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eriss research summary 2009–2010

DR Jones & AL Webb (eds)

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Department of Sustainability, Environment, Water, Population and Communities
Supervising Scientist

It is SSD policy for reports in the SSR series to be reviewed as part of the publications process.

This Supervising Scientist Report is a summary of the 2009–2010 research program of the Environmental Research Institute of the Supervising Scientist and has been reviewed internally by senior staff and the editors of this volume.

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Contents

Preface

The Environmental Research Institute of the Supervising Scientist (***eriss***) is part of the Supervising Scientist Division (SSD) of the Australian Government's Department of Sustainability, Environment, Water, Population and Communities (SEWPaC). ***eriss*** provides specialist technical advice to the Supervising Scientist on the protection of the environment and people of the Alligator Rivers Region (ARR) from the impact of uranium mining. Its major function is to conduct research into developing best practice methodologies for monitoring and assessing the impact of uranium mining on water and air (transport pathways) and soil, and on the bushfoods that are consumed by the local indigenous people. This research spans the operational, decommissioning, and post rehabilitation phases of mining.

eriss also applies its expertise to conducting research on the sustainable use and environmental protection of tropical rivers and their associated wetlands, and engaging in a limited program of contract research on the impacts of mining elsewhere in the north Australian tropics.

The balance and strategic prioritisation of work within the uranium component of ***eriss***'s project portfolio is defined by Key Knowledge Needs (KKNs) developed by consultation between the Alligator Rivers Region Technical Committee (see ARRTC membership and function in Appendix 2), the Supervising Scientist, Energy Resources of Australia Ltd (ERA) and other stakeholders. The KKNs are reviewed periodically (approximately every three years) to ensure their currency in the context of any significant changes that may have occurred in U-mining related activities and issues in the ARR. The current revision of the KKNs will apply until the end of 2010.

Not all of the KKN research areas (Appendix 3) are able to be covered by ***eriss***, since not all of the required disciplines are available within the Institute. To address these particular gaps, collaborative projects are conducted between ***eriss*** and researchers from other organisations, and consultants are commissioned by ***eriss*** and others to undertake specific pieces of work. For example, KKN projects related to detailed hydrogeology or tailings management on the Ranger lease are conducted and reported separately by consultants engaged by ERA. A more complete picture of the scope of research work that is conducted by all parties can be obtained by referring to the minutes that are produced for the meetings of ARRTC: www.environment.gov.au/ssd/communication/committees/arrtc/meeting.html.

This report documents research projects undertaken by ***eriss*** over the 2009–10 financial year (1.7.09 to 30.6.10). A particular focus was the consolidation of all components of the research program that underpin the acquisition and interpretation of continuous water quality monitoring data, as SSD moves towards this being its primary monitoring method for the 2010–11 wet season and beyond. A key component was continuing with the extensive ecotoxicological testwork, involving exposure of a suite of five aquatic test organisms to pulses of magnesium over periods of 4, 8 and 24 h, required to derive appropriate trigger values spanning this range of exposure durations. Other major areas of activity related to the acquisition of data from erosion plots constructed on the Ranger Trial Landform and assisting Parks Australia Operations with the final phases of radiological assessment for rehabilitation activities in the South Alligator River valley. The final phases of data analysis and reporting were also completed for projects to assess the current status of surface and groundwater at the Rum Jungle legacy site.

The uranium mining section of the research summary is structured according to the five major topic areas in the KKN framework, noting that this year there are no research papers for Jabiluka or Nabarlek.

1 Ranger – current operations

2 Ranger – rehabilitation

3 Jabiluka

4 Nabarlek

5 General Alligators Rivers Region

The Jabiluka site is in long-term care and maintenance and the current work of the Supervising Scientist is focused on maintaining a routine continuous monitoring program for flow and electrical conductivity downstream of the formerly disturbed area. The Nabarlek lease was taken over by Uranium Equities Ltd to pursue exploration activities. Environmental monitoring and assessment for this site is being conducted via Mining Management Plans submitted to the Northern Territory Government.

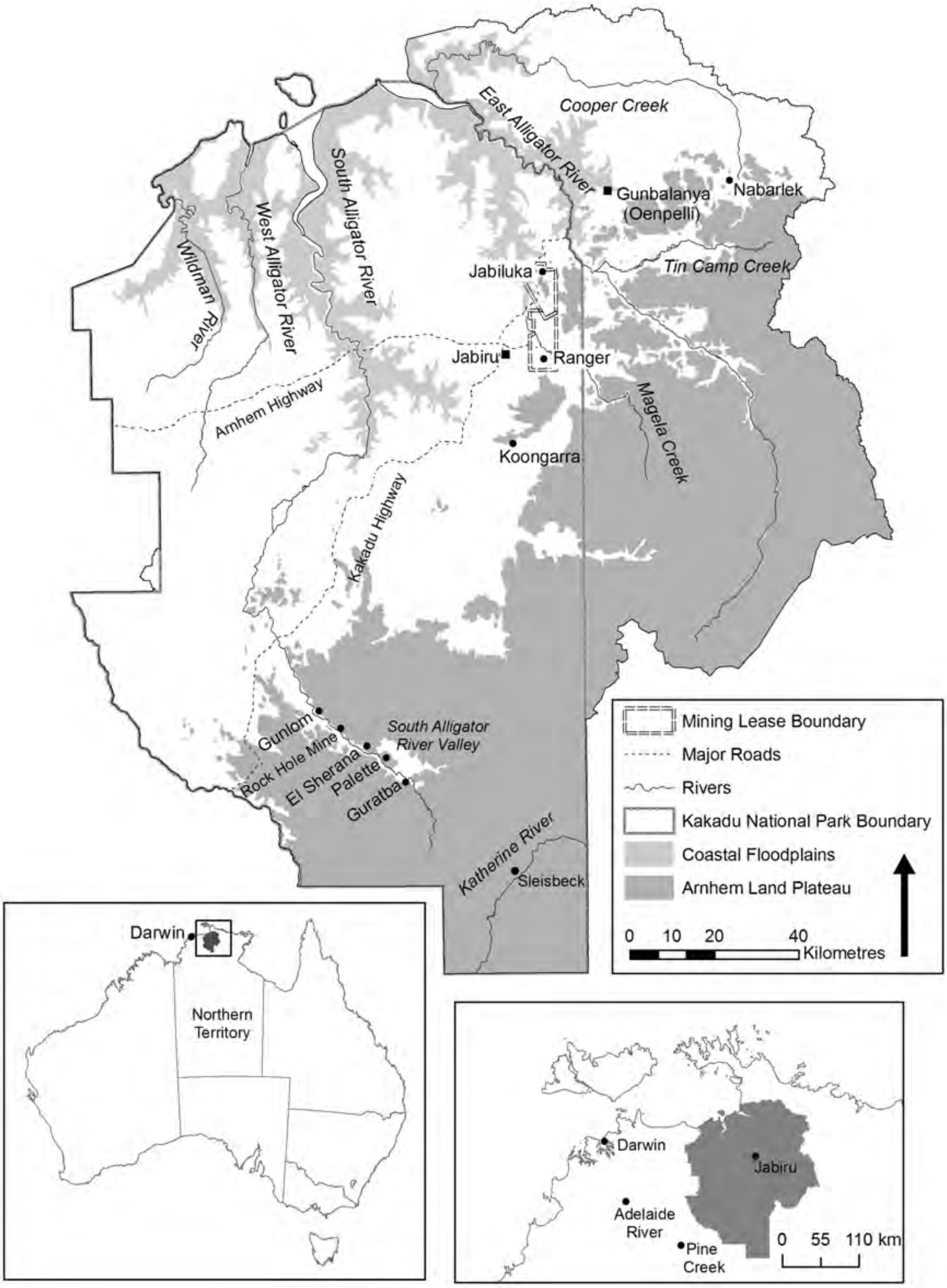
Three maps (following this Preface) provide the regional context for the locations that are referenced in the research papers. Map 1 shows Kakadu National Park and the locations of the Ranger mine, Jabiluka project area, the decommissioned Nabarlek mine, and the South Alligator River valley. A schematic of the Ranger minesite is provided for reference in Map 2. Map 3 shows the locations of billabongs and other waterbodies used for the aquatic ecosystem monitoring and atmospheric and research programs for assessing impacts from Ranger mine.

The final section of this report contains summaries of the non-uranium mining related external projects. Commercial-in-confidence projects have been excluded from this compilation.

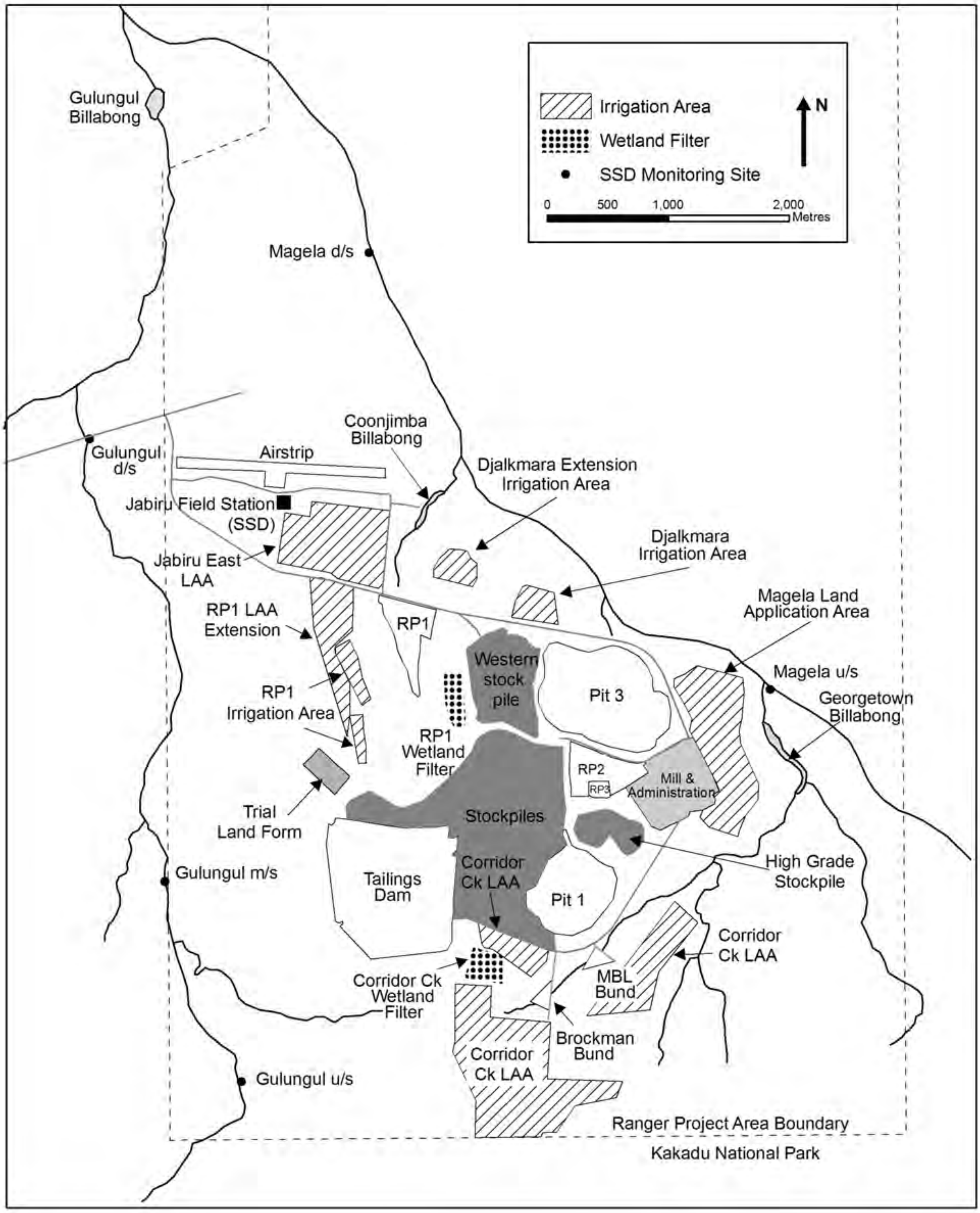
For additional information, readers are referred to the annual publications list (Appendix 1) that details all of the material published, and conference and workshop papers presented by ***eriss*** staff in 2009–10.

Dr DR Jones

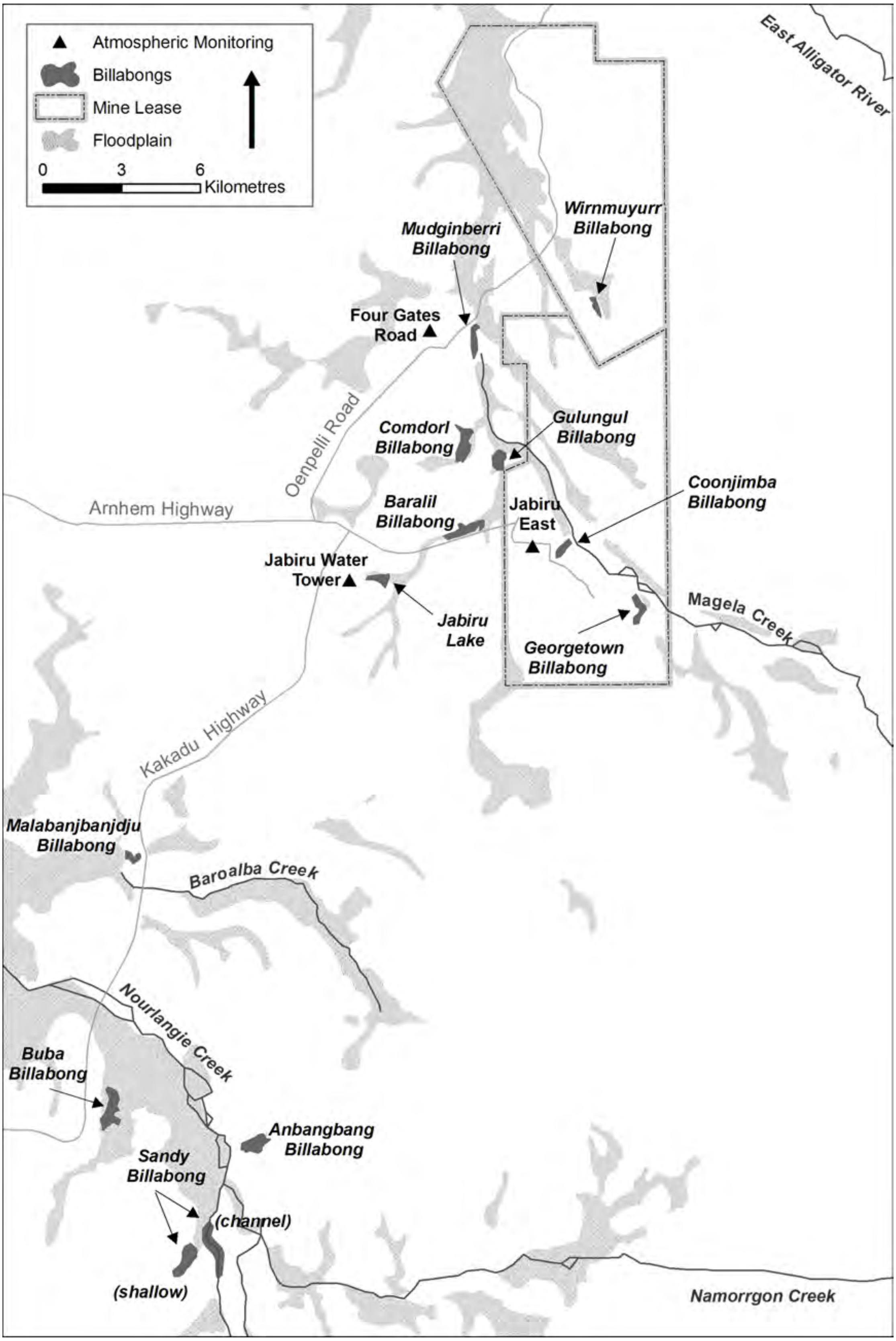
Director, Environmental Research Institute of the Supervising Scientist



Map 1 Alligator Rivers Region



Map 2 Ranger minesite showing adjacent billabongs, creek systems and key water quality monitoring sites



Map 3 Location of waterbodies and atmospheric monitoring sites used in the SSD environmental research and monitoring programs

Part 1: Ranger – current operations

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

S Parker, R van Dam & R Bartolo

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.

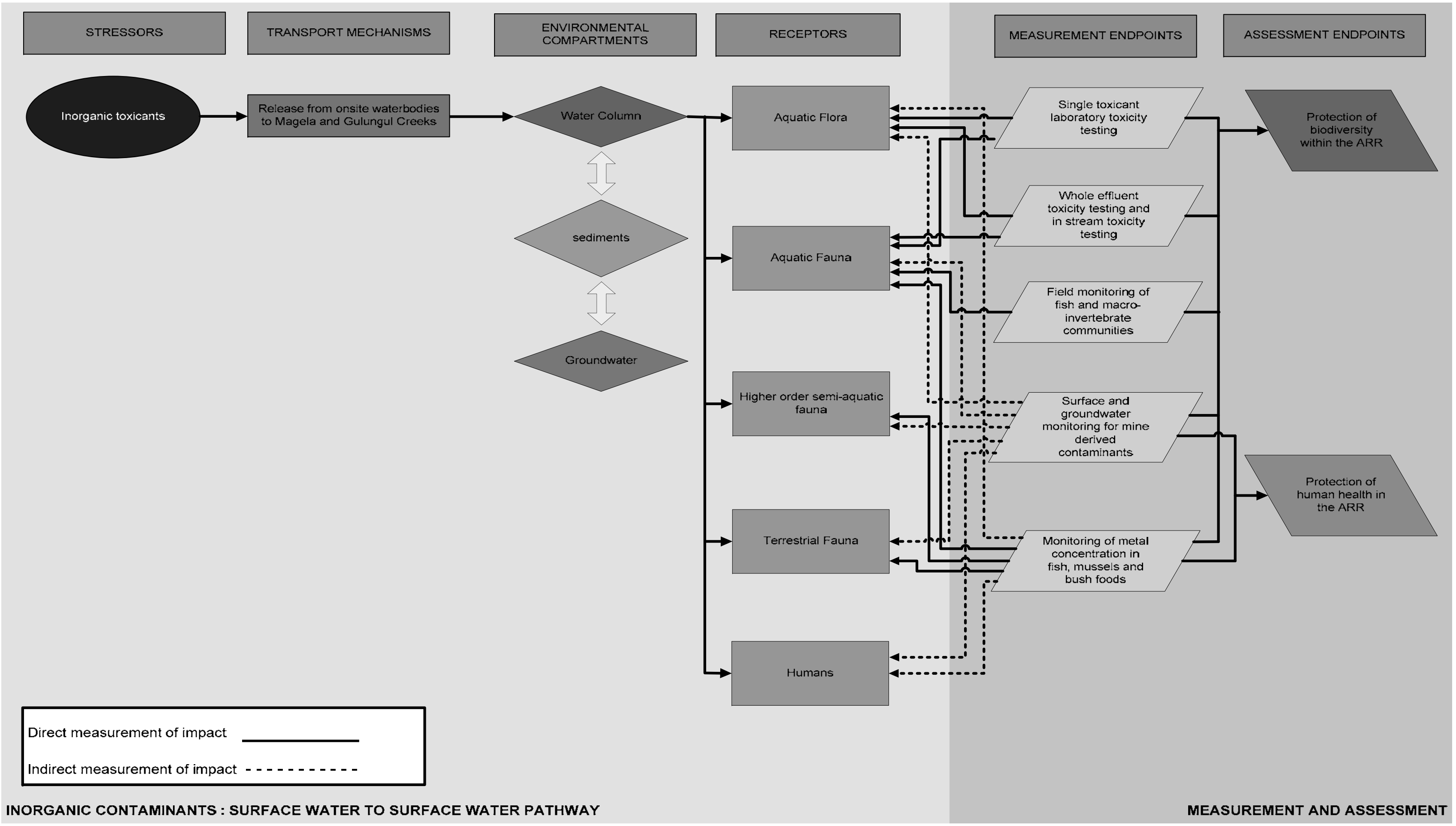


Figure 1 Conceptual model diagram for transport of inorganic toxicants from Ranger uranium mine via surface water to surface water pathway

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Radiological characterisation of Ranger mine land application areas

R Akber[1], A Bollhöfer & P Lu[2]

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.

