometry. An aliquot of the tissue sample was sent for nitric acid digestion and ICP-MS analysis of uranium, stable lead isotopes and other metals, ie Al, Ba, Ca, Cu, Fe, Mg, Mn, Rb, U and Zn.

Results

Uranium in mussels

The 2009 data for the composite mussel sample and water concentrations in Mudginberri Billabong are shown on Figure 1. Uranium concentrations in freshwater mussels, water and sediment samples collected from 2000 onwards from Mudginberri and Sandy Billabongs are included in Figure 1 to provide historical context. The concentrations of uranium measured in mussels from Mudginberri Billabong are very similar from 2000 onwards, with no evidence of an increasing trend in concentration over time.

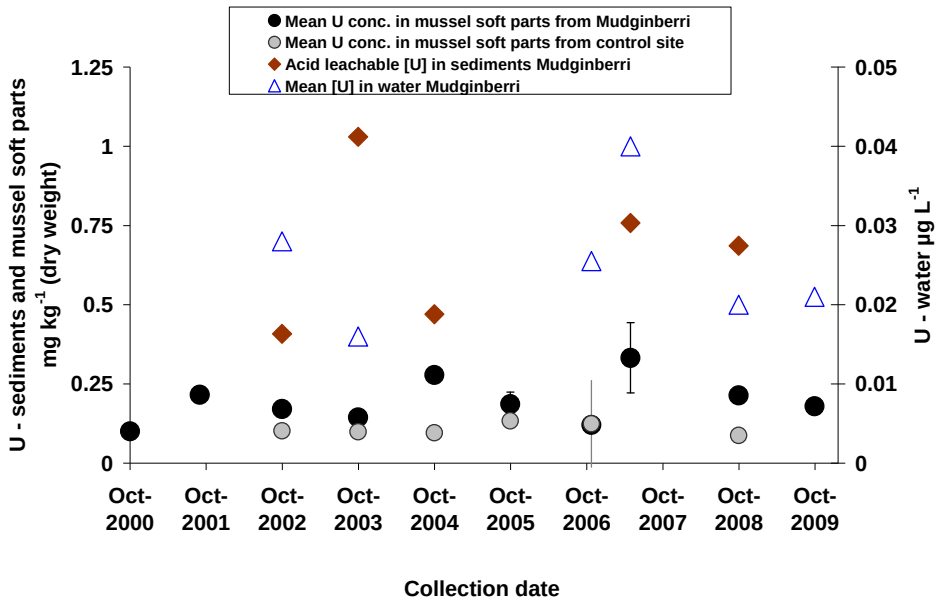

Figure 1 Mean concentrations of U measured in mussel soft-parts, 1M HCl extracts of sediment and water samples collected from Mudginberri Billabong and Sandy Billabong since 2000

The consistently low levels and lack of any increase in concentration of U in mussel tissues through time indicate absence of any mining influence.

^{226}Ra and ^{210}Pb in mussels

Activity concentrations of ^{226}Ra and ^{210}Pb in mussels are age-dependent and are also related to growth rates and seasonal soft body weights (Ryan et al 2005, Bollhöfer et al 2010a). Consequently, ^{226}Ra and ^{210}Pb activity concentrations in mussels vary depending on the timing of collection through the year.

Based upon the concentrations of ^{226}Ra and ^{210}Pb in mussel flesh, the average annual committed effective doses can be calculated for a 10-year old child (to be conservative) who eats 2 kg (wet weight) of mussel flesh from Mudginberri Billabong. Figure 2 shows the doses estimated for the mussel collections from each year, and the median, 80 and 95 percentiles for all collections. The higher committed effective dose for the 2002 and 2003 collections were caused by higher dry:wet weight ratios determined during those years. The most likely cause of these higher dry:wet weight ratios was a change in the preparation method. During shucking, or opening, of the mussels, liquid inside the mussel is usually retained and included in the wet weight of the mussels. During the 2002 and 2003 collections, the water was drained before wet weights were measured, resulting in a higher dry:wet weight ratio.