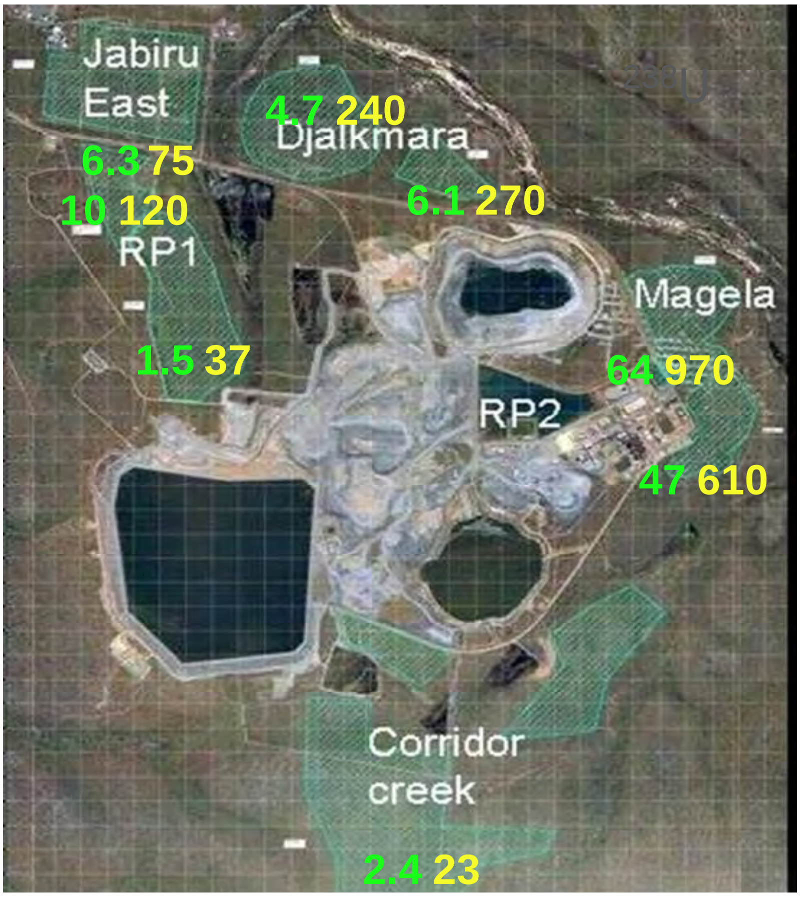

Figure 1 ^{238}U (yellow – white in printed copy) and ^{226}Ra (green – grey in printed copy) aerial activity densities ($kBq \cdot m^{-2}$) calculated from historical radionuclide inventories in RP2 water and irrigation rates provided by ERA (from Akber et al 2010)

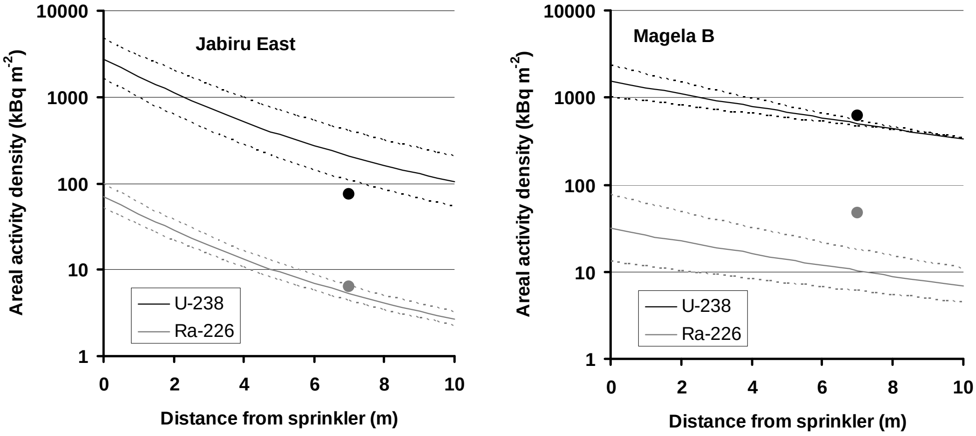

Figure 2 Modelled aerial activity densities for ^{238}U (black) and ^{226}Ra (grey) and comparison with the applied loads (dots) calculated from historical radionuclide inventories in RP2 water and irrigation rates provided by ERA

Preliminary dose estimate

Moroney (1992) has developed a model for the LAAs that allows to calculate a committed effective dose rate equivalent ($Sv \cdot yr^{-1}$) for all exposure pathways from the aerial activity densities ($kBq \cdot m^2$) 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 $(8.9 \pm 4.3)\cdot10^{-7}$ $Bq\cdot m^{-3}$ in air per $Bq\cdot kg^{-1}$ 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}		
	$(Sv\cdot yr^{-1})$ per $(kBq\cdot m^{-2})$	kBq/m^2	$mSv\cdot yr^{-1}$	$(Sv\cdot yr^{-1})$ per $(kBq\cdot m^{-2})$	kBq/m^2	$mSv\cdot yr^{-1}$
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

M Trenfield, J Ng[1], B Noller[2], S Markich[3] & R van Dam

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.

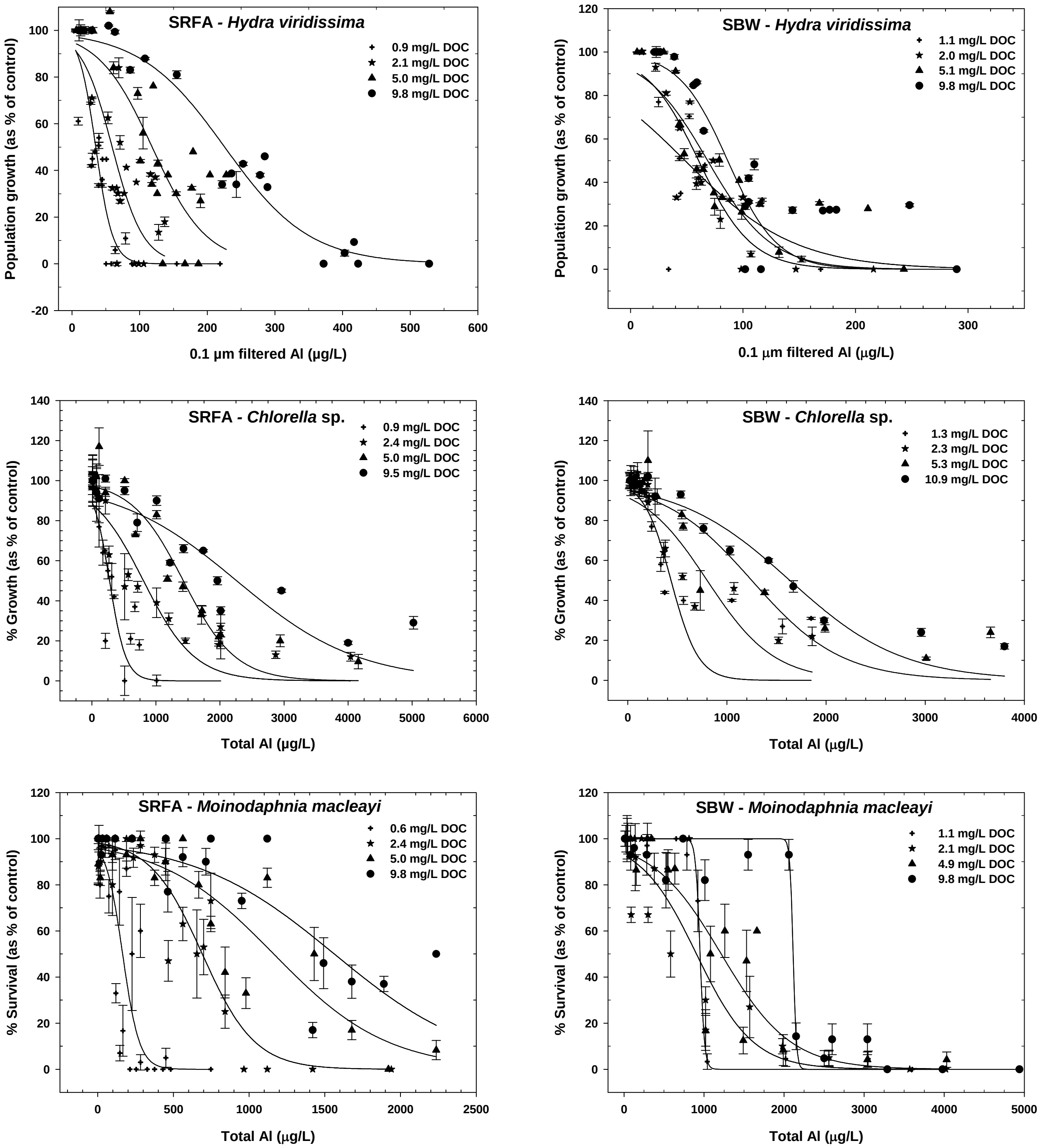

Figure 1 Concentration-response plots for Al exposures. Left: using Suwannee River fulvic acid (SRFA) in dilute Magela Creek water, 4 pooled tests for each species. Right: Sandy Billabong Water (SBW) diluted in synthetic Magela Creek water, 3 pooled tests for each species. Data points represent the mean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

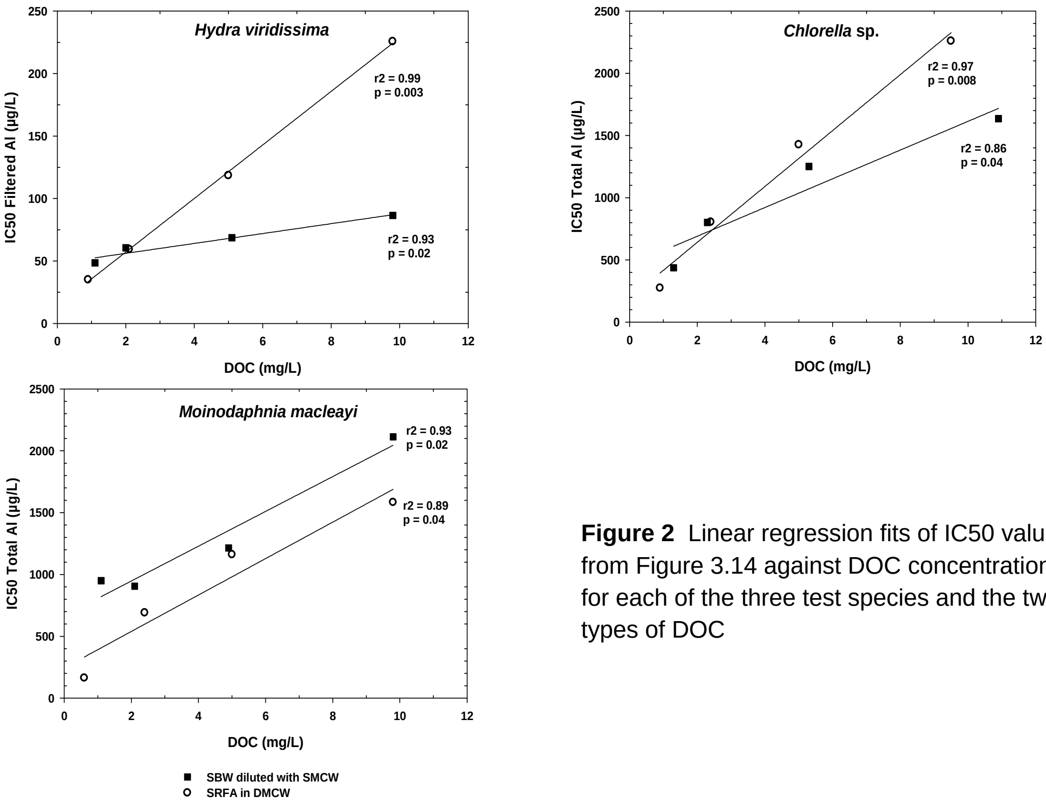

Figure 2 Linear regression fits of IC50 values from Figure 3.14 against DOC concentrations for each of the three test species and the two types of DOC

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)	21	4.0
(green hydra)	2	59 (40–71)	61 (48–72)		
	5	119 (91–138)	69 (54–81)		
	10	226 (204–242)	87 (65–101)		
Chlorella sp	1	275 (189–384)	437 (315–679)	225	115
(unicellular alga)	2	805 (560–1032)	801 (560–1134)		
	5	1427 (1242–1582)	1251 (870–1724)		
	10	2260 (1830–2867)	1635 (1410–1895)		
Moinodaphnia macleayi	1	164 (123–206)	950 (939–983)	147	141
(cladoceran)[g]	2	691 (610–767)	905 (608–1293)		
	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

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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.

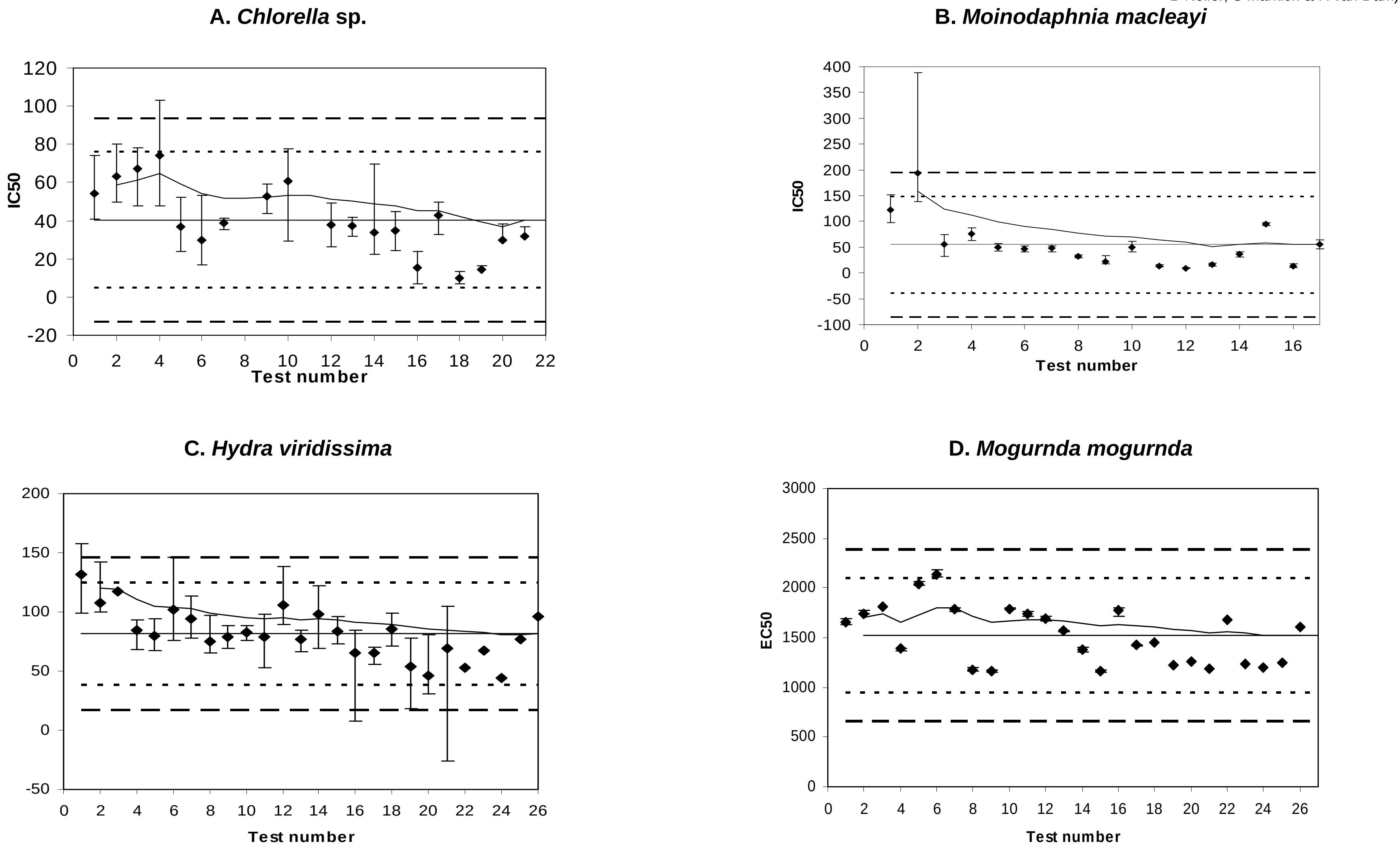

Figure 1 Reference toxicant control charts for A. *Chlorella* sp., B. *M. macleayi*, C. *H. viridissima* and D. *M. mogurnda*, as of September 2010. Data points represent IC_{50} or EC_{50} toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – 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	1064G	10 (7, 13)	Yes	
(72-h cell division rate)	1080G	15 (11, 17)	Yes	
	1091G	30 (26, 38)	Yes	
	1106G	32 (26, 37)	Yes	
Hydra viridissima	1029B	68 (65, 71)	Yes	
(96-h population growth)	1077B	45 (32, 59)	Yes	
	1099B	77 (72, 81)[a]	Yes	
	1136B	98 (86, 109)	Yes	
Moinodaphnia macleayi	1023I	95 (93, 98)	Yes	
(48-h immobilisation)	1078I	NC[b]	No	No effect at highest concentration[a]
	1103I	14 (11, 17)	Yes	
	1129I	55 (47, 64)	Yes	
Mogurnda mogurnda	1060E	1202 (1043, 1349)	Yes	
(96-h sac fry survival)	1098E	1252 (1108, 1373)	Yes	
	1123E	1582 (1469, 1689)	Yes	
Lemna aquinoctialis (96-h	1049L	10000 (7800 12000)	Yes	
population growth	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.

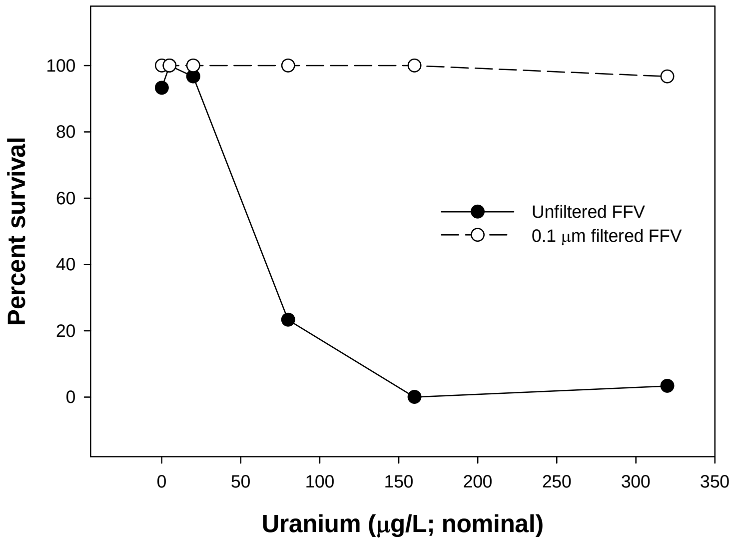

Figure 2 Effect of unfiltered (Test 1129I) and 0.1 µm filtered (Test 1130I) FFV (food source) on the toxicity of uranium to *Moinodaphnia macleayi*

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.

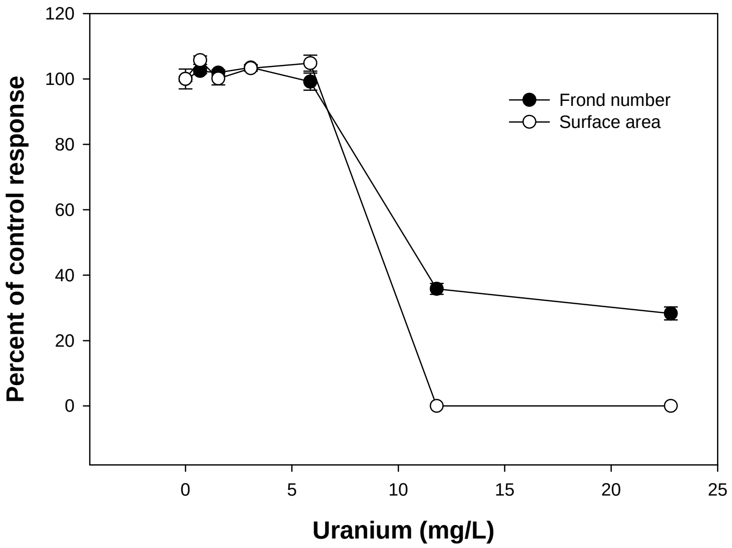

Figure 3 Comparison of growth rate based on frond number and surface area as endpoints to assess the toxicity of U to *Lemna aequinoctialis*

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).

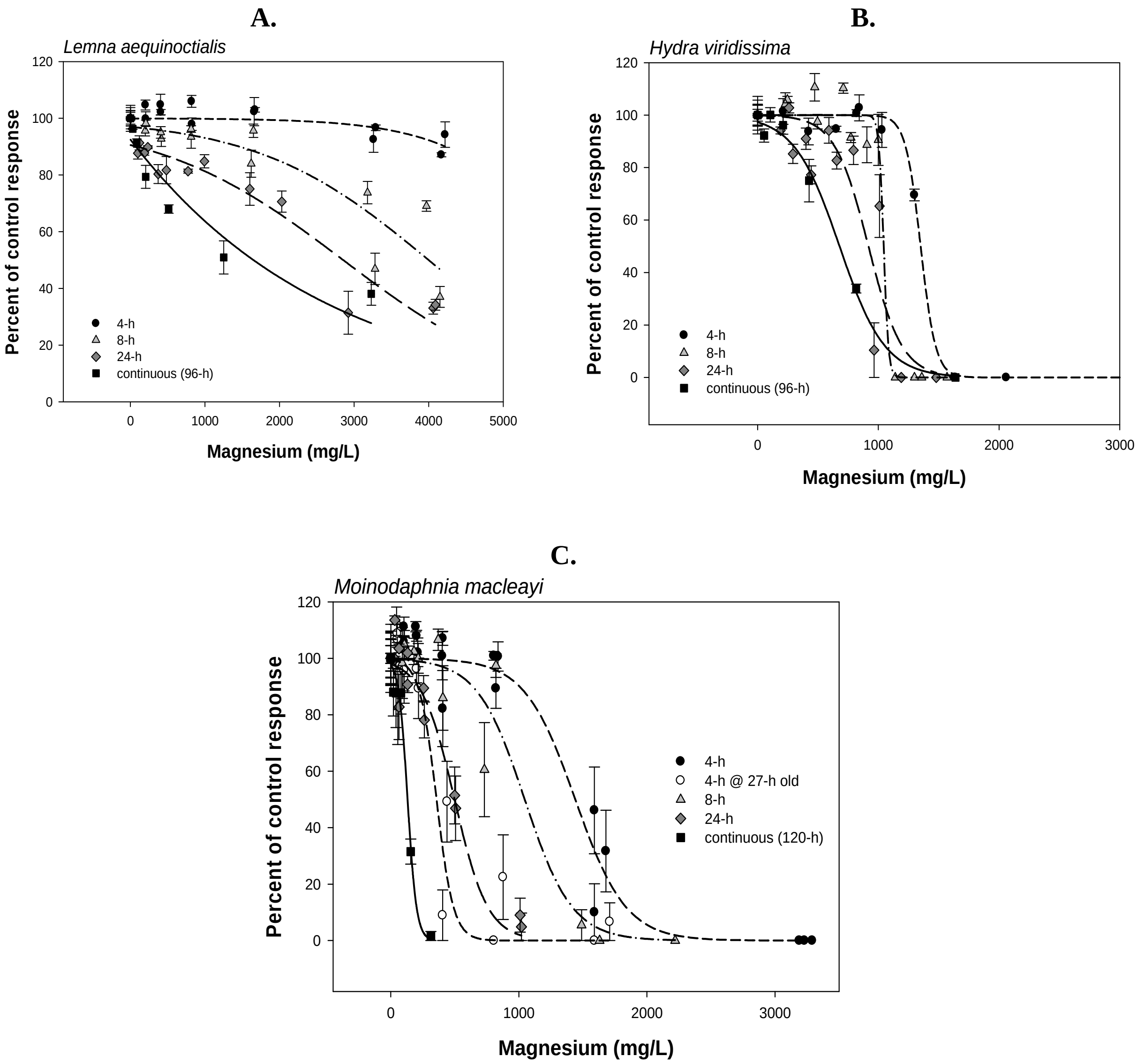

Figure 1 Toxicity of magnesium to A. duckweed, *Lemna aequinoctialis*, B. green hydra, *Hydra viridissima*, C. cladoceran, *Moinodaphnia macleayi*. Data from continuous exposure experiments are represented by a solid line, while 4 h pulse data are represented by short dashed lines, 8 h pulse data by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

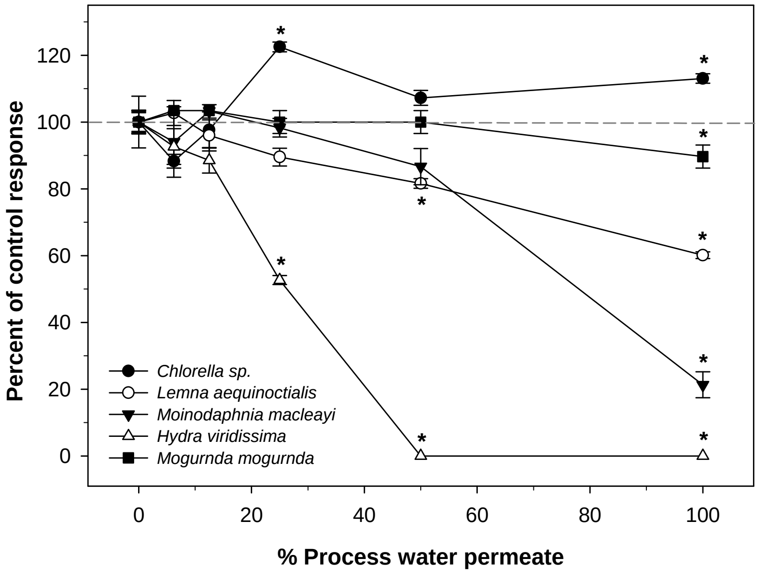

Figure 1 Responses of five tropical freshwater species to treated process water from Ranger Uranium Mine, expressed as percentages of the control response (see Table 2 for control response data). Data points represent the mean ± standard error of three replicates (10 replicates for *Moinodaphnia macleayi*). 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.

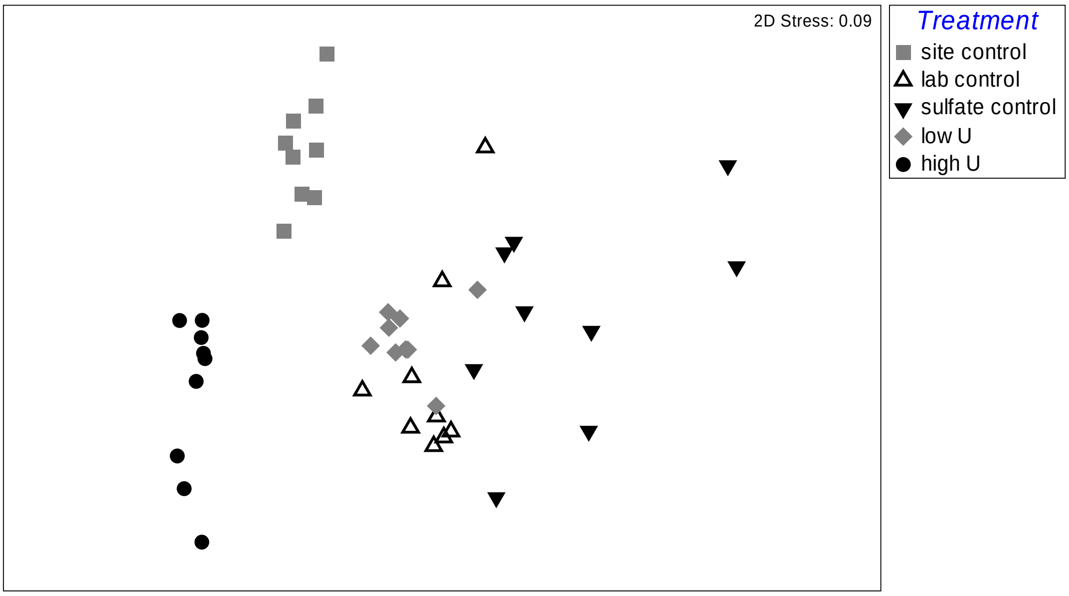

Figure 1 MDS (Multi Dimensional Scaling) ordination of microbial communities measured in treatment replicates through ecogenomics for the site (Gulungul) control and sediments treated with U and pure water (lab control). Low uranium = 400 mg/kg and high uranium = 4000 mg/kg.

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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 (*alphaNucle*ar, 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.

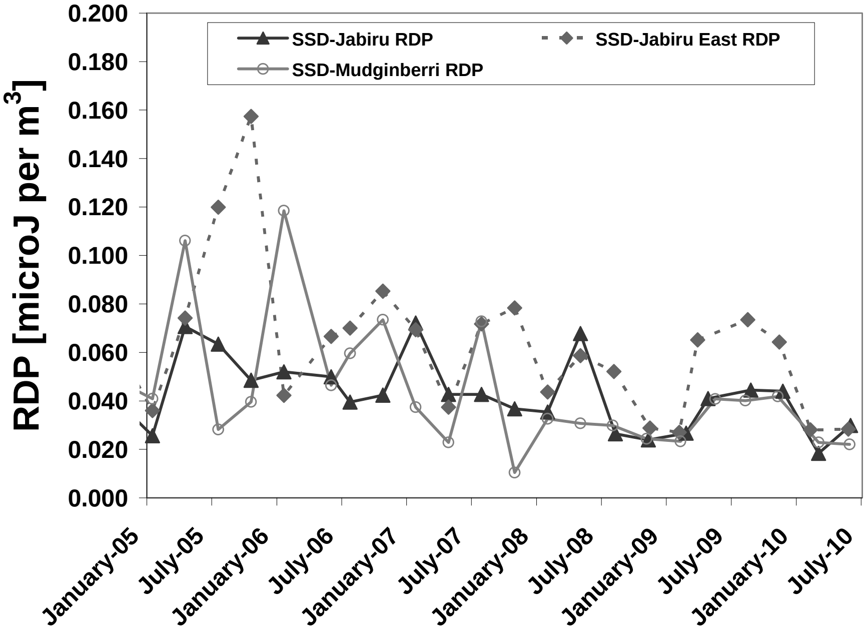

Figure 1 Radon decay product concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

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
	Jabiru East	0.064 (0.059)	0.046 (0.033)	0.057 (0.095)
RDP concentration [μJ/m^3]	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 nightime 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. .

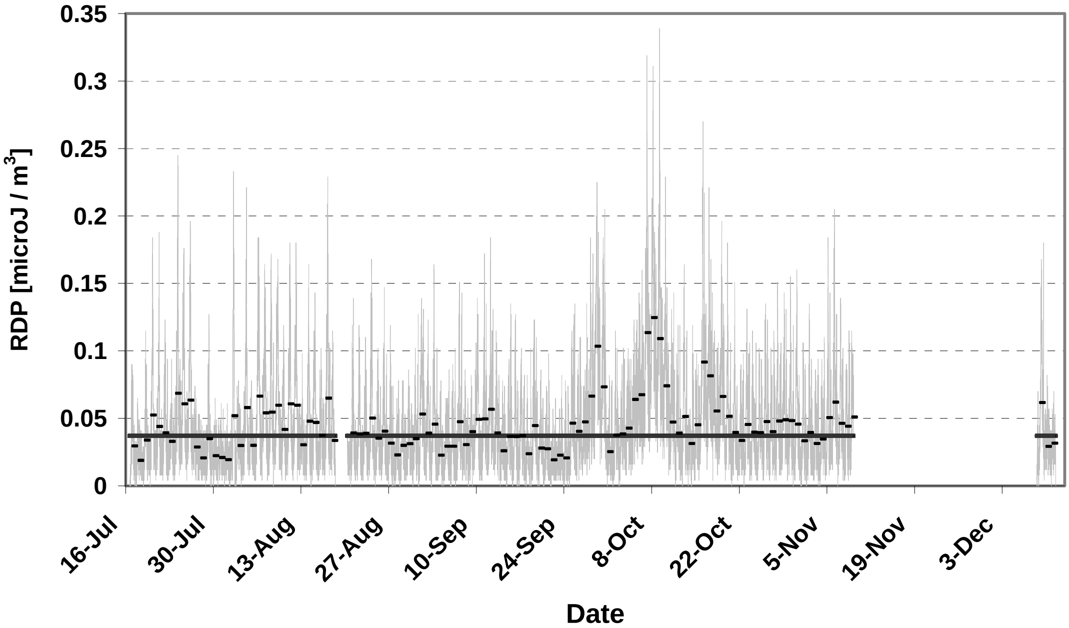

Figure 2 Continuous radon decay product concentrations (grey) measured at Mudginberri Radon Station between 16 July and 11 December 2009. The annual average (solid black line) and daily averages (black dashes) are overlaid for comparison.

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·m^{-3}, respectively. There is no statistically significant difference between the Mudginberri and Jabiru sites ($p = 0.289$).

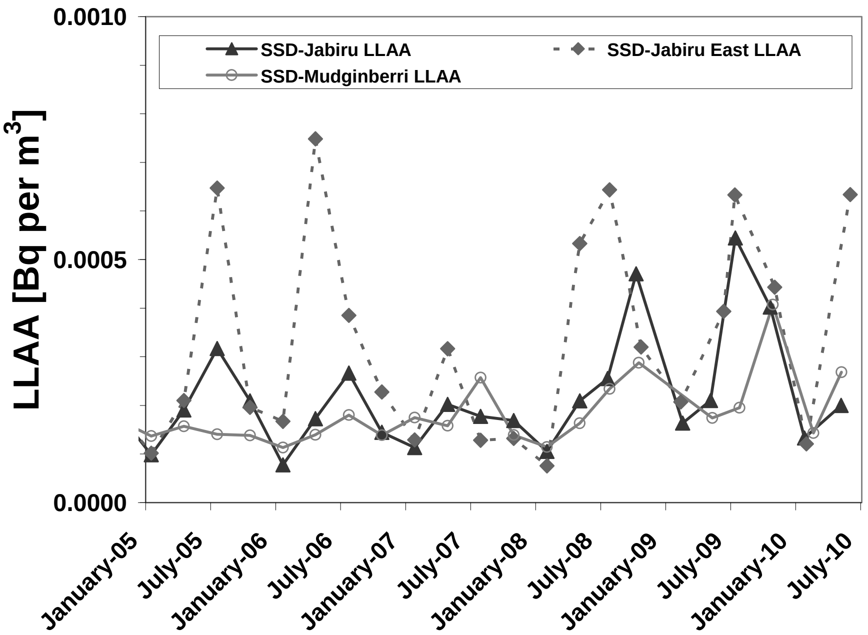

Figure 2 Long lived alpha activity concentration measured at Jabiru, Jabiru East and Mudginberri

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.

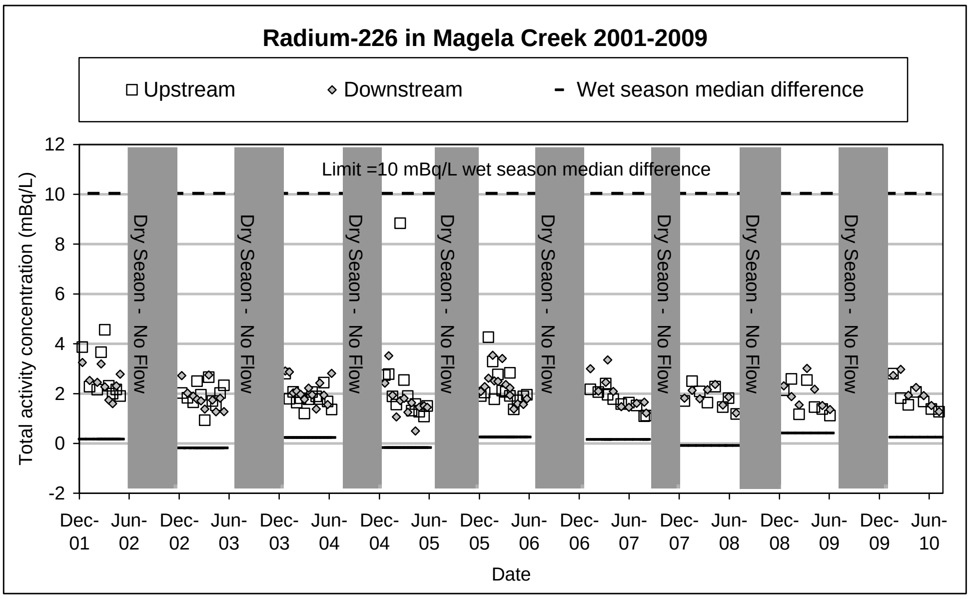

Figure 1 Total Radium-226 in Magela Creek for the 2001–2010 wet seasons

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.

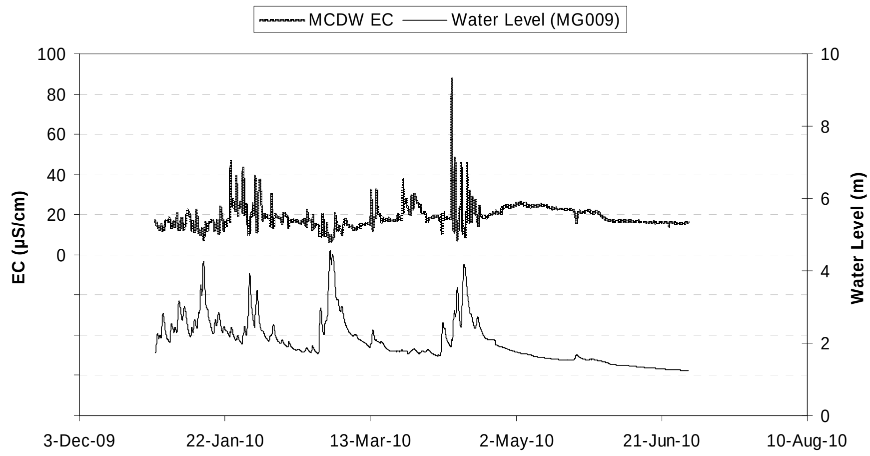

Figure 1 Electrical conductivity and discharge measurements in Magela Creek between December 2009 and July 2010 – continuous monitoring data

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).

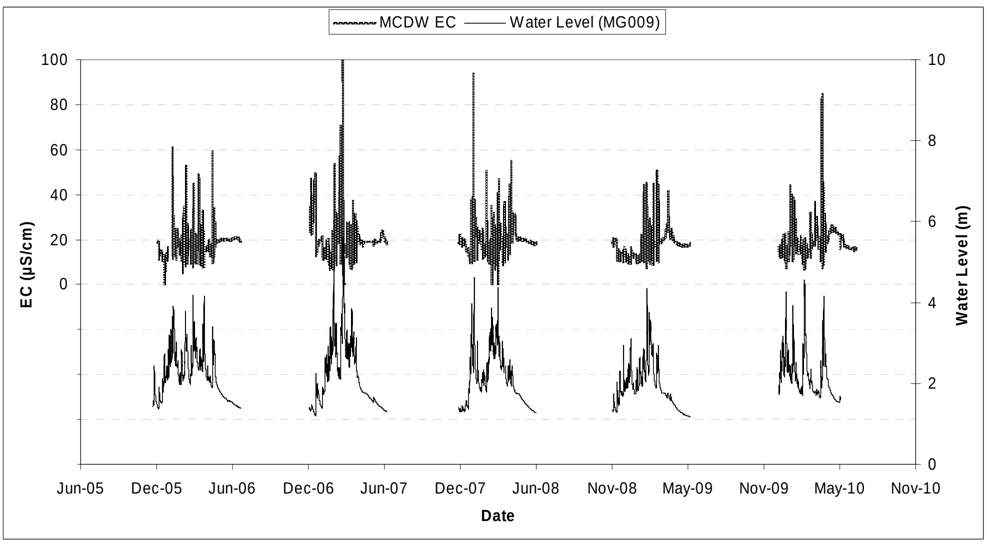

Figure 2 Electrical conductivity measurements and water level (lower trace) in Magela Creek (SSD data) between December 2005 and July 2010 – continuous monitoring data

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.

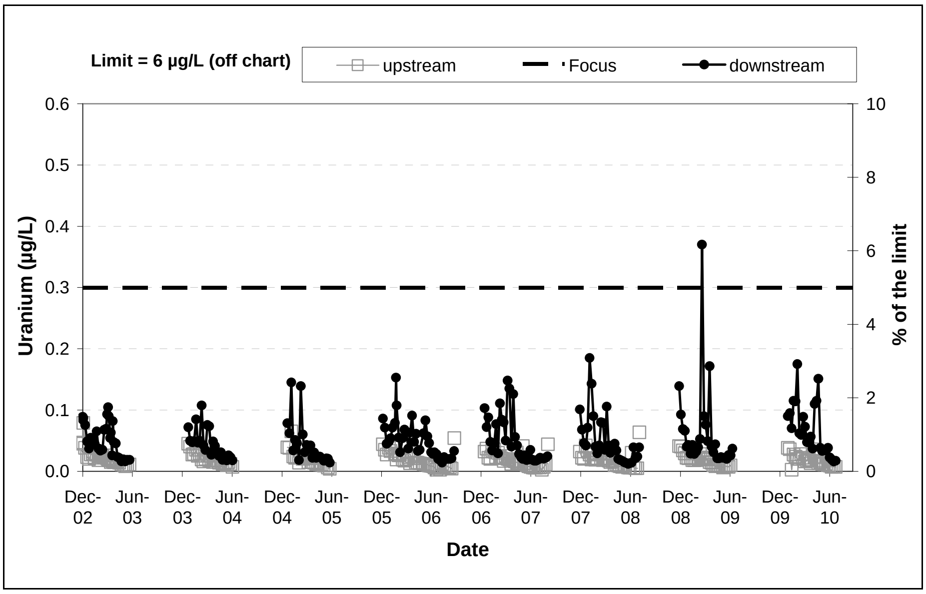

Figure 3 Uranium concentrations in Magela Creek since the 2002–03 wet season – grab sample data

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.