The average annual dose using all data from 2000 to 2009 is 0.175 mSv. The annual committed effective dose from the ingestion of mussels collected in 2009 is indistinguishable from previous collections, and of no concern to human health. Moreover the measured dose originates from natural catchment sources, rather than mining influences (Bollhöfer et al 2010a).

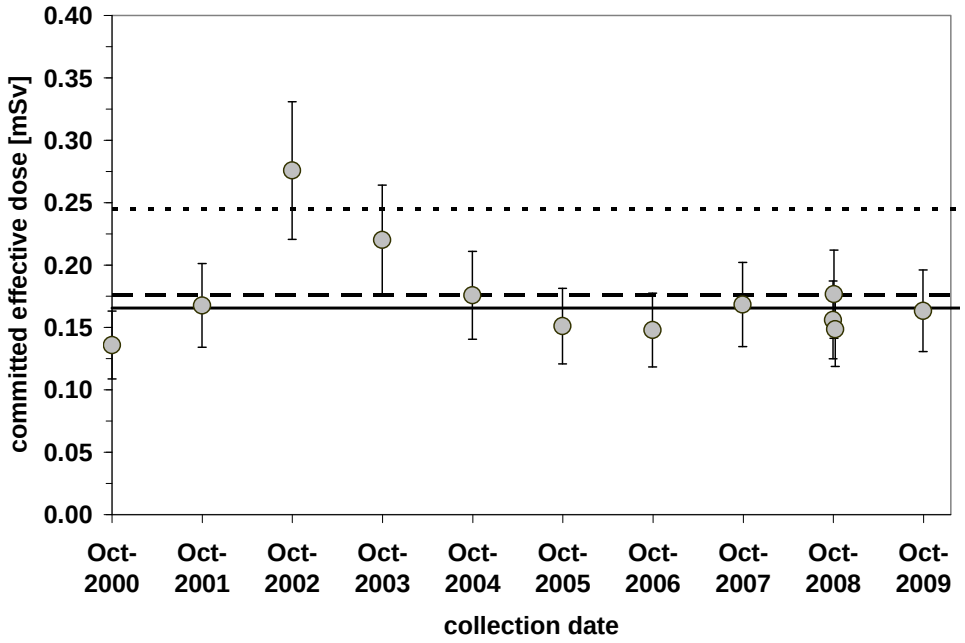

Figure 2 Annual committed effective doses from ^{226}Ra and ^{210}Pb for a 10 year old child eating 2 kg of mussels (wet) collected at Mudginberri Billabong. The median over all collections (solid line), the 80th percentile (dashed line) and 95thpercentile (dotted line) are overlaid.

Conclusion and future work

There has been no significant change in activity concentrations and committed effective doses from the ingestion of ^{226}Ra, ^{210}Pb and uranium in mussels from Mudginberri Billabong over the past decade. Calculated doses are of no concern to human health, and the source is due to natural catchment influences rather than mine-related.

Bulk annual sampling and analysis of mussels will continue. A bulk sample of mussels from Mudginberri Billabong was collected at the end of the 2010 dry season (October). This bulk sample will be analysed for radionuclides and metals, and the data compared with the historical record. The existing sampling program will be reviewed in 2011–12.

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Monitoring using macroinvertebrate community structure

C Humphrey, L Chandler & C Camilleri

Background

Macroinvertebrate communities have been sampled from a number of sites in Magela Creek at the end of significant wet season flows, each year from 1988 to the present. The design and methodology have been gradually refined over this period (changes are described in the 2003–04 Supervising Scientist Annual Report, section 2.2.3). The design is now a balanced one comprising upstream and downstream sites on each of two 'exposed' streams (Gulungul and Magela Creeks) and two control streams (Burdulba and Nourlangie Creeks).

Replicate samples are collected from each site at the end of each wet season (between April and May). For each sampling occasion and for each pair of sites for a particular stream, Bray-Curtis dissimilarity indices are calculated using PRIMER software (Clarke & Gorley 2006). These indices are a measure of the extent to which macroinvertebrate communities of the two stream sites differ from one another. A value of 'zero%' indicates macroinvertebrate communities at the two locations are identical in structure while a value of '100%' indicates totally dissimilar communities, sharing no common taxa. Disturbed sites may be associated with significantly higher dissimilarity values compared with undisturbed sites. The extent of dissimilarity in community structure is the basis for the model that is used for impact detection.