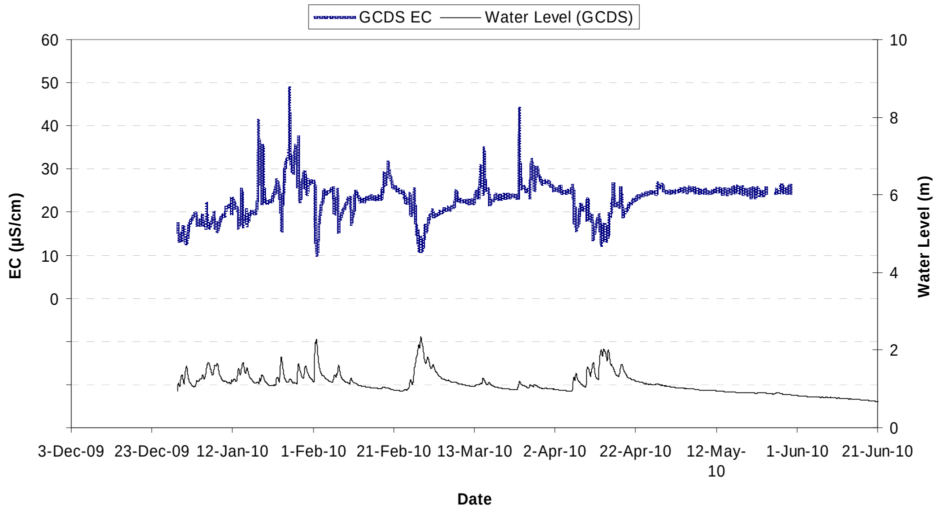

Figure4 Electrical conductivity measurements in Gulungul Creek between December 2009 and June 2010 – continuous monitoring data

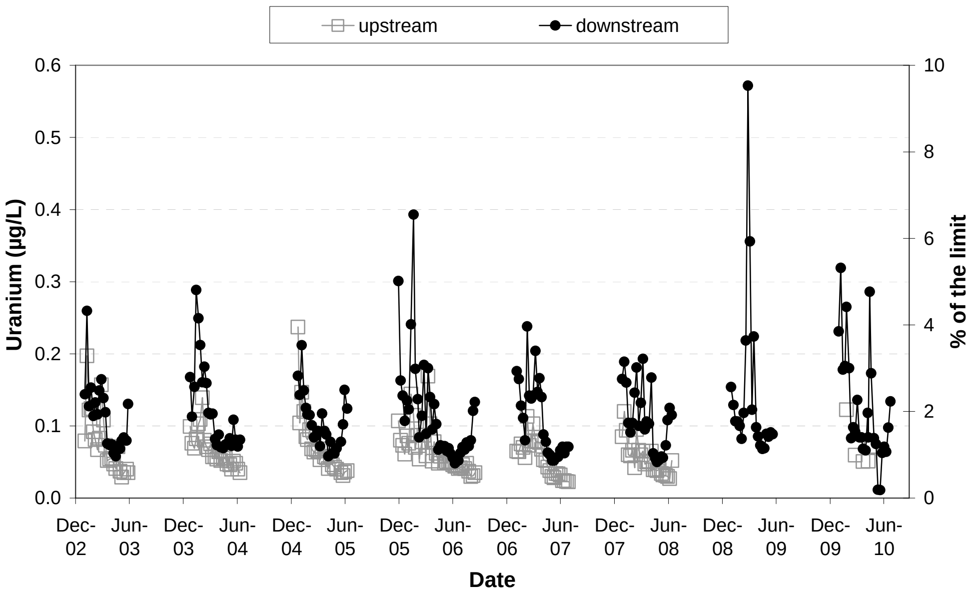

Figure 5 Uranium concentrations measured in Gulungul Creek by SSD between December 2002 and June 2010 – grab sample measurements.

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.

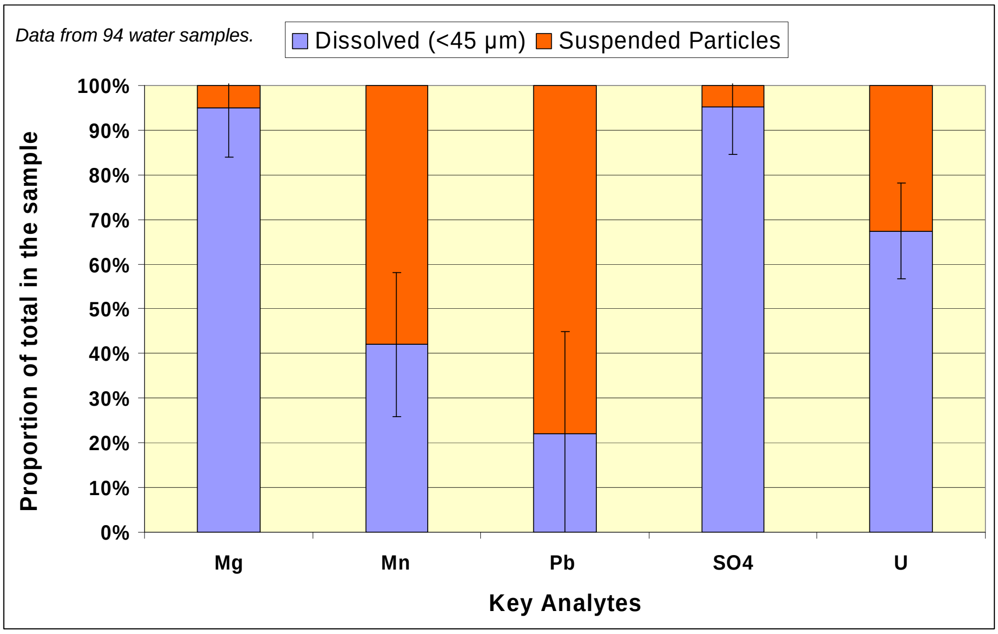

Figure 7 Proportions of the total analyte concentration associated with wither the dissolved (filterable through a 0.45 µm filter) or particulate fractions. Error bars represent the standard deviation.

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.

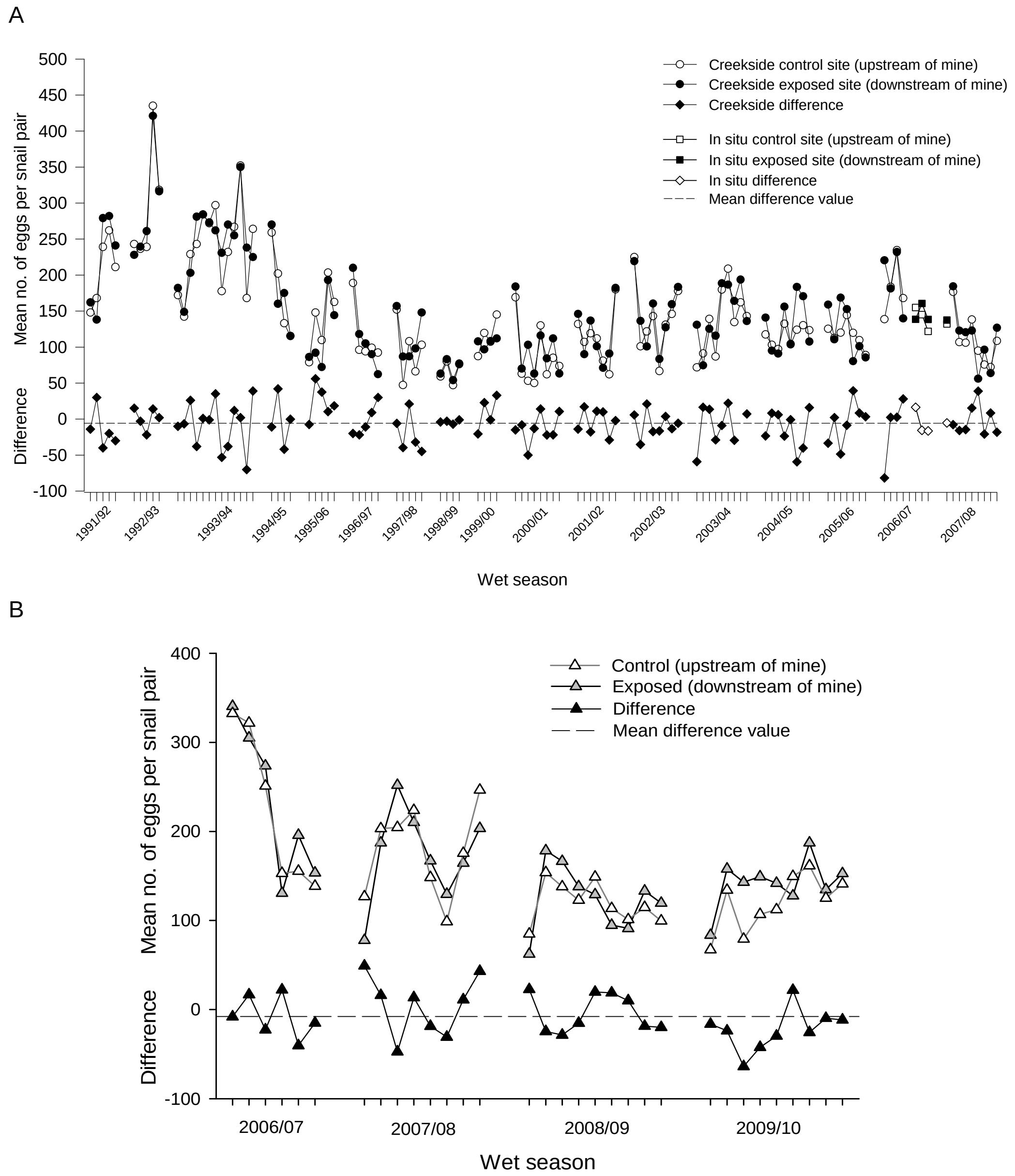

Figure 1 Time-series of snail egg production data from toxicity monitoring tests conducted in Magela Creek using A: (mostly) creekside tests, and B: in situ tests

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.

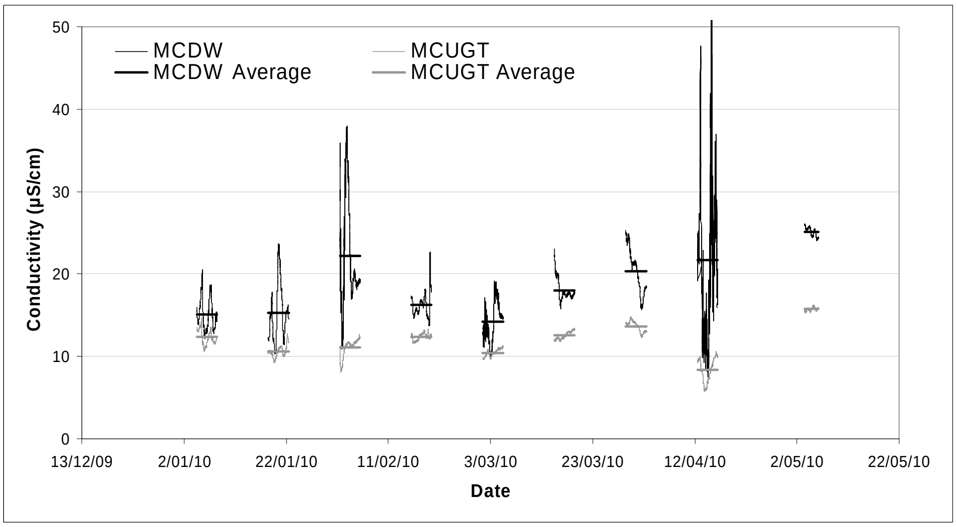

Figure 2 Plots of continuous electrical conductivity (trace) and four-day average values (horizontal line) at monitoring sites in Magela Creek for days in the 2009–10 wet season coinciding with conduct of toxicity monitoring tests

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).

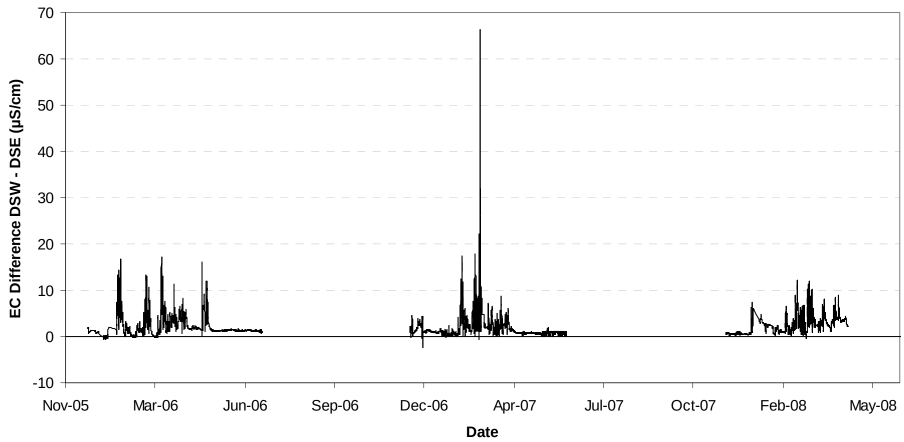

Figure 3 Difference in electrical conductivity (EC) measurements in Magela Creek between west and east sides of the channel downstream of Ranger for the 2005–06, 2006–07 and 2007–08 wet seasons – continuous (hourly) monitoring data (from SSD)

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).

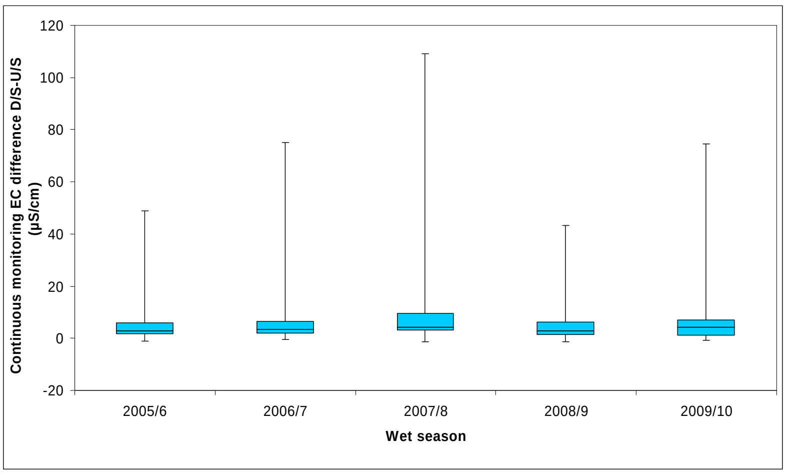

Figure 4 Box plots (median and quartiles) of Electrical conductivity (EC) differences (D/S-U/S) for continuous monitoring sites in Magela Creek

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.

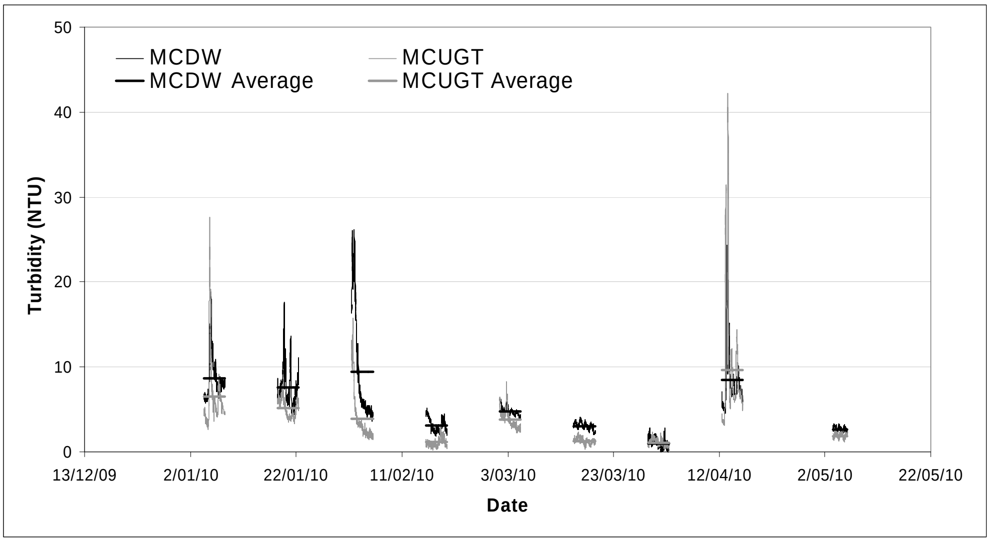

Figure 5 Continuous plots of turbidity (trace) and four-day average values (horizontal line) at monitoring sites in Magela Creek for days in the 2009-10 wet season coinciding with conduct of toxicity monitoring tests. (Note that turbidity traces for test periods 2 and 3 include some error and may be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

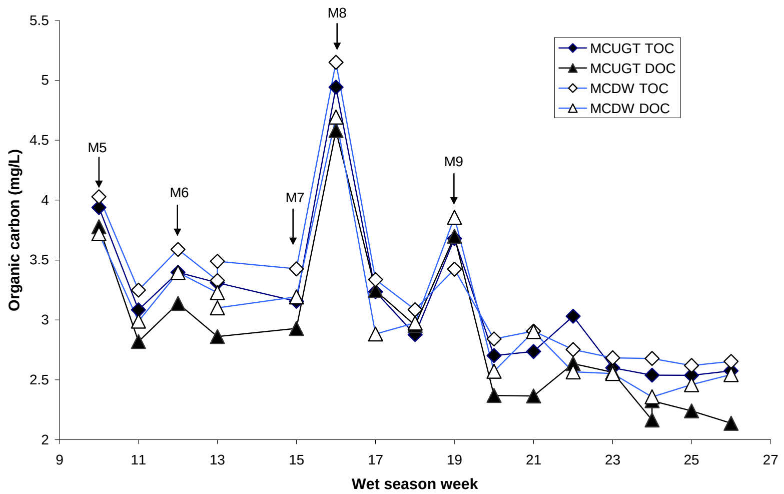

Figure 6 Plots of weekly wet season Total (TOC) and Dissolved (DOC) Organic Carbon in Magela Creek surface waters from grab samples collected from upstream (MCUGT) and downstream (MCDW) sites. Symbols M5 to M9 refer to toxicity monitoring tests 5 to 9 respectively.

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).

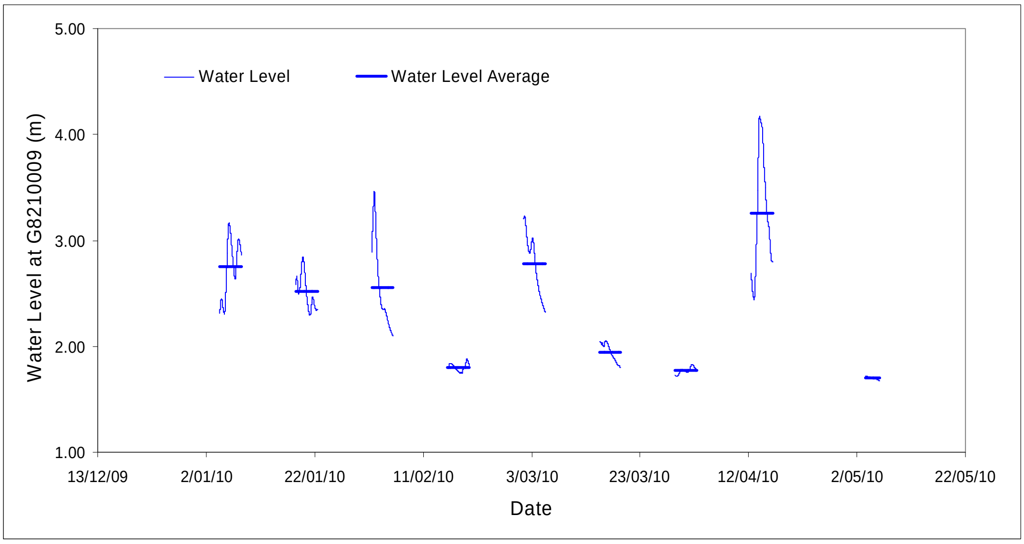

Figure 7 Plots of continuous water levels at Magela Creek downstream (MG009) for days in the 2009–10 wet season coinciding with conduct of toxicity monitoring tests

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.

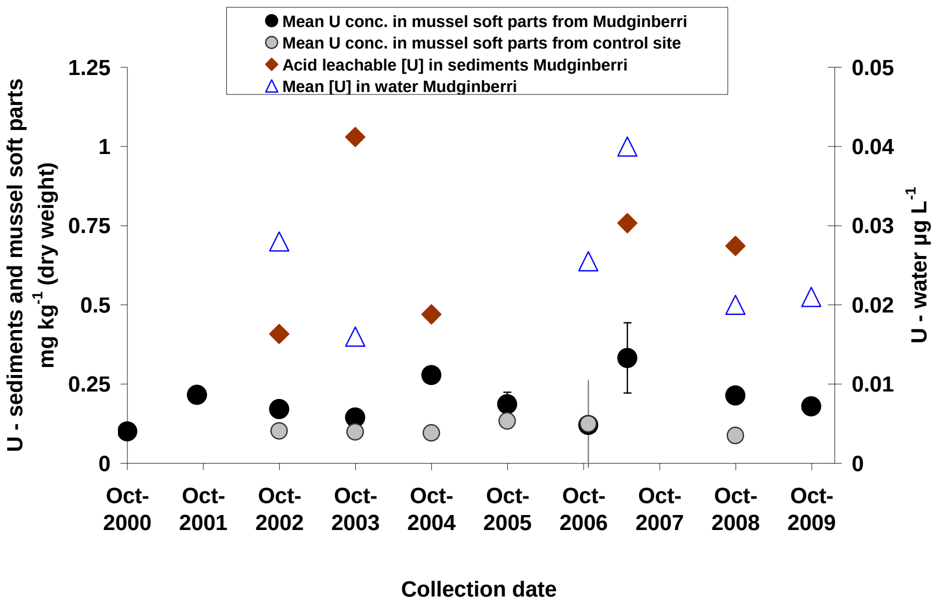

Figure 1 Mean concentrations of U measured in mussel soft-parts, 1M HCl extracts of sediment and water samples collected from Mudginberri Billabong and Sandy Billabong since 2000

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).

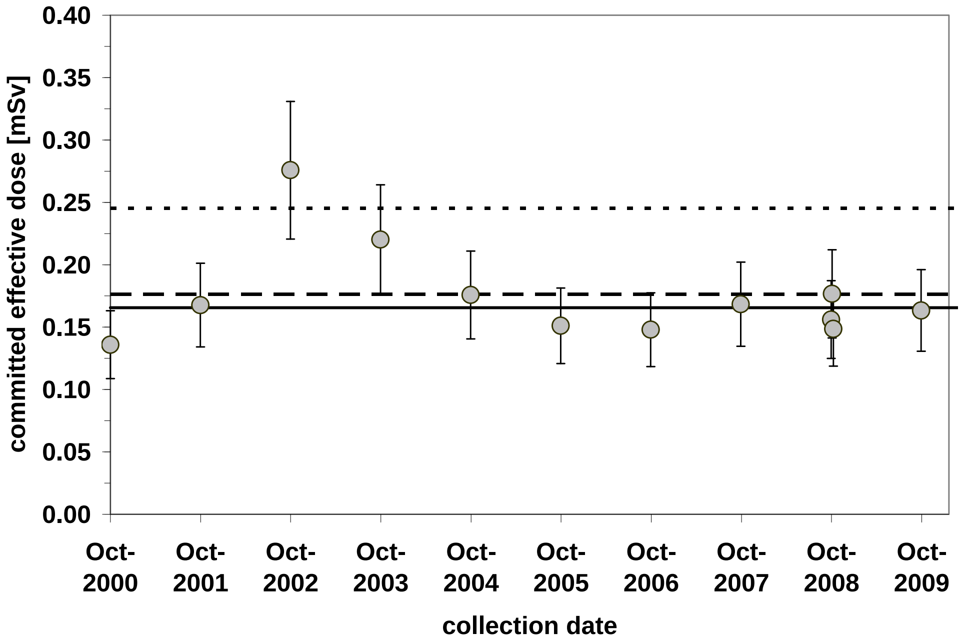

Figure 2 Annual committed effective doses from ^{226}Ra and ^{210}Pb for a 10 year old child eating 2 kg of mussels (wet) collected at Mudginberri Billabong. The median over all collections (solid line), the 80th percentile (dashed line) and 95thpercentile (dotted line) are overlaid.

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.

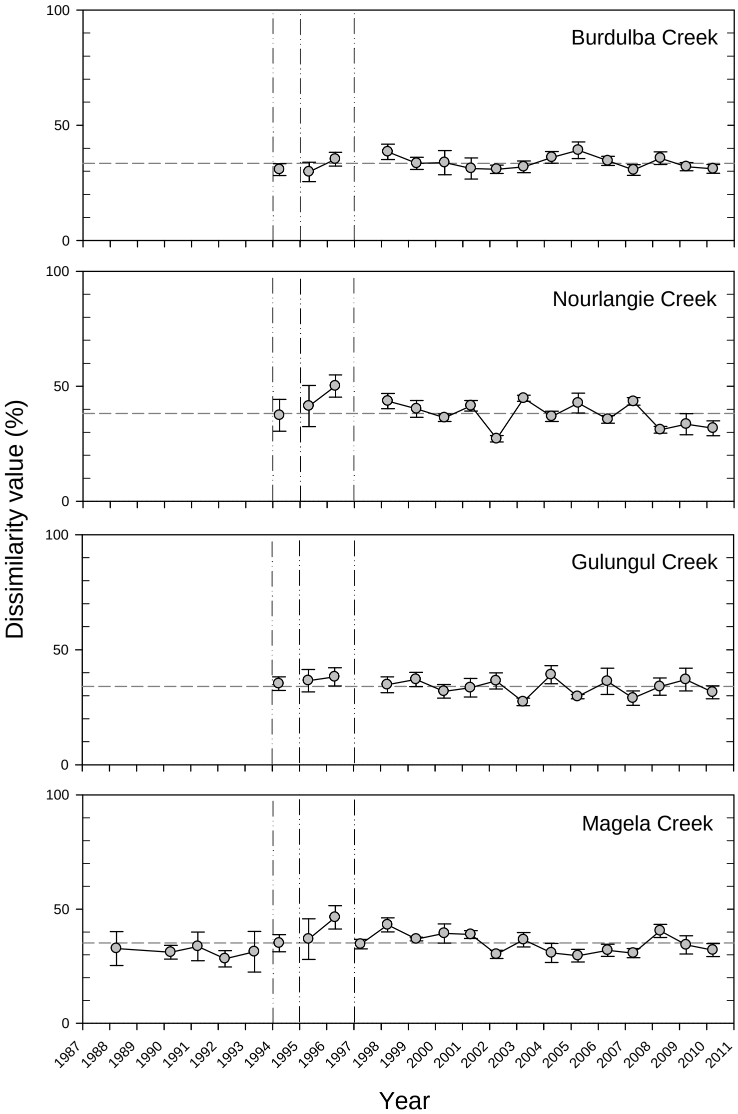

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2010. The dashed vertical lines delineate periods for which a different 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.

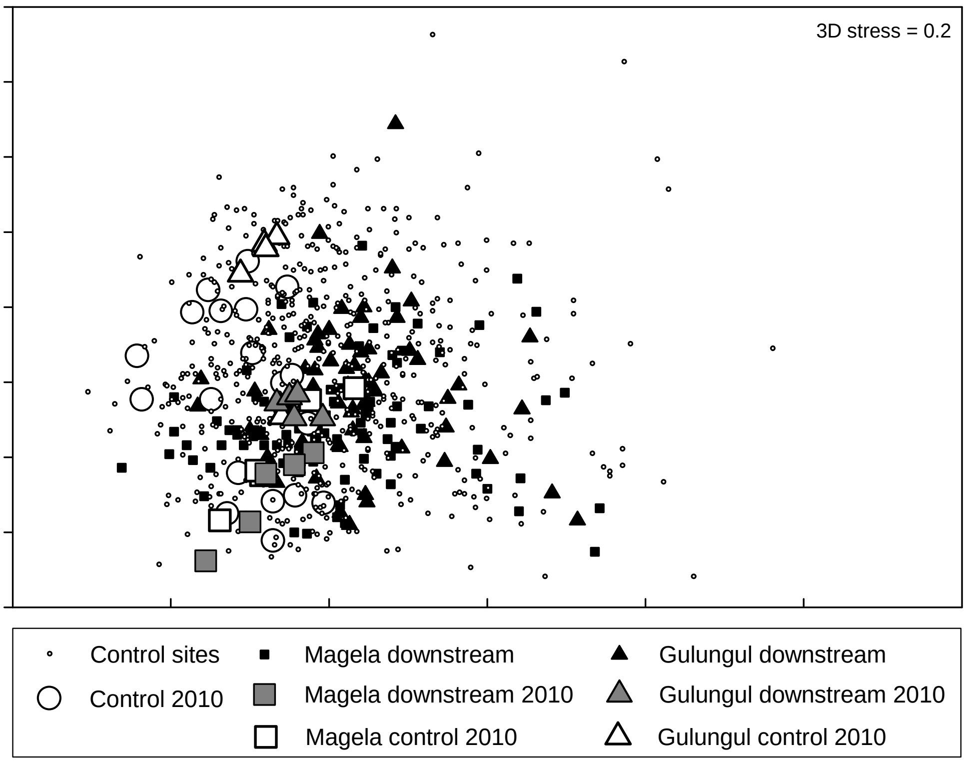

Figure 2 Ordination plot of macroinvertebrate community structure data from sites sampled in several streams in the vicinity of Ranger mine for the period 1988 to 2010. Data from Magela and Gulungul Creeks for 2010 are indicated by the enlarged symbols.

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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.

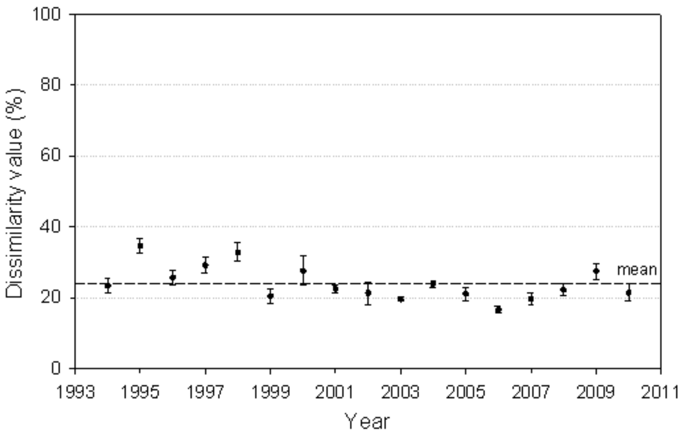

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of the Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

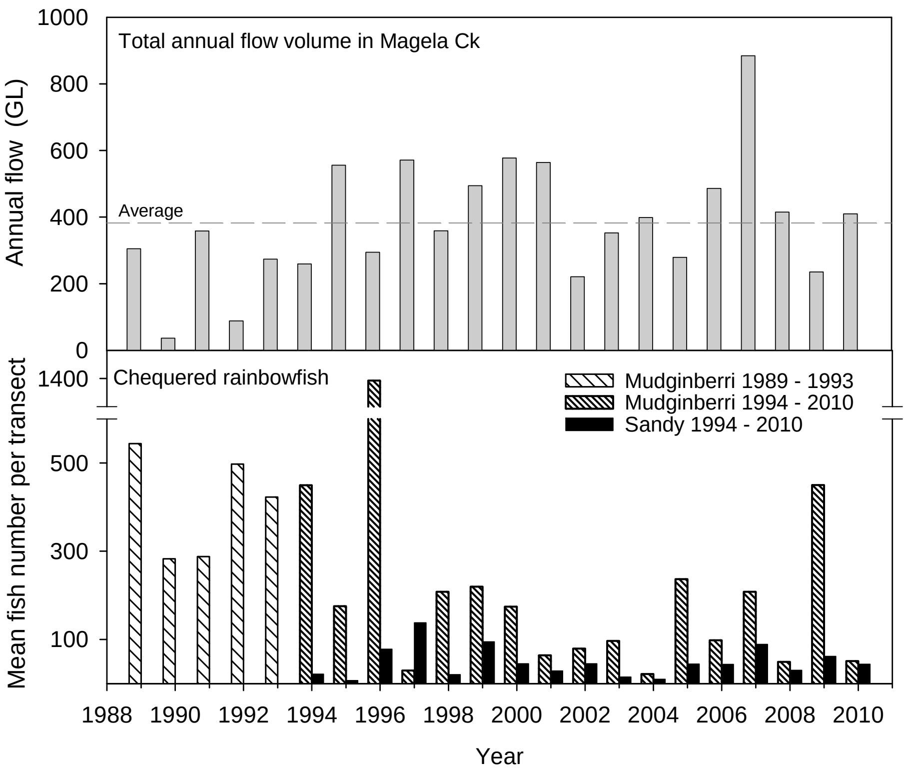

Figure 2 Relative abundance of chequered rainbowfish in Mudginberri and Sandy Billabongs from 1989 to 2010 with associated total discharge in Magela Creek (gauging station G8210009)

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.

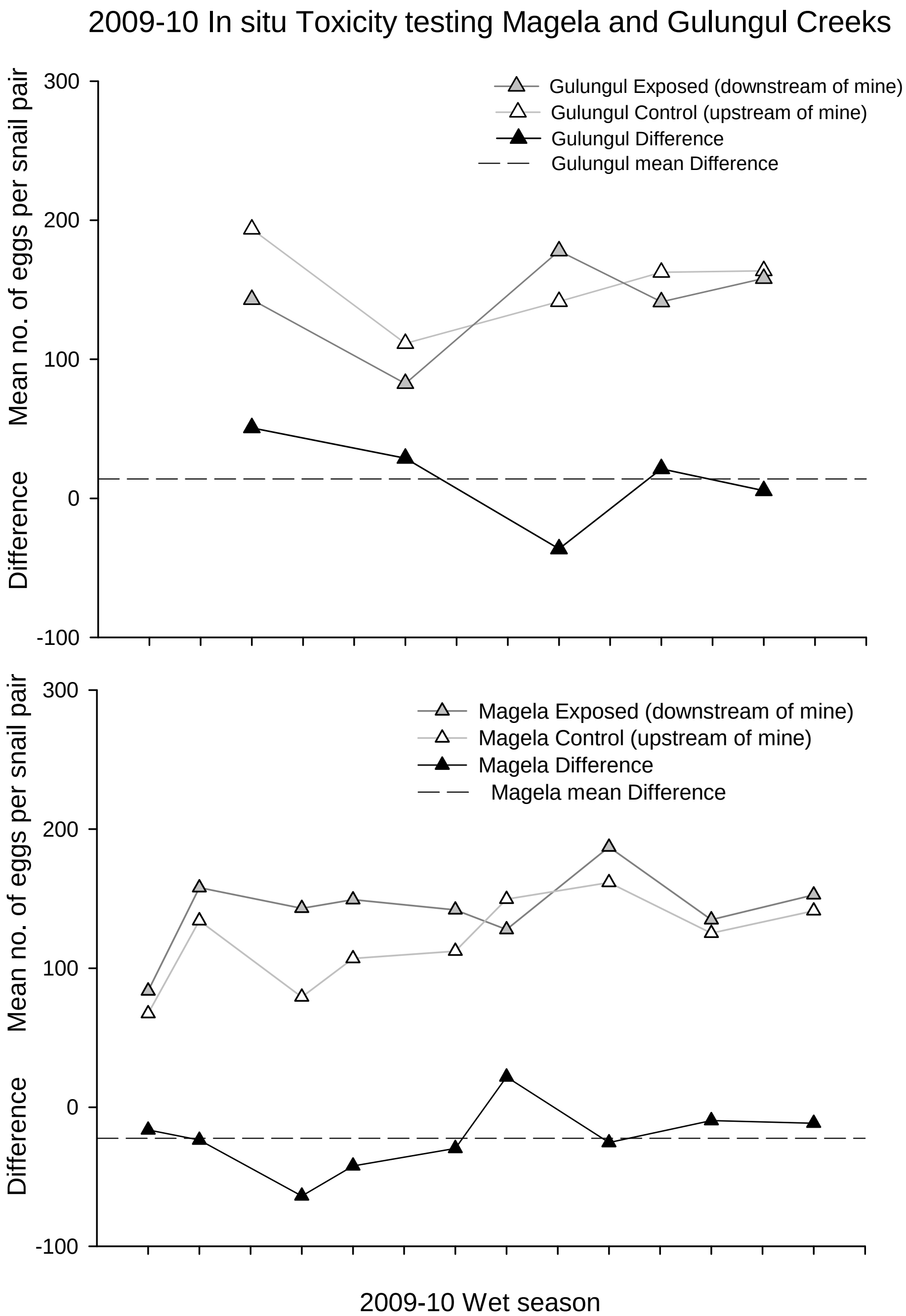

Figure 1 In situ toxicity monitoring results for freshwater snail egg production for Gulungul Creek compared with results from Magela Creek, 2009–10 wet season

Part 2: Ranger – Rehabilitation

Assessing the geomorphic stability of the Ranger trial landform

J Lowry, TJ Coulthard[1], G Staben & A Beraldo

Introduction

The Ranger trial landform, to the north-west of the tailings storage facility (TSF) at Ranger mine, was constructed by Energy Resources of Australia (ERA) during late 2008 and early 2009 (Figure 1). The trial landform covers a total area of 8 hectares and was constructed to allow ERA to test landform design and revegetation strategies to assist in the development of a robust rehabilitation strategy once mining and milling have finished. Specifically, the landform was designed to test two types of potential final cover layers: waste rock alone; and waste rock blended with approximately 30% v/v fine-grained weathered horizon material (laterite).

Figure 1 Location of the elevated trial landform (bottom right of photograph) at Ranger mine

During 2009, the Supervising Scientist Division constructed four erosion plots (30 m x 30 m) on the trial landform surface, with two plots in the area of waste rock, and two in the area of laterite-waste rock blend (Figure 2). The plots were physically isolated from runoff from the rest of the landform area by engineered borders. The erosion plots were constructed to enable:

1 measurement of erosion rates through time to assess effects of different surface treatments and vegetation establishment strategies.

2 generation of input data for predictive computer modelling of the long-term geomorphic behaviour of the proposed landform designs.

3 determination of loads of key contaminants present in the dissolved and fine suspended-sediment fractions available for export from the trial landform via the surface water runoff pathway.

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Full details of the construction of the erosion plots and description of the monitoring infrastructure installed are provided in a companion paper in this volume (Monitoring of erosion and solute loads from the Ranger trial landform – Saynor et al, following paper).

It is intended that the geomorphic stability of the trial landform will be assessed through the use of landform evolution models such as CAESAR (Cellular Automaton Evolutionary Slope and River), which will integrate field data and measurements collected in the erosion plots with digital elevation data that have been collected through survey or laser scanning of the landform.

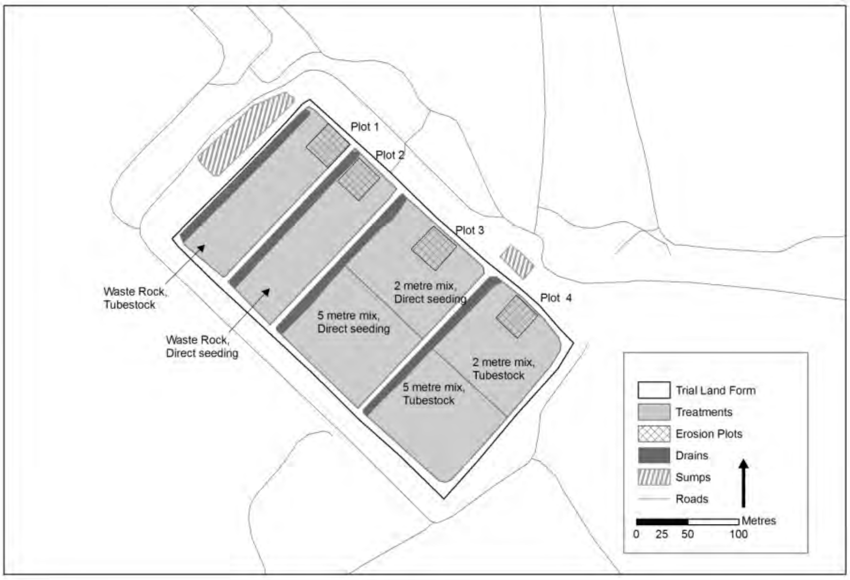

Figure 2 Layout of the plots on the trial landform. Plots 1 and 2 occur on waste rock; plots 3 and 4 have laterite substrate.

Progress/results

Two topographic surveys of the trial landform were completed during 2009–2010. The first manual survey using a Topcon GTS-220 Total Station was undertaken in December 2009 prior to the onset of heavy rains. A total of 1737 elevation points were collected across the surface at approximately 5-metre intervals and used to generate a medium resolution (5 metre) digital elevation model (DEM) (Figure 3).