Results

Figure 1 shows the paired-site dissimilarity values using family-level (log-transformed) data, for the two 'exposed' streams and the two 'control' streams for the full macroinvertebrate dataset from 1988 to 2010

Improvements made over this period of time to the presentation and statistical analysis of macroinvertebrate data were described in Humphrey et al (2009). In particular, rather than deriving a single dissimilarity value between each of the paired (upstream-downstream) sites using pooled data (ie representing the averaged data for the five replicates collected at each site), five separate values are now calculated corresponding to each of the five possible randomly-paired upstream and downstream replicates. The within-watercourse replication allows for application of powerful analyses that can be used to test whether or not the macroinvertebrate community structure has altered significantly (compared with previous wet seasons) at the exposed sites for the recent wet season of interest. For this multi-factor ANOVA, only data gathered since 1998 have been used. (Data gathered prior to this time were based upon different and less rigorous sampling and sample processing methods, and/or absence of sampling in three of the four watercourses.)

Inferences that may be drawn from the time series data shown in Figure 1 are weakened because there are no pre-mining baseline (pre-1980) data upon which to assess whether or not significant changes have occurred as a consequence of mining. Notwithstanding, a four-factor ANOVA based upon replicate, paired-site dissimilarity values and using the factors Before/After (BA;

fixed), Control/Impact (CI; fixed), Year (nested within BA; random) and Site (nested within CI; random) shows no significant difference between the control and exposed streams from 1998 to 2010 (ie the BA x CI interaction is not significant). While the Year x Site (BA CI) interaction is significant in the same analysis ($p = 0.014$), this simply indicates that dissimilarity values for the different streams – regardless of their status (Before, After, Control, Impact) – show differences through time. The dissimilarity plots shown in Figure 1 corroborate these results, showing reasonable constancy in the mean dissimilarity values for each stream across all years.

Dissimilarity indices such as those used in Figure 1 may also be 'mapped' using multivariate ordination techniques to depict the relationship of the community sampled at any one site and sampling occasion with all other possible samples.