Figure 3 Digital Elevation Model of trial landform (with overlaid 0.1 m contour lines) produced from the December 2009 survey

During the course of this survey, it was noted that the vegetation growth that will occur over the next few years will progressively compromise line-of-sight or optical surveying methods. Consequently, it will be necessary in the future to employ survey technologies capable of penetrating through vegetation cover to measure ground level (for example, LIDAR – Light Detection and Ranging).

A second survey was undertaken in June 2010 during the early dry season using a Leica ScanStation2 terrestrial laser scanning instrument and differential GPS. In contrast to the earlier manual point survey, the use of the laser scanner enabled both surface elevation data as well as surface features (such as the current status of vegetation communities) to be captured.

Twenty-five scans were made across the landform (Figure 4). Three scans were undertaken within each of the erosion plots, at a scan resolution of 2 cm. A further 13 scans were made across the landform at a coarser resolution of 20 cm.

The very-high resolution DEM generated by the laser scanner was acquired to underpin several components of ***eriss***’s minesite rehabilitation research. Specifically, it provides the digital elevation input data needed for the CAESAR and Siberia landform evolution models that are being used to test the long-term stability of the trial landform against the erosive effects of high intensity rainfall events.

Further high resolution surveys will be required in the event of major erosive activity or other significant changes occurring to the structure of the landform.

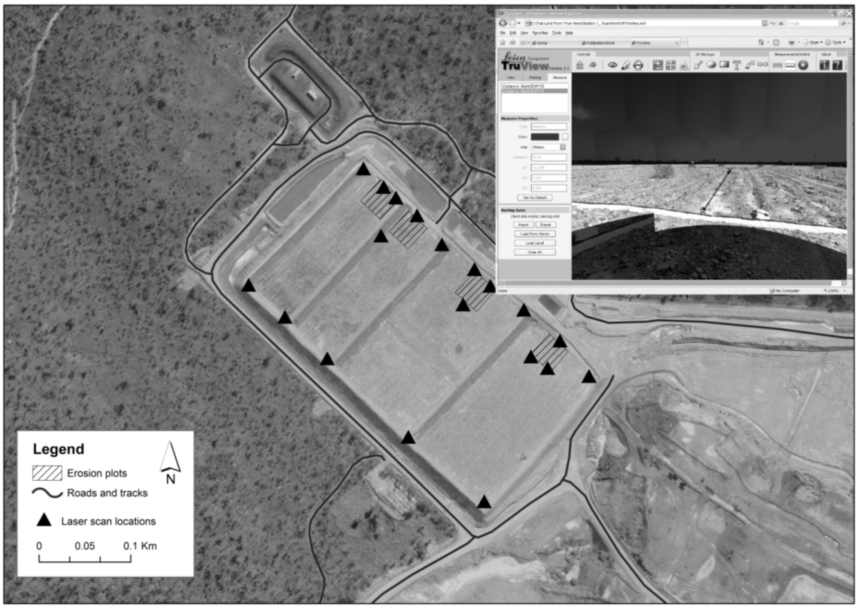

Figure 4 Locations (marked by triangles) of scanning laser instrument. Inset shows an example of the type of composite digital image synthesised from multiple images captured at each scan location.

To date, DEMs with a horizontal resolution of 20 cm have been extracted from the laser scan data sets for Erosion Plot 1 and Erosion Plot 2 (Figure 5). The DEM for Plot 2 spans an elevation range of 1.24 m between the highest and lowest points in the plot, whilst the elevation range for Plot 1 is 0.97m. At this resolution, the rip lines, individual boulders and shallow depressions are clearly visible.

The DEM of each erosion plot was rotated by 137^{o} to ensure that drainage flowed from west to east (a CAESAR pre-requisite), and then processed to ensure that the DEMs were pit-filled and hydrologically corrected. Pit filling was important in order to remove data artefacts, which included remnants of vegetation (peaks) as well as artificial depressions, or sinks from the DEM. The DEMs were hydrologically corrected to ensure that water drained in the correct direction.

In July 2010, the model was modified by Professor Coulthard (the author of CAESAR) to enable it to simulate erosion and deposition at the erosion-plot scale. This represents a significant enhancement to the model. A comparison of the results predicted by CAESAR with the field-measured erosion data will be presented in SSD's next annual report.

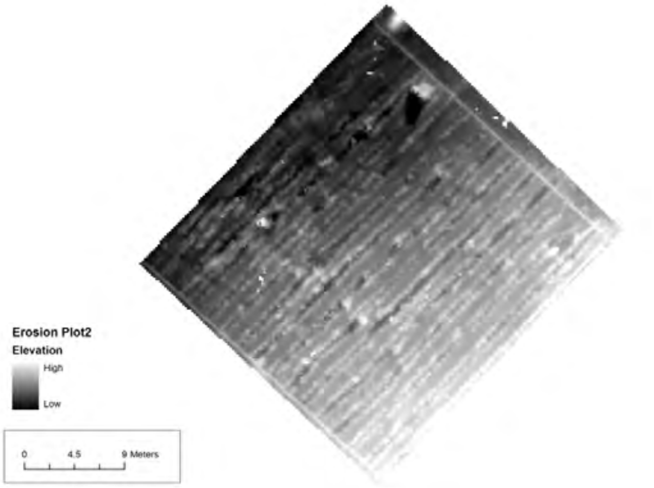

Figure 5 High resolution digital elevation model of erosion plot 2. Lighter colours represent areas of greater elevation. Riplines, boulders (light) and pits (dark) visible on surface.

Steps for completion

The following steps are required to complete this project:

1 Cleaning and collation of 2009–10 wet-season rainfall and sediment load data for the erosion plots on the trial landform,

2 Simulations of the 2009–10 wet-season using the CAESAR model and comparison with the field-measured data,,

3 Extension of modelling to other erosion plots on the trial landform and extrapolation of results to the broader Ranger trial landform

4 Further CAESAR simulations of erosion plots 1 and 2 including simulating the effects of changing slope angle and investigating the effects of extreme rainfall event scenarios.

5 Ongoing publication of results in SSD internal and annual reports, in peer-reviewed journal articles and at national and international conferences

6 Continued collection of field measurements through to 2014 to provide a time series record of changes that occur as the landform evolves and vegetation develops.

Reference

Saynor MJ, Evans KG & Lu P 2009. Erosion studies of the Ranger revegetation trial plot area. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 125–195.

Monitoring of erosion and solute loads from the Ranger trial landform

M Saynor, K Turner, K Tayler & R Houghton

Introduction

A trial landform of approximately 200 m x 400 m was constructed during late 2008 and early 2009 by Energy Resources of Australia Ltd (ERA) adjacent to the north-western wall of the tailings storage facility (TSF) at Ranger mine (Figure 1). The trial landform will be used to test landform designs and revegetation strategies to assist ERA develop a robust rehabilitation strategy for the site once mining and milling have finished.

The landform was designed to test two types of potential final cover layers:

1 Waste rock alone.

2 Waste rock blended with approximately 30% v/v fine-grained weathered horizon material (laterite).

The landform is divided into six treatment areas (Figure 2). Each treatment was designed to test different planting methods and substrate types as follows:

1 Tube stock planted in waste rock material
2 Direct seeded in waste rock material
3 Direct seeded in waste rock mixed with laterite to a depth of 2 m
4 Direct seeded in waste rock mixed with laterite to a depth of 5 m
5 Tube stock planted in waste rock mixed with laterite material to a depth of 2 m
6 Tube stock planted in waste rock mixed with laterite material to a depth of 5 m

Figure 1 Location of the elevated trial landform at the Ranger mine

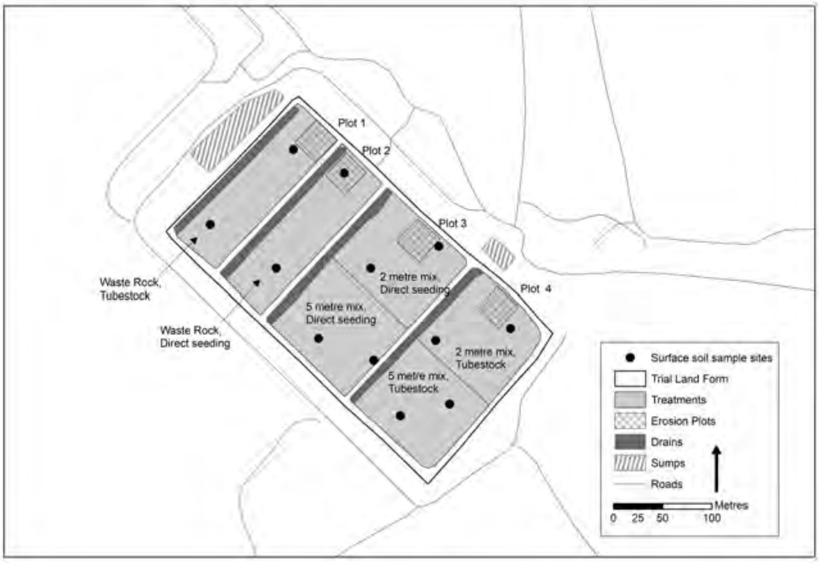

Figure 2 Layout of the plots on the trial landform

Methods

Surface sediment collection

During the 2009 dry season, surface samples were collected by spade to a maximum depth of 10 cm from 12 locations over the trial landform surface to characterise the particle size distribution (for locations see Figure 2). For all 12 samples, more than half of each sample (by weight) (53–78%) was larger than 2.0 mm in diameter showing the influence of the waste rock on the composition of the cover treatments. The fraction greater than 2.0 mm from surface soil on the natural surrounding Koolpinyah surface is a generally no greater than 10%.

Erosion plots

Four erosion plots (30 m x 30 m) (location marked by cross hatched small squares on Figure 2) were constructed on the landform surface and physically isolated by engineered borders from runoff from the rest of the area. Plots 1 and 2 are located in the waste rock only surface treatment, whereas plots 3 and 4 were located in waste rock blended with approximately 30% v/v fine-grained weathered horizon material.

Half-section 300 mm diameter U-PVC stormwater pipes were placed at the down slope ends of the plots to catch runoff and channel it through rectangular broad-crested (RBC) flumes (Figure 3) where rainfall event triggered discharge is measured. A reservoir (stilling basin) is located upstream of the inlet to each flume to trap coarser material eroded from the plot. Plot construction is described in detail in Saynor et al (2009).

The outlet of each erosion plot is instrumented with the following sensors:

- pressure transducer and shaft encoder to measure stage height
- a turbidity probe
- electrical conductivity probes located at the inlet to the stilling well and in the entry to the flume to provide a measure of the concentrations of dissolved salts in the runoff
- an automatic water sampler to collect event based samples
- a data logger with mobile phone telemetry connection.

Figure 3 Runoff through flume on trial landform erosion plot 3 during a storm event

A rain gauge was also installed at the downstream end of each plot near the instrument shelter. Data acquired during the 2009–10 wet season were downloaded daily by mobile phone access and then stored in the hydrological database *Hydstra*.

During the 2009–10 wet season runoff, turbidity (surrogate of fine suspended sediment), bedload (coarser material deposited in the stilling basin) and EC (surrogate of water quality) were measured. The first rainfall event of 26 mm occurred on 23/9/09 and the last significant rainfall event of 21.6 mm occurred on 28/8/10. The total rainfall for the 2009–10 wet season (averaged across the four plots) was 1505 mm.[1] The hydrologic water year runs from beginning of September until the end of August (Chiew & Wang 1999, Moliere et al 2002).

During rainfall induced runoff events water samples were collected by automatic water samplers triggered by pre-programmed increases in stage height, turbidity and EC. The trial landform was visited once a week to collect the water samples and the bedload. This task was shared between staff from SSD and ERA, with the allocation of staff resources and workplan defined in a formal memorandum of understanding between SSD and ERA. SSD was responsible for processing and analysis of all of the samples collected for the sediment transport component of the project; ERA was responsible for chemical analysis of the water samples.

Results

Sediment transport

Fine suspended sediment

Turbidity provides a measure of the concentration of fine suspended sediment. It is this fine material that is of most immediate relevance from the perspective of the potential for downstream environmental impact of material eroded from a newly constructed mine landform.

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[1] The rainfall values presented here are slightly higher than those presented in the 2009–10 Supervising Scientist Annual Report as there was un-seasonal rain in August which forms part of the 2009–10 hydrologic water year.

An example of concurrent typical turbidity events occurring across each of the four erosion plots in response to a rainfall event is shown in Figure 4 below. The magnitude of the pulses for the waste rock plots (plots 1 and 2) are generally similar to one another and lower than the pulses observed for the mixed waste rock and laterite plots (plots 3 & 4). Throughout the season, the turbidity measured at plot 3 was consistently higher than that measured at plot 4.

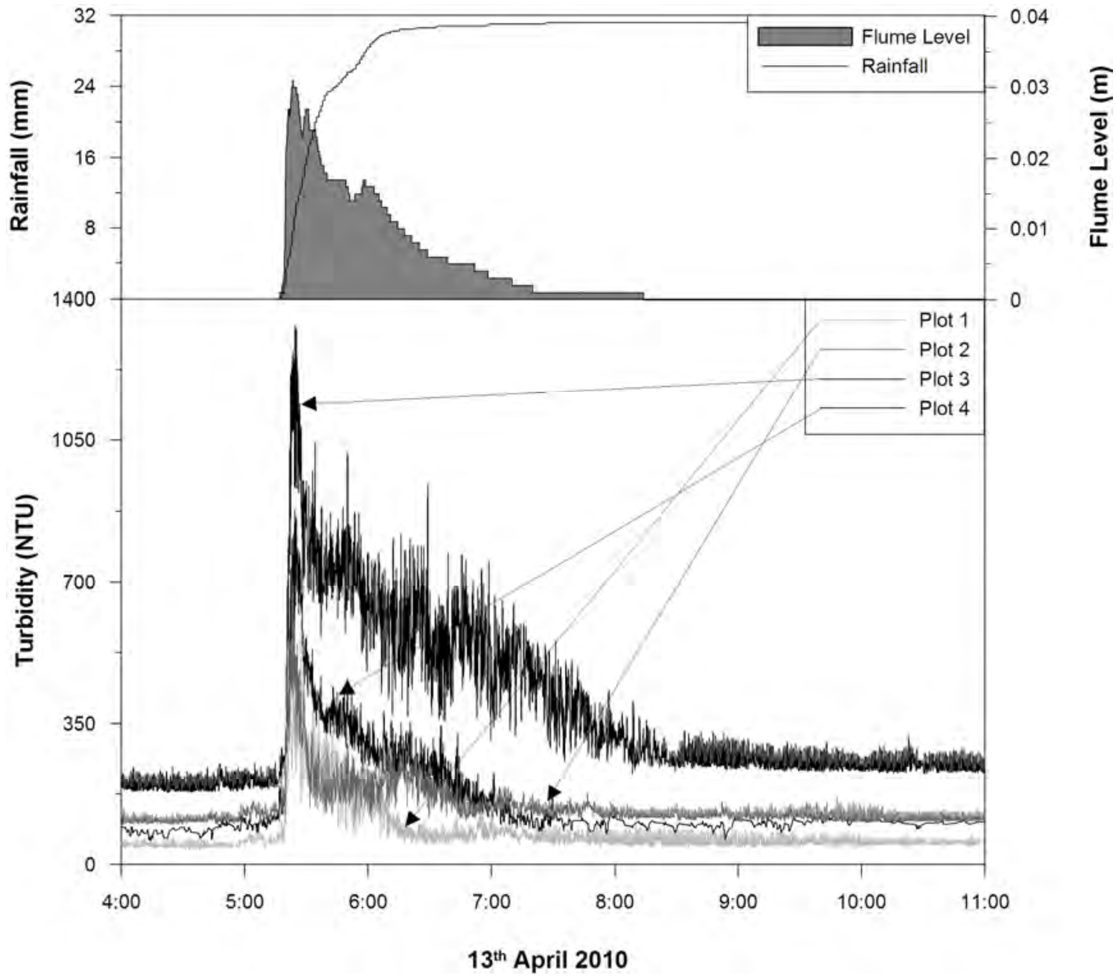

Figure 4 Rainfall induced turbidity events occurring between 4 and 11am on 13 April 2010. Top panel shows the cumulative rainfall and flume water level (surrogate of flow) for plot 2. The bottom panel displays the continuous turbidity data from each of the four erosion plots.

Water samples were collected during rainfall events using autosamplers activated using a combination of pre-programmed stage height, EC and turbidity values. All samples triggered by turbidity were analysed for total suspended sediment (TSS) concentration (sediment fraction between 63 µm and 0.45 µm). The TSS concentration was determined by firstly passing the water sample through a 63 µm sieve and then filtering a standard volume through a 0.45 µm filter. The weight of the dried residue on the filter paper was then measured. The TSS data will be used to define the relationship between TSS and turbidity measured in situ, allowing estimation of continuous TSS concentration from the continuous turbidity data.

The TSS is the most readily transportable fraction of sediment and is a key indicator of landform surface erosion rates. Selected TSS samples will be analysed for associated trace metal concentrations (including uranium and radium) to derive the loads of sediment-associated contaminants transported from each of the erosion plots during the 2009–10 wet season.

Bedload

The coarser bedload material is deposited in both the half pipe defining the downslope boundary of the plot and in the stilling basin upstream of the flume. The total amount of bedload collected from each plot over the wet season is shown in Table 1. Similar amounts of bedload material were washed from each of the plots, with no systematic difference between the two surface treatments.

Table 1 Total bedload collected for 2009–10 wet season[1]

Erosion plot	Basin (kg)	Half-pipe (kg)	Total (kg)
EP1	24.5	72.7	97.2
EP2	9.7	119.3	129.0
EP3	16.3	87.1	103.4
EP4	65.3	58.1	123.4

1 The bedload values presented here are slightly higher than those presented in the 2009-2010 Supervising Scientist Annual Report as there was un-seasonal rain in August which caused runoff from the erosion plots.

The particle size distributions measured for bedload samples collected on 17/03/2010 and 15/04/2010 are provided in Table 2 and show the influence of rainfall event magnitude. Sieving was used for size classification above 63 µm. The hydrometer (gravity settling) method was used for more detailed classification (not shown here) of the less than 63 µm fraction.

Table 2 Bedload particle size distribution data (dry weight basis) for samples collected on 17 March 2010 and 15 April 2010

Sample erosion plot	Sample date	Sample mass (kg)	% > 2.00 mm	% < 2.00 mm	
				% > 0.063 mm	% < 0.063 mm
EP1	17/03/2010	1.5	18.7	73.6	7.7
EP2	17/03/2010	1.9	17.9	59.7	22.4
EP3	17/03/2010	1.3	28.2	61.0	10.8
EP4	17/03/2010	1.5	15.0	75.1	9.9
EP1	15/04/2010	14.4	33.3	61.7	5.0
EP2	15/04/2010	15.2	24.6	63.7	11.7
EP3	15/04/2010	12.9	53.6	44.5	1.9
EP4	15/04/2010	12.4	45.2	52.2	2.6

The rainfall events that produced the amounts of bedload reported in Table 2 are shown in Table 3. The bedload collected on 17/3/10 resulted from 49 mm of rainfall over 4 events and was correspondingly much lower in mass than the bedload collected from 15/4/10 which was the result of 254 mm of rainfall over 8 events.

Table 3 Rainfall events during the week prior to bedload collection

Sample date	Total rain (mm)	No of events	Event 1 (mm)	Event 2 (mm)	Event 3 (mm)	Event 4 (mm)	Event 5 (mm)	Event 6 (mm)	Event 7 (mm)	Event 8 (mm)
17/3/10	49	4	5	16	9	15				
15/4/10	254	8	58	5	11	47	30	41	25	26

Solute transport

EC sensors were installed at the entrance and the exit of the sediment settling basin at each of the erosion plots. The information from both of the sensors was used to derive event-based EC data for each site over the 2009–10 wet season. The behaviour of EC observed over an event will be determined by the condition of the basin preceding the rainfall. Two possible conditions apply for this system:

1. The basin was empty and clean prior to rainfall, in which case the EC is indicating the composition of surface runoff throughout the event.
2. The basin was full prior to rainfall, in which case the EC trace measured at the exit to the basin could be impacted by 'stale' water that has remained in the basin between rainfall events.

Condition 1 events give a clear indication of the surface runoff water quality. Condition 2 events are confounded due to the mixing of the surface runoff with 'stale' water in the basin that has accumulated from a varying number of antecedent events. While the majority of events occurring throughout the wet season occurred under condition 2, the potential confounding caused by the 'stale' water can be removed by comparison of the EC values measured at the entrance and the exit of the basin. The time at which the two EC readings converge will indicate when complete flushing of the 'stale' water has occurred. Detailed analysis of the time series EC data for the condition 2 events is still in progress. Consequently the results reported here will focus on condition 1 events.