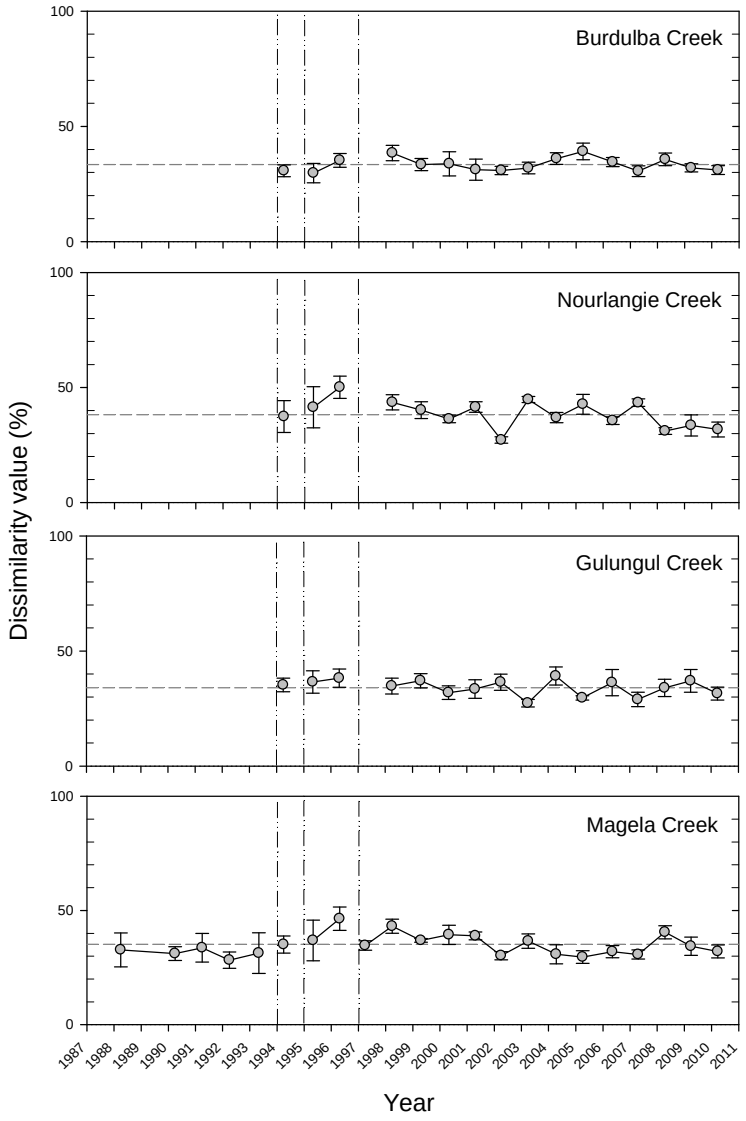

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2010. The dashed vertical lines delineate periods for which a different sampling and/or sample processing method was used. Dashed horizontal lines indicate mean dissimilarity across years. Dissimilarity values represent means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of upstream-downstream replicate samples within each stream.

Figure 2 depicts the ordination derived using the same replicate macroinvertebrate data used to construct the dissimilarity plots in Figure 1. Samples close to one another in the ordination indicate a similar community structure. Data points are displayed in terms of the sites sampled in Magela and Gulungul Creeks downstream of Ranger for each year of study (to 2010), relative to Magela and Gulungul Creek upstream (control) sites for 2010, and all other control sites sampled up to 2010 (Magela and Gulungul upstream sites, all sites in Burdulba and Nourlangie).

Because the data-points associated with the two 'exposed' sites (downstream Magela and Gulungul) are generally interspersed among the points representing the control sites, this indicates that these sites have macroinvertebrate communities that are similar to those occurring at control sites. This was verified using PERMANOVA (PERmutational Multivariate ANalysis Of Variance) (Anderson et al 2008), a new multivariate statistical approach used to determine if a priori groups, exposure type ('exposed' Magela and Gulungul Creeks vs control Burdulba and Nourlangie Creeks) and site location (upstream vs downstream), and the interaction between these two factors, show significant differences. PERMANOVA conducted on (i) all replicate data from all available years and sites, and (ii) replicate data from all sites from 2010 only, showed no significant differences for both factors and their interaction ($P>0.05$).

Collectively, these graphical and statistical results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining activities between 1994 and 2010 have not adversely affected macroinvertebrate communities.

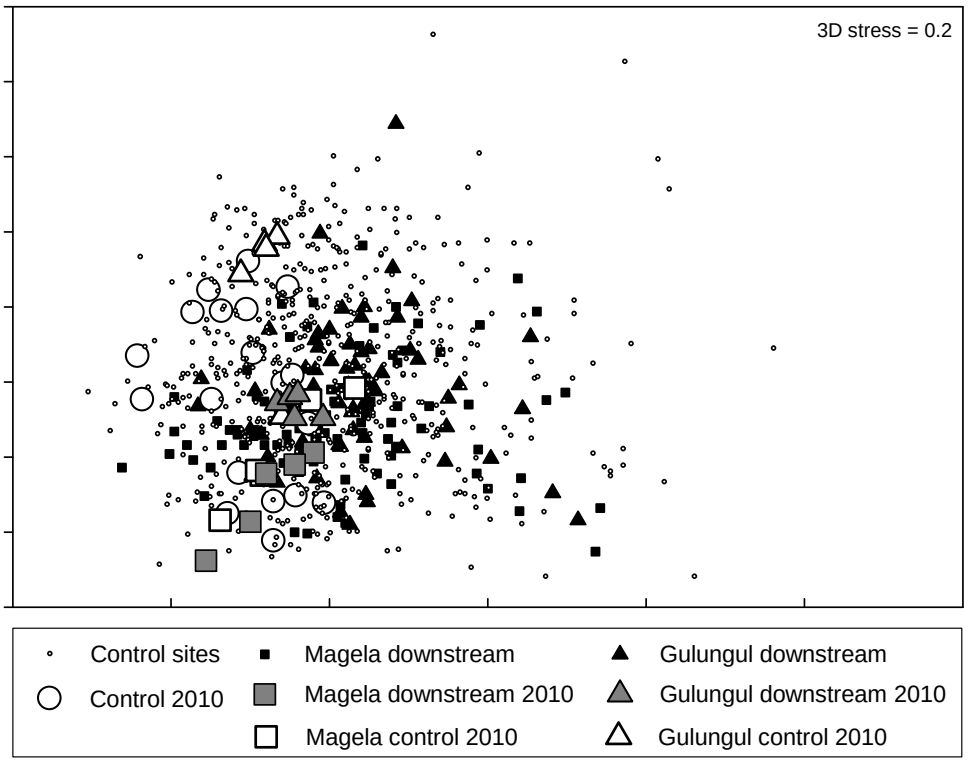

Figure 2 Ordination plot of macroinvertebrate community structure data from sites sampled in several streams in the vicinity of Ranger mine for the period 1988 to 2010. Data from Magela and Gulungul Creeks for 2010 are indicated by the enlarged symbols.

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Monitoring using fish community structure

D Buckle, C Davies & C Humphrey

Background

Assessment of fish communities in billabongs is conducted during the recessional flow period between late April and July each sampling year, with the precise timing being determined by the magnitude of rainfall and associated stream flow in the antecedent wet season. Data are gathered using non-destructive sampling methods from 'exposed' and 'control' sites. Visual (ie census) recording of each fish species and their relative abundances is done annually in deep channel billabongs, while trap and release sampling is done every second year in shallow lowland billabongs dominated by aquatic plants. Details of the sampling methods and sites are provided in the 2003–04 Supervising Scientist Annual Report (chapter 2, section 2.2.3). The scope of the fish sampling programs was last reviewed in October 2006 and the refinements made at that time to the designs for shallow and channel billabong fish communities, respectively, are detailed in Buckle and Humphrey (2008, 2009).

For both deep channel and shallow lowland billabongs, comparisons are made between a directly-exposed billabong in the Magela Creek catchment downstream of Ranger mine and control billabongs from an independent non-mining catchment (Nourlangie and Wirnmuyurr Creeks). The similarity of fish communities in exposed sites to those in control sites is determined using multivariate dissimilarity indices, calculated for each sampling occasion. The use of dissimilarity indices has been described and defined above (see 'Monitoring using macroinvertebrate community structure' pp 70–73, in this volume). A significant change or trend in the dissimilarity values over time could imply mining impact.

Results

Channel billabongs

The similarity of fish communities in Mudginberri Billabong (directly exposed site downstream of Ranger in Magela Creek catchment) and Sandy Billabong (control site in the Nourlangie Creek catchment) was determined using multivariate dissimilarity indices calculated for each annual sampling occasion. A plot of the dissimilarity values from 1994 to 2010 is shown in Figure 1.

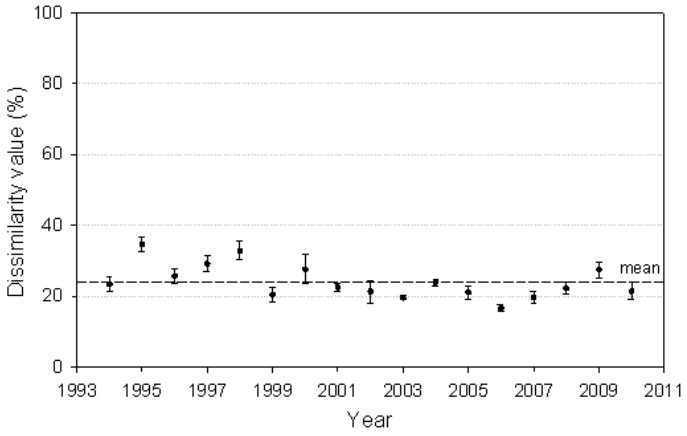

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of the Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of transect data between the two waterbodies.