Thirteen condition 1 events occurred during the 2009–10 wet season. However, the intensity of the rainfall and associated runoff volume for the majority of these events was low, with only five of the 13 events falling in the upper 50th percentile of rainfall volume and intensity for the season. As a result, these events only generated a small volume of flow through the flumes and were generally short lived. Figure 5 shows summary statistics describing the peak (maximum) EC values recorded for each of the 13 events for plots 2 and 4, representing the waste rock and waste rock mixed with laterite, respectively. The box and whisker plot shows that the medians and general distribution of the peak EC values for each plot are similar. The scatter plot shows that the response of peak EC values as a function of total rainfall (up to 35mm only) for each event are similar for both plots, indicating that the total amount of solutes derived from both treatments are similar for condition 1 events.

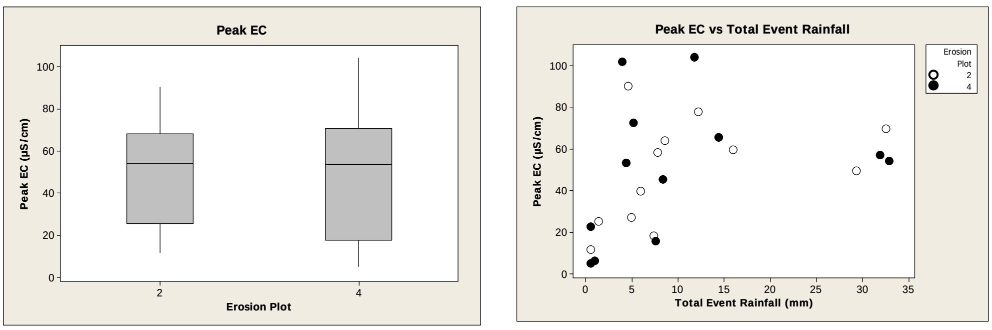

Figure 5 Box plot summarising the mean, maximum, minimum, third quartile and first quartiles of the maximum first flush EC values; and scatter plot of the maximum first flush EC values and total event rainfall

Water samples were collected for chemical analysis from each of the erosion plots using autosamplers, which were activated using a combination of stage height and EC triggers. The

EC-triggered samples were analysed by ERA in its on-site laboratory for a suite of trace elements and major ions. The results obtained for filtered Mg, SO_4 and U only are presented here (Figure 6) since these solutes are the most relevant for potential environmental impact from the site. The box plots in Figure 6 show the concentration means and ranges measured for each of the three solutes in the water from each of the four plots.

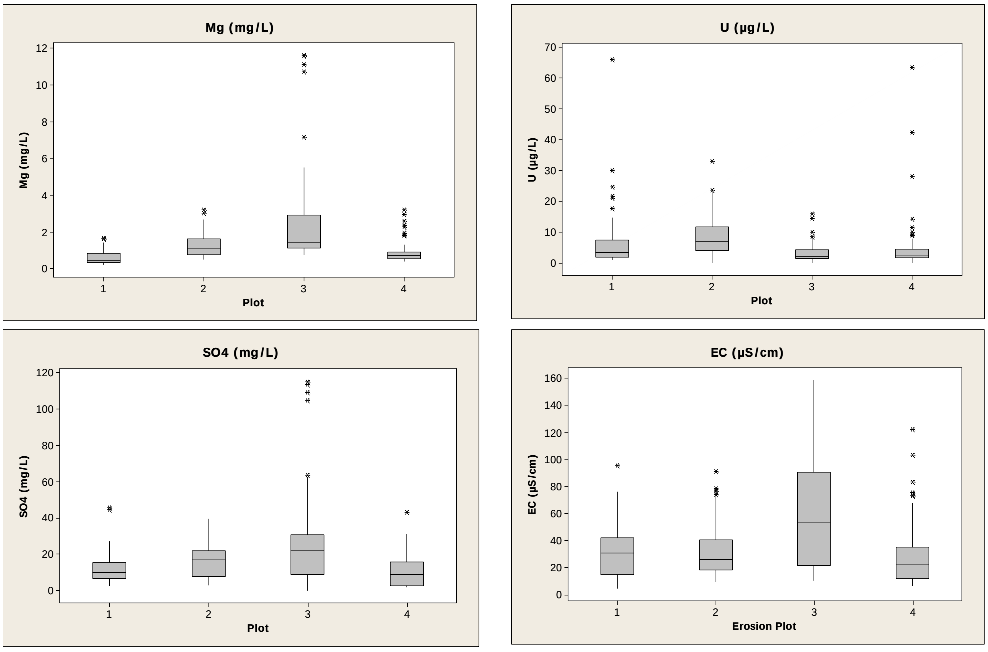

Figure 6 Box plots showing the mean, maximum, minimum, third quartile and first quartile values of Mg, SO4 and U concentration and EC measured in the water samples collected from each plot over the 2009–10 wet season. The number of data points (n) for erosion plots 1, 2, 3 and 4 are 54, 76, 67 and 86, respectively.

The information in Figure 6 shows that:

- EC exhibits similar behaviour to Mg and SO_4, indicating that these ions are major contributors to the EC of the surface runoff from each plot
- Plots one, two and four all show similar water concentration ranges for both Mg and SO_4 (and hence EC). However, plot three has a broader range and maximum values at least double that of each of the other plots
- The highest concentrations of U were measured runoff from plots one and four, noting that the majority of U concentrations were less than 30 µg/L and that the means, except for plot two, were less than 6 µg/L, which is the current ecotoxicologically derived limit for U in Magela Creek.

Being a composite of all of the data, the box plot summaries do not demonstrate the dynamic range of concentrations that occur through a rainfall event. To do this, individual events need to be analysed. Figures 7 and 8 show examples of the time series concentrations of Mg and U measured through two rainfall events that produced sustained flow through the installed flumes.

The concentrations of Mg are very similar between the two plots for these events. There is a difference in EC between 0530 and 0700h in Figure 7. However, this particular event represents the low end of the EC range (0–700 µS/cm) measured over the wet season so the effect on solute load of the differences observed between the two plots for this event is low. Generally, further data analysis is required to statistically define the significance of such variation.

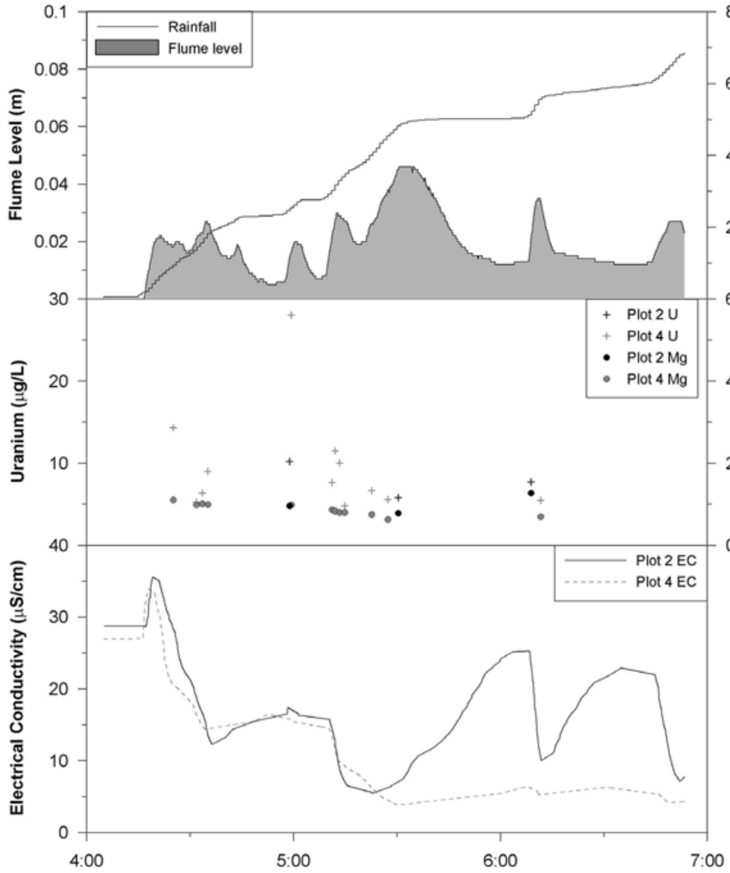

Figure 7 Rainfall event occurring between 4 and 7 am on 1 February 2010. Rainfall, EC, Mg, U and flume level from plots two and four are shown for comparison between wasterock and laterite treatments.

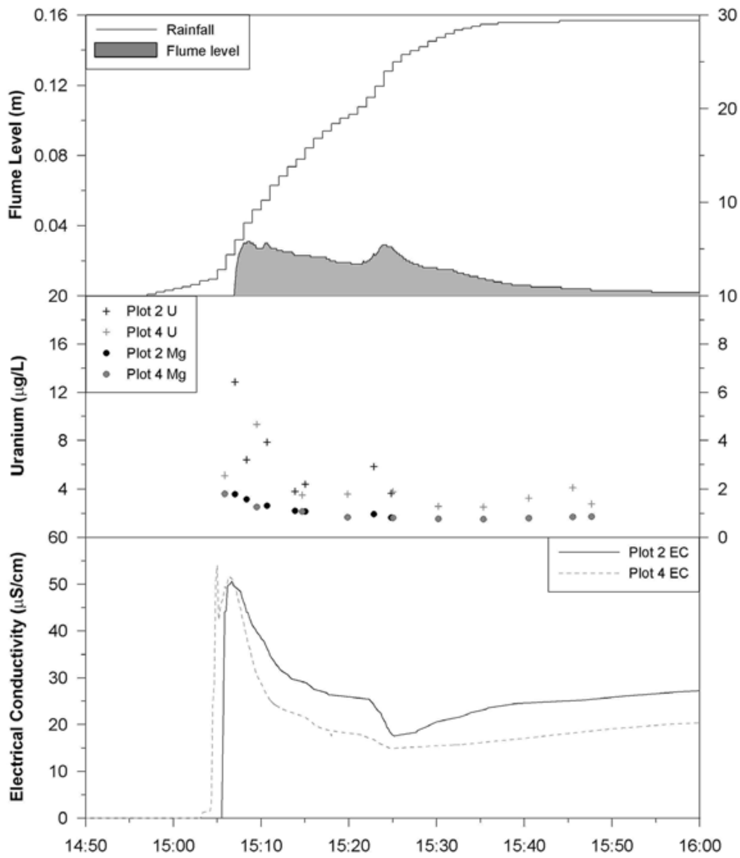

Figure 8 Rainfall event occurring between 3 and 4 pm on 23 March 2010. Rainfall, EC, Mg, U and flume level from plots two and four are shown for comparison between waste rock and laterite treatments.

Future work

Considerable resources are being devoted to processing, collating and analysing the large amounts of data produced from the trial landform during the 2009–10 wet season. Examples have been provided in this report of the wide range of information that is being produced by the project. The findings will be used to inform analysis of the suitability of options for the design and revegetation of the final rehabilitated Ranger site.

During Q1 and Q2 of the 2010–11 financial year it is anticipated that runoff loads of solutes, suspended sediment and bedload material will be derived for each of the plots, enabling quantitative comparison of the behaviours of the two types of surface treatments. These results will be documented in the next ***eriss*** Research Summary.

The scope of the trial landform monitoring program for the 2010–11 wet season will be refined using the findings from the 2009–10 season, with more selective sampling and analysis of the runoff streams.

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Assess the impact of extreme rainfall events on Ranger rehabilitated landform geomorphic stability using the CAESAR landscape evolution model

JBC Lowry, TJ Coulthard[1] & GR Hancock[2]

Introduction

The ability to predict the stability of post-mining landscapes through time scales ranging from decades to thousands of years is a critical element in the assessment of the rehabilitation practices of uranium mines. Erosion may lead to increased sediment loads and the transport of other mine-related contaminants off site and into downstream waterways.

Computer modelling is an important tool to assist in the understanding of the interactions between geomorphology and erosion and hydrologic process because of its ability to explore the sensitivity of a mine landform to a wide range of design and climate variables. Simulations of the impact of extreme rainfall events on a conceptual rehabilitated landform of the Ranger minesite using the CAESAR landscape evolution model have been previously reported at the 22nd ARRTC meeting in October 2008 and in Evans et al (2010).

During 2009–10 significant progress was made in collaboration with Professor Coulthard to enhance the capacity of the CAESAR model to simulate erosion in areas with different surface conditions/soil grain size distributions.

Progress/results

During the 2009–10 year significant progress was made in enhancing the capacity of the CAESAR model to simulate the geomorphic evolution of mining landforms composed of multiple surface cover treatments or soil grainsize distributions. Previously CAESAR had only been able to model surfaces with a uniform grainsize/surface treatment. As a result of enhancements made to the model by Professor Tom Coulthard (the author of CAESAR) during his October 2009 visit to Darwin, up to five different surface cover treatments are now able to be used in a single simulation.

For the purposes of this study, the CAESAR model was applied to a conceptual rehabilitated landform (Figure 1) of the Ranger mine supplied by ERA. An outline of current mine infrastructure has been overlain for reference over the surface of the conceptual landform in Figure 1. The surface has low relief (<50 m) and comprises a north-south ridge between backfilled Pit 3 and the Tailings Pond, and an east-west ridge between backfilled Pit 1 and the southern wall of the Tailings Pond.

The digital elevation model of the conceptual landform produced by ERA was supplied in raster format with a 20 m cell size. The cell size defines the minimum size of features that are

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represented on the surface of the landform, and hence that are able to be included and tracked through the modelling process.

The dataset was processed to ensure that the DEMs were pit-filled and hydrologically corrected. Pit filling was important in order to remove data artefacts, such as artificial depressions, or sinks from the DEM. The DEMs were hydrologically corrected to ensure that water drained in the correct direction. Drainage features were generated from the dataset using hydrological modelling tools in the ArcGIS environment. Due to the limited resolution of the DEM, the drainage features were assigned a minimum width of 40 metres (ie two 20 m 'pixels').

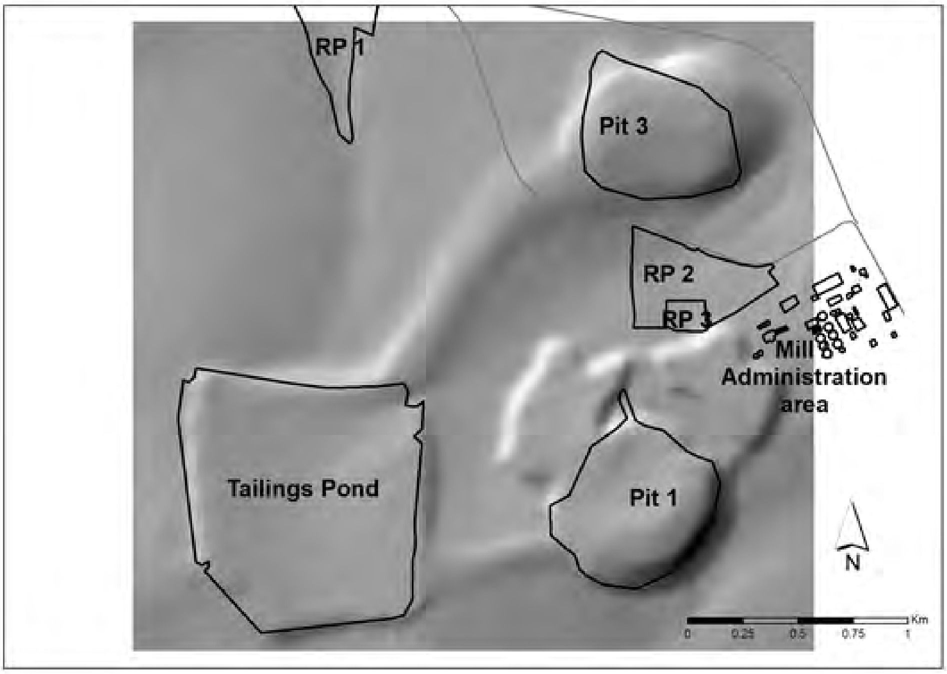

Figure 1 Conceptual rehabilitated landform utilised by CAESAR. Existing infrastructure is superimposed for reference.

Initial test simulations used particle size distribution (PSD) data (Figure 2) obtained from field measurements of four different types of surfaces relevant to the Ranger situation. These PSDs represent a natural soil surface; waste rock and waste rock plus laterite cover treatments used on the Ranger trial landform (see Saynor et al this volume, for a description of the trial landform); and a 50–150 mm mix representing coarse-grained material of between 50–150 mm diameter that would be used to line drains conveying runoff from the landform. The particle size distribution of a natural surface was represented by soil collected from an undisturbed vegetated area of the Gulungul Creek catchment (see Map 2 for the location of this creek). In this context it should be noted that it is only the PSD of the extracted soil that is being input into the model, and not the actual landscape properties (ie an intact soil profile stabilised by vegetative cover).

The final key inputs required for CAESAR are rainfall event data (mm/hr). Rainfall data collected at the Jabiru airport between 1972 and 2006 and aggregated into a 21 year continuous sequence were used for the simulations reported here.

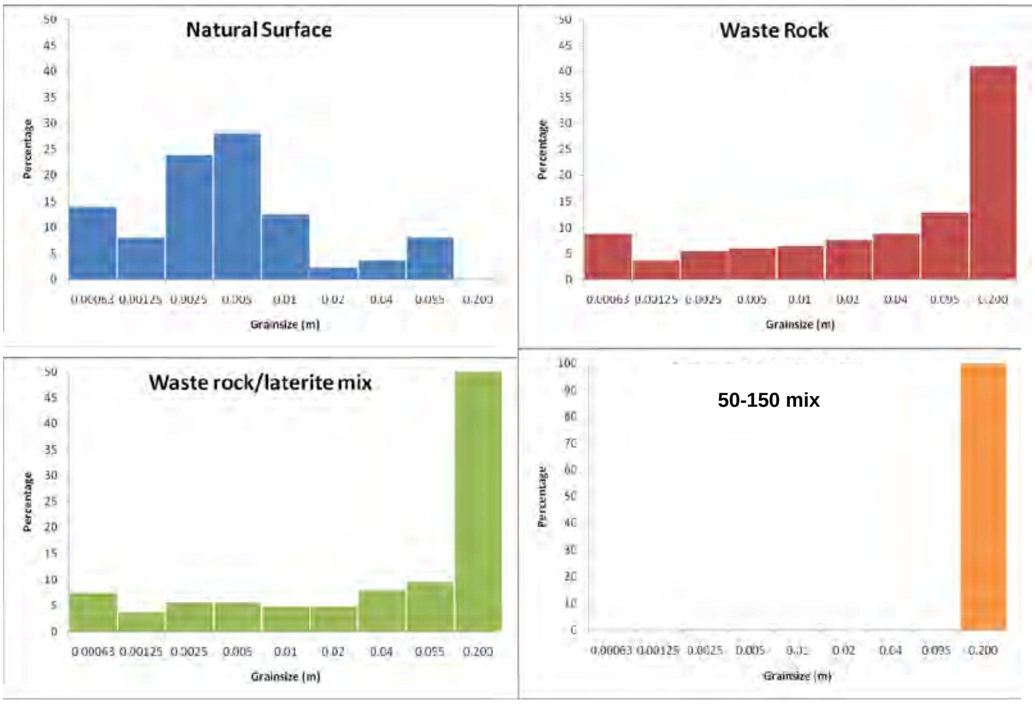

Figure 2 Particle size distribution for the different surface conditions used in CAESAR simulations

Three scenarios, each incorporating two surface treatments, were simulated using the particle size data for the different surface conditions. The drainage lines were fixed as the 50–150 type for all simulations with the broad area of the landform surface comprising the second treatment 'natural', waste rock, waste rock plus laterite). As stated above the drainage lines were assigned a uniform width of 40 m, with the PSD characteristics of the 50–150 mix. The areas representing each of the different surface types are shown in Figure 3.