In previous reports, possible causes of trends in the annual paired-site dissimilarity measure over time have been advanced and assessed. Because the dissimilarity measure is most influenced by numerically-abundant fish species, it was possible to demonstrate that fluctuations in the measure over time were directly associated with longer-term changes in abundance in Magela Creek of the chequered rainbowfish (*Melanotaenia splendida inornata*), the most common fish species in this creek system (Supervising Scientist 2004, pp 35–38). Thus, effort has been directed at understanding the possible causes of interannual variations in the abundance of this fish species in Magela Creek.

Buckle et al (2010) observed a negative correlation between annual rainbowfish abundance in Mudginberri Billabong and the magnitude of wet season discharge (total for the wet season, January total and February total) in Magela Creek. While a full analysis of community structure, and in particular chequered rainbowfish abundance, data for the channel billabongs in 2010 was still being conducted at the time of completing this report, the greatly reduced abundance of rainbowfish in Mudginberri Billabong for the 2010 sampling compared with the higher fish numbers recorded in 2009 (Figure 2) supported the abundance-discharge correlation, as the antecedent wet season discharge was above average (Figure 2), whilst the 2008–09 wet season discharge was well below average when abundance of rainbowfish was high. Furthermore, the late rains during April 2010 may have resulted in greater upstream migration of rainbowfish past Mudginberri Billabong, thereby reducing the concentration of fish in Mudginberri Billabong during the recessional flow period.

The dissimilarity value for 2010 is consistent with the range of values reported since 2001, a period over which there has been no evidence of mine-associated changes to fish communities in Mudginberri Billabong, downstream of Ranger (Buckle & Humphrey 2009).

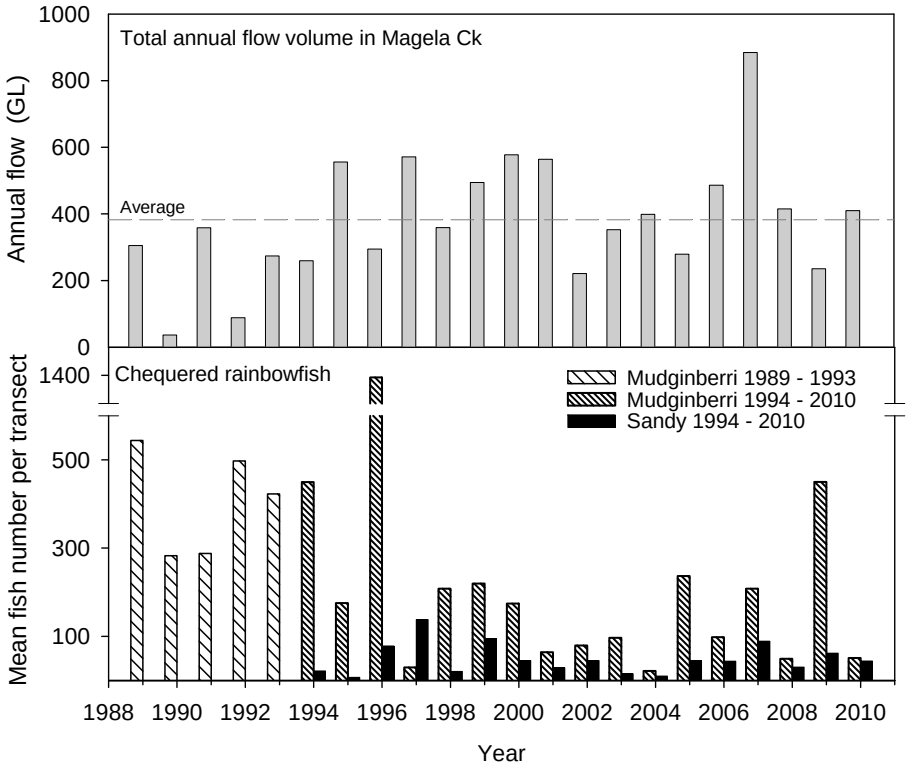

Figure 2 Relative abundance of chequered rainbowfish in Mudginberri and Sandy Billabongs from 1989 to 2010 with associated total discharge in Magela Creek (gauging station G8210009)

Shallow lowland billabongs

Monitoring of fish communities in shallow billabongs is conducted every other year (see Buckle & Humphrey 2008). The last assessment of fish communities in shallow lowland billabongs was conducted in May 2009 with results reported in Buckle et al (2010). The next assessment is scheduled for the recessional flows sometime between late April and June 2011.

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Stream monitoring program for the Magela Creek catchment: research and development

Introduction

C Humphrey, A Bollhöfer & D Jones

Progress under this component of the stream monitoring program for the Magela Creek catchment is reported by way of (i) results from the monitoring program conducted in the 2009–10 period, and (ii) monitoring support tasks for the same period, including research and development, reviews and reporting. Results under Part (i) are reported in 'Results from the routine stream monitoring program in Magela Creek catchment, 2009–10', pp 46–76, this volume.

Tasks under Part (ii) are reported below where a summary is provided on in situ biological (toxicity) monitoring in Gulungul Creek. Prior to reading this summary, it is advisable to read the introductory section of the accompanying Part (i) paper describing the rationale of the monitoring program and hence the context for the research and development outlined below.

In situ biological monitoring in Gulungul Creek

C Humphrey, D Buckle & C Davies

SSD has expanded its environmental monitoring effort in Gulungul Creek in recognition of the increasing potential for catchment impacts due to runoff from the recently lifted Ranger mine tailings dam walls and the prospect of construction of additional mine-site infrastructure. In addition to upgrading the continuous monitoring equipment in the creek, biological (toxicity) monitoring was also initiated in the 2009–10 wet season with the trial deployment of the in situ freshwater snail reproduction test. This method of biological monitoring has been routinely deployed in Magela Creek over many years and the results have been documented in previous Annual Reports and Annual Research Summaries. As with toxicity monitoring in Magela Creek (see KKN 131 Stream Monitoring Results), it is intended that in situ biological monitoring will be used in Gulungul Creek as an early detection method for identifying changes in water quality.

The trial deployment was conducted firstly to establish the logistics of reliably conducting toxicity monitoring in the creek and secondly to start acquiring biological response data to develop a baseline prior to any significant future disturbance in the catchment. The test design used was the same as that used for the routine monitoring of Magela Creek ((KKN 131 Stream Monitoring Results) with upstream 'control' and downstream 'exposed' sites co-located with water quality monitoring (Gulungul u/s and Gulungul d/s on Map 2). While the control and exposed sites in Magela Creek are accessible by boat throughout the wet season, the upstream control site on Gulungul Creek is not accessible by boat at any time, nor by road for the majority of the wet season. Hence it is necessary to access this site by helicopter.

Five tests were conducted during the 2009–10 wet season, over a range of flow conditions, and in alternate weeks to the routine Magela Creek testing. Tests were conducted in the periods 25–29 January, 22–26 February, 22–26 March, 9–13 April and 19–23 April 2010. The results, together with comparative results from Magela Creek, are shown in Figure 1. The range in egg number observed in Gulungul Creek was similar to that recorded in Magela Creek (Figure 1).

Four out of the five tests resulted in positive difference values, ie egg production was higher upstream than downstream. This pattern was opposite to that observed in Magela Creek during the same period, where eight of the nine tests resulted in a negative difference value (Figure 1). High statistical power in this toxicity monitoring technique is potentially available when, in the absence of human-related disturbance downstream of potential sources of impact, the responses measured at upstream and downstream sites are very similar in magnitude to one another over time. This concordance (or 'tracking') in egg number between upstream and downstream sites is the typical pattern in Magela Creek (Figure 1), and also appears to be the pattern in Gulungul Creek.

It is anticipated that fortnightly in situ toxicity monitoring will be implemented in Gulungul Creek during the 2010–11 wet season.

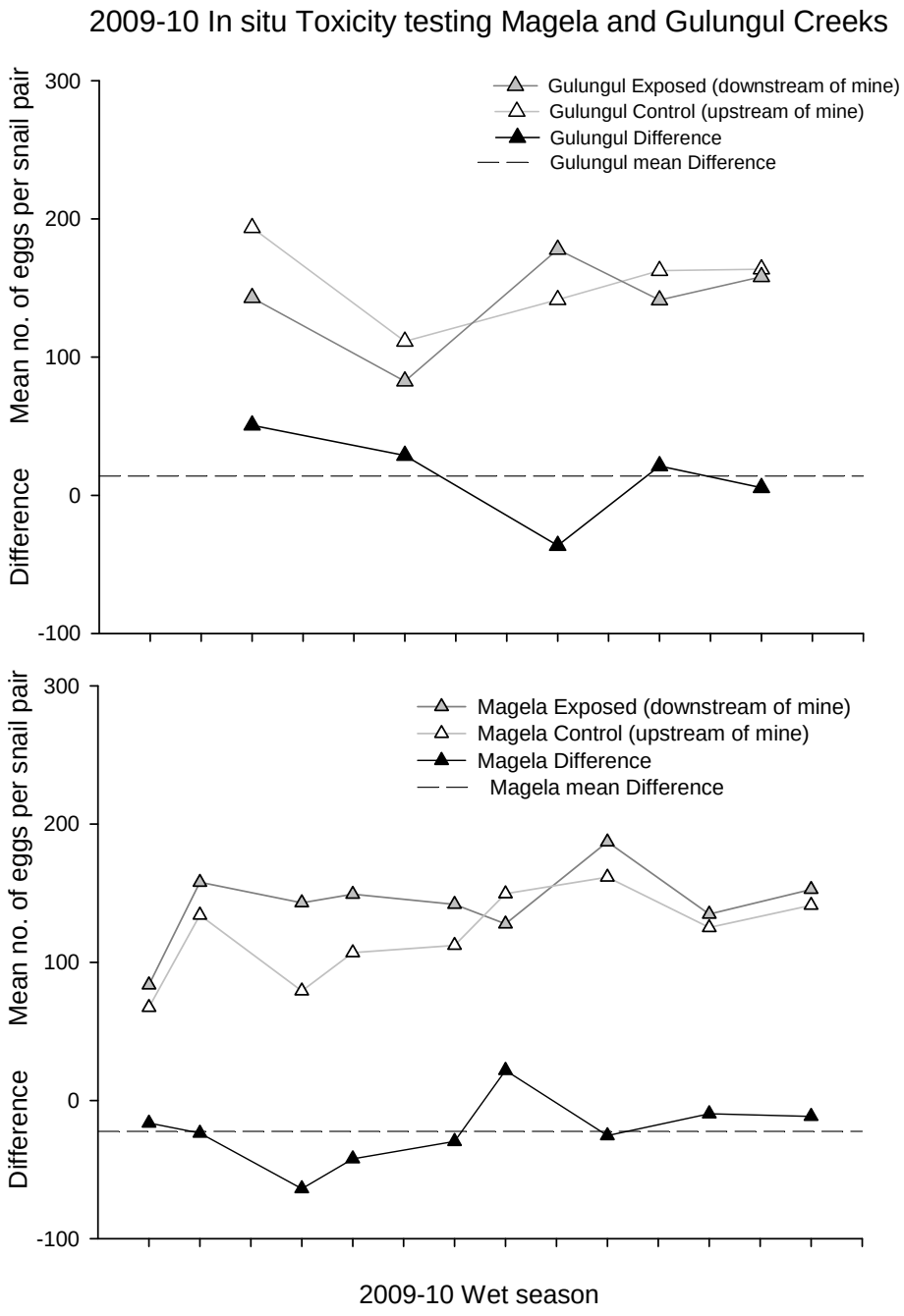

Figure 1 In situ toxicity monitoring results for freshwater snail egg production for Gulungul Creek compared with results from Magela Creek, 2009–10 wet season