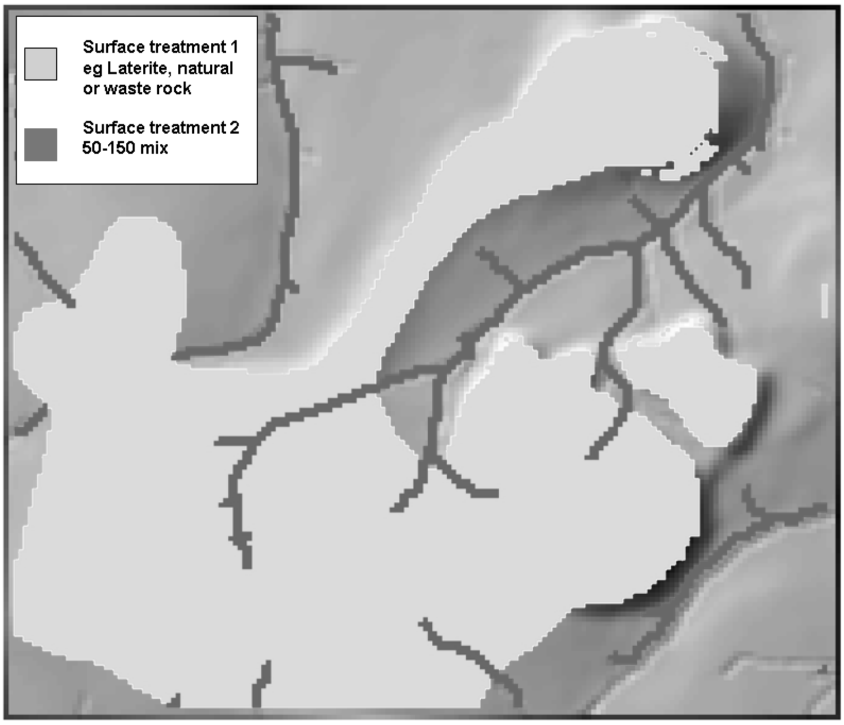

Figure 3 Areas of surface treatments modelled in CAESAR

As shown in Table 1, the initial simulations run for each of the treatments indicate very similar outputs of eroded sediment for the areas composed of waste rock and waste rock mixed with laterite, reflecting the quite similar particle size distribution of these two cases. In contrast, areas simulating the PSD of the natural surface were predicted to more erodable. This was to be expected since the smaller (more erodable) grainsizes comprised a bigger proportion of the material. As stated above this is not meant to be a simulation of the erodability of a natural land surface in the area of the mine since the material is not in its environmental context. Future work will investigate the effect of stabilisation produced by self-armouring processes and the development of vegetation.

Table 1 Mean annual sediment yield produced from different surface treatments

Surface material	Sediment yield	
Natural	22086 m^3	2.212 mm / yr
Waste rock	7644 m^3	0.766 mm / yr
Laterite mix	8144 m^3	0.816 mm / yr

A uniform width of 40 m, covered with the 50–150 mix, was assumed for the constructed drainage lines used for each of the simulations. However, the modelling predicted drainage channel widening of up to 160 m. The extent of this change indicates that very close attention will need to be paid to the engineering design specification and construction of these drainage features such that they will be able to accommodate the volume of flow coming from the surfaces of the landform, Attention will also need to be paid to the potential for at least temporary infill of the channels (and hence reduction in capacity) as a result of the erosion and in-channel deposition of sediment from the landform.

The different results produced by the initial simulations clearly demonstrate that is it is possible for CAESAR to model two surface conditions at the same time. Further work is required to create scenarios which realistically depict the configuration and different surface treatments of the final landform eg Pit 1 area covered by waste rock versus the area of the tailings dam covered by natural vegetation and laterite.

During the 2009–10 year, a paper ‘Assessing the impact of extreme rainfall events on the geomorphic stability of a conceptual rehabilitated landform in the Northern Territory of Australia’ was presented at the Fourth International Conference on Mine Closure in Perth in September 2009 (Lowry et al 2009). In addition, collaborative research with Associate Professor Greg Hancock comparing the Siberia and CAESAR models resulted in the publication of a paper in the *Journal of Earth Surface Processes and Landforms* (Hancock et al 2010).

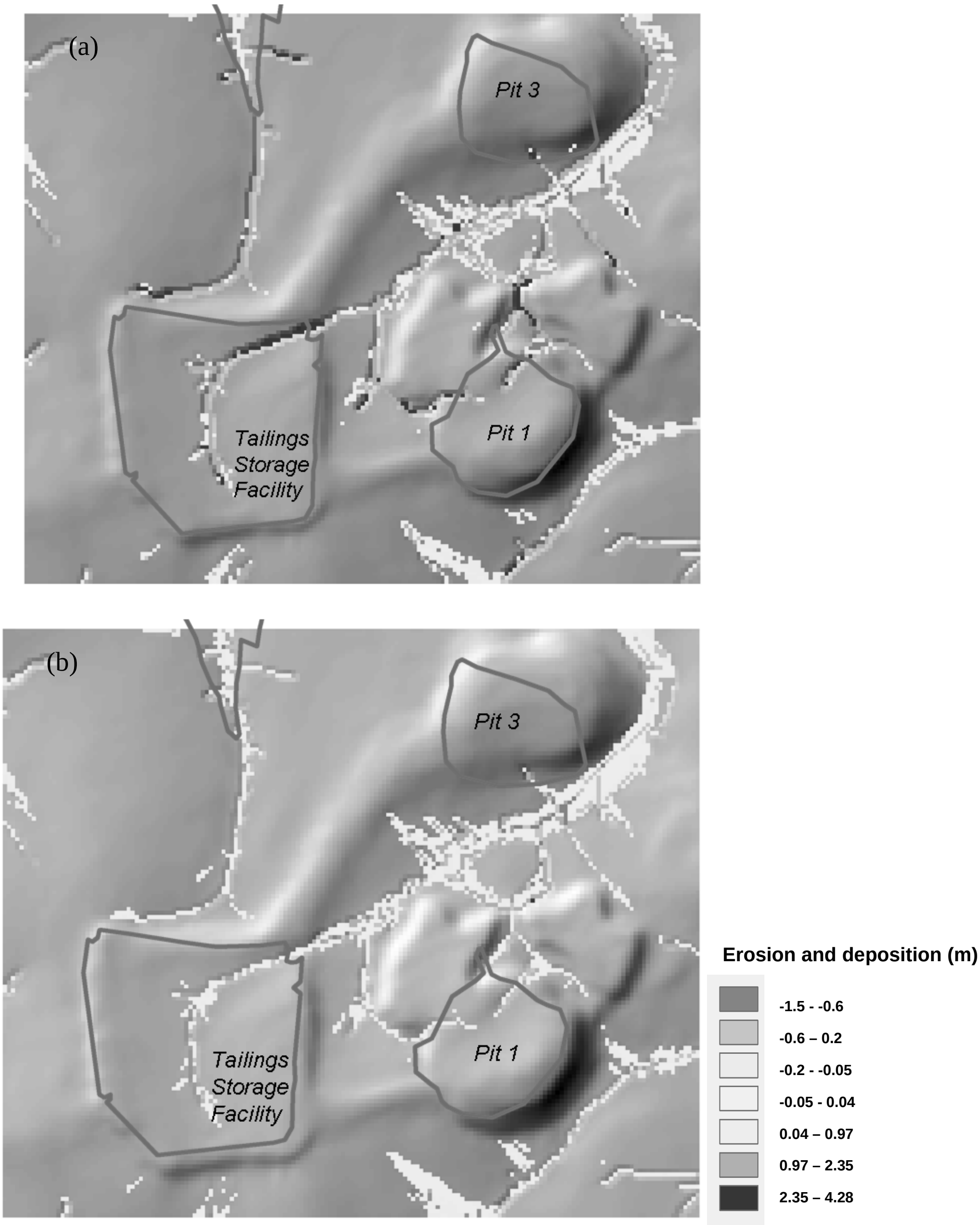

Figure 4 Areas of erosion / deposition on (a) surface with natural/Koolpinyah PSD and (b) waste rock PSD characteristics. A similar result to the waste rock only was produced by the laterite plus waste rock mixture. Lighter colours indicate areas of shallow erosion (0.05–0.2 m), indicating widening of channel.

Steps for completion

Further work on the application of the CAESAR model to assessing closure design proposals for the Ranger site is awaiting the development and supply of a more refined rehabilitated landform design by ERA. Work on analysing and identifying extreme rainfall scenarios for input to the model has been delayed due to the absence of critical staff in the 2009–10 year. The focus for development and application of CAESAR in 2010–11 will be the application of it to the high resolution time series flow and turbidity data that are being produced from the erosion plots on the trial landform. It is also anticipated that ERA will be requesting a geomorphic assessment to be done of design proposals for the capping of Pit#1, the approval for which is likely to be requested in early 2011.

The following steps are required to complete this project:

1 Use the results obtained from the Ranger Trial Landform to calibrate/validate landform evolution model performance.

2 Define the duration and intensity characteristics for extreme rainfall events and apply these to simulations to conduct a risk assessment of landform stability.

3 Compare long-term erosion rates measured on natural undisturbed sites with CAESAR and Siberia simulations for those sites.

4 Apply the CAESAR model to the final landform design.

5 Assess the importance of vegetation development to landform stability, and enhance the ability of CAESAR to account for the effects of developing vegetation. This is a significant area of research that will be expedited by the data being obtained each year from the trial landform, tracking the effects of vegetation cover as it progressively matures.

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Pre-mining radiological conditions at Ranger mine

A Bollhöfer, A Esparon & K Pfitzner

Introduction

The ICRP (2007) recommends that the total annual effective radiation dose to a member of the public from practices such as uranium mining should not exceed 1 milli Sievert (mSv). This dose is in addition to the natural pre-mining background dose and includes the external gamma, inhalation and ingestion pathways. In a high natural background area such as the area around the Ranger mine, determining an additional dose due to mining activities presents a challenge, especially when pre-mining data are scarce and focus on delineating the extent and location of an orebody, rather than determining area-wide radiological conditions.

Pre-mining radiological conditions need to be quantified so that post-mining changes can be assessed in the context of the success of rehabilitation from a radiological perspective. Historical airborne gamma surveys (AGS), coupled with ground truthing surveys, have the potential to provide a powerful tool for an area-wide assessment of pre-mining terrestrial gamma dose rates. AGS and ground truthing surveys have been commissioned and used for regional assessments of radiological conditions at rehabilitated and historic minesites elsewhere in the Alligator Rivers Region (Martin et al 2006, Bollhöfer et al 2008). Whilst a pre-mining AGS was flown over the Alligator Rivers Region, including the Ranger site, in 1976, no ground radiological data of the resolution and spatial coverage needed to calibrate the AGS data are available from that time. The novelty of this project is to use recently-measured, high resolution ground data from an appropriate undisturbed radiologically anomalous area to calibrate the AGS survey data for this anomaly, and then to use the calibrated 1976 AGS to infer pre-mining radiological conditions over the whole Ranger lease.

Methods

1976 AGS data were acquired from Rio Tinto by the NT Government and are available in the public domain (the *Alligator Geophysical Survey,1976.* This subset, or complete NT-wide grids can be downloaded from the Geophysical Archive Data Delivery System (GADDS) at http://www.geoscience.gov.au). Data were re-processed in 2000 by the Northern Territory Geological Survey (NTGS) and then resampled by NTGS at a pixel size of 70 m in 2003. While the line spacing of the survey was 300 m, the flying height is unknown. The 1976 AGS has been used to identify undeveloped radiological analogues in the vicinity of the Ranger lease as potential candidates for ground truthing. A comparison of signal intensity with known uranium occurrences in the MODAT database (NTGS 2007) suggested that Anomaly 2 to the south of the Ranger lease (Figure 1) may be a suitable analog site for Ranger pre-mining radiological conditions as it exhibits a strong airborne gamma signal in the 1976 data, has not been mined, nor is it influenced by operations associated with the Ranger mineral lease.

In addition, Energy Resources of Australia (ERA) has made available to SSD, data from an AGS that was flown in 1997 at a low flying height (50 m) and a higher spatial resolution (200 m line spacing) than the 1976 survey. This data set was used to further refine extensive groundtruthing fieldwork conducted in the 2007 to 2009 dry seasons to establish the exact location and intensities of the Anomalies immediately to the south of the Ranger lease. During the course of the recent ground survey work approximately 2000 external gamma dose rate measurements have been made using environmental dose rate meters, in addition to the determination of soil uranium, thorium and potassium activity concentrations via gamma spectrometry at selected sites.

Figure 1 2006 Quickbird image of the Ranger mine showing the location of the three radiometric anomalies of Anomaly 2

Dry season radon exhalation rates were measured using conventional charcoal cups, with 3 charcoal cups deployed at each of 25 sites for a period of three days. The charcoal cups were then analysed using the SSD NaI gamma detector. In addition, external gamma dose rates were measured and soil scrape samples were taken at the 25 sites for high resolution gamma spectrometry analyses. Track etch detectors were deployed for three months at these sites to measure dry season airborne radon concentration and to establish whether there is a correlation between airborne radon concentration and radon exhalation flux or soil ^{226}Ra activity concentrations. At some of the sites, track etch detectors were deployed at various heights to represent the breathing zones of a person lying down with the head slightly raised, sitting and standing.

Results

Groundtruthing of the airborne gamma survey

Figure 2 shows the results from the 1997 ERA AGS data (total counts) compared with external gamma dose rate measurements (µGy·hr^{-1}) from SSD's groundtruthing. Hegge et al (1980) identified three separate radiometric highs that characterise Anomaly 2. It is apparent that the groundtruthing survey has clearly distinguished these three highs, Anomalies 2A (in the middle) and 2B (to the northeast), and a third Anomaly further to the southwest. These anomalies also show in the 1997 AGS.

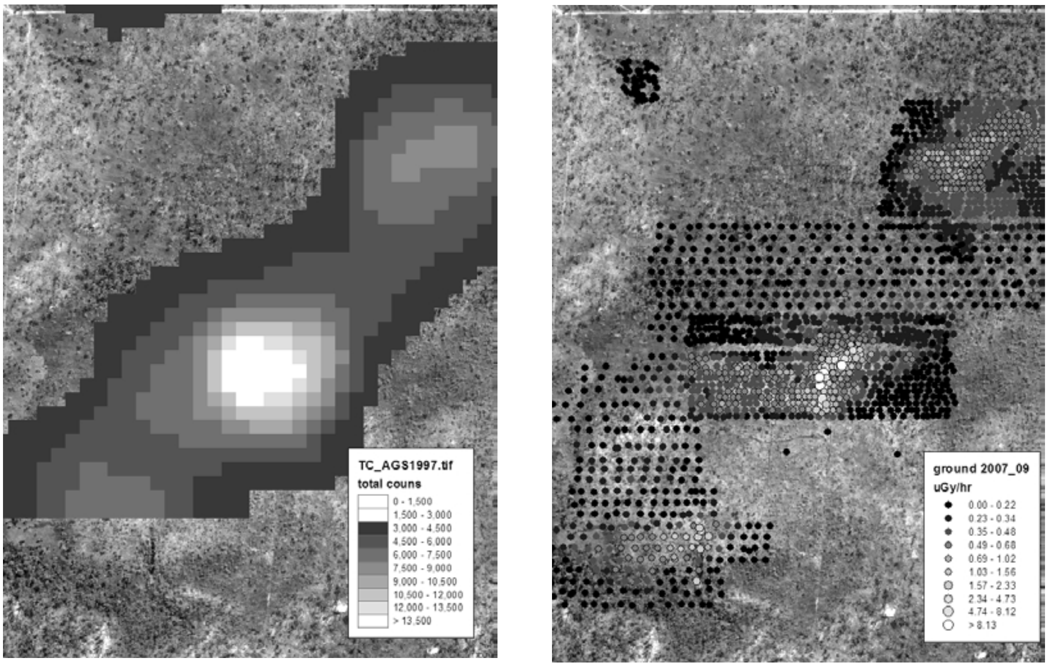

Figure 2 1997 AGS data (courtesy of ERA, left) and the results of the on ground gamma dose rate measurements (right) performed from 2007 to 2009, overlaid on a 2006 Quickbird image of the area immediately south of the Ranger lease

Maximum uranium concentrations at the surface of Anomaly 2A are greater than 6000 mg/kg and maximum gamma dose rates measured at 1m height exceed 20 µGy·hr^{-1}. Typical environmental background uranium concentrations in the vicinity are 4–6.5 mg/kg and background gamma dose rates, including the cosmogenic background, are approximately 0.16 µGy·hr^{-1}.

To groundtruth an AGS, the data acquired in the field (gamma dose rates, uranium, thorium, and/or potassium concentrations) are plotted against the count rates from the respective channels in the AGS. As the groundtruthed data at Anomaly 2 have been acquired at a much higher resolution than both the 1997 and 1976 AGS data, the image is much 'sharper', and it is thus essential to determine appropriate 2-dimensional smoothing algorithms which allow a comparison to be made between the groundtruthed and the AGS data. Ground-based data are typically smoothed by averaging such that the resolution is similar to that of the AGS.

An algorithm to smooth the groundtruthed data has been developed using *MatLab*. The best correlation between the 1997 AGS and the ground based dataset (averaging over a circular footprint) is achieved after applying a small spatial shift and using a smoothing radius of ~80 m for the ground data. To take into account the fact that the aircraft was in motion as data were being acquired, more work is currently underway to use an ellipsoidal footprint to further refine the smoothing of the ground data.

Radon

Radon (^{222}Rn) is a radioactive noble gas and part of the ^{238}U decay series. It emanates from soils and rocks, and exhalation is generally higher for fine grained soils rich in its parent, ^{226}Ra. Once airborne, the shortlived radon decay products (^{218}Po, ^{214}Pb, ^{214}Bi) are produced by the decay of radon and it is these decay products that deliver a radiation dose to humans and animals following inhalation, rather than the radon gas itself.

To determine the source strength, or radon flux, expected for an undisturbed uranium anomaly, radon flux densities have been measured across the Anomaly 2 area. Sampling locations are indicated by the black dots in Figure 2. In addition soil ^{226}Ra activity concentrations were measured at these sites to investigate whether they can be used as a proxy to predict radon flux from the area (Szegvary et al 2007) (Figure 2).

Figure 3 shows the geometric means of the radon flux densities versus the soil ^{226}Ra activity concentrations measured at the sampling sites, both plotted on a logarithmic scale. The sampling sites have been distinguished according to soil type (identified by visual inspection in the field) and sampling location, so that results are plotted for fine gravel, loamy sand and coarse gravel/rocks on top of the anomalies. It appears that radon exhalation does not change significantly with increasing ^{226}Ra activity concentration of the soil directly above the outcropping anomaly, with typical radon flux densities (geometric mean) of 5.6 ± 2.4 Bq·m^{-2}·s^{-1}, similar to values measured above the Ranger #1 and #3 orebodies before mining commenced (2.5–4.1 Bq·m^{-2}·s^{-1}) (Mudd 2004). For soil ^{226}Ra activity concentrations in the lower range of 10–2500 Bq·kg^{-1}, radon flux densities can be predicted by multiplying the measured soil ^{226}Ra activity concentrations by 2.2 g·m^{-2}·s^{-1}. This value is similar to the value reported earlier for non-compacted fine grains in the region close to the minesite (2.7 ± 0.4 g·m^{-2}·s^{-1}) (Lawrence et al 2009).

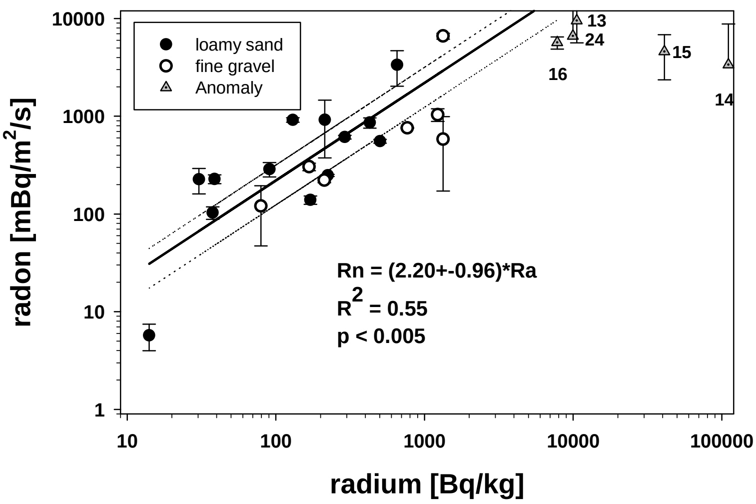

Figure 3 Radon flux densities plotted versus soil radium activity concentrations measured at Anomaly 2. The solid line is a linear fit to the data, the dotted line represents the 95% confidence interval.

Figure 4 shows the radon concentration measured at three different heights at various sites across the area surveyed, and the corresponding soil ^{226}Ra activity concentrations. Whereas radon flux from the soil into air varies by three orders of magnitude, the radon activities measured in air (Bq·m^{-3}) at 1.5 m height vary much less, indicating rapid aerial dispersion and effective mixing at this height. The typical dry season radon concentration (geometric mean) 1.5 m above the area investigated is ~150 Bq·m^{3}, which is about 5 times higher than typical dry season radon concentration measured at Jabiru. There is still a positive correlation between the radon concentration at 1.5 m height and the ^{226}Ra activity concentrations in the underlying soil ($p < 0.005$; $R^2 = 0.4$), but radon concentration only increases slightly above the 150 Bq·m^{3}, rising by ~1 Bq·m^{-3} for every 370 Bq·kg^{-1} increase in soil ^{226}Ra activity concentration. Wet season radon concentrations in air are generally lower than the values cited above, as previously determined at other areas in the Alligator Rivers Region (Martin et al 2004).

The figure illustrates that at areas away from 'hot spots', radon concentration is relatively uniform vertically, but concentrations, and thus inhalation doses, are higher when sitting or lying in close vicinity to the outcropping uranium anomalies that have high ^{226}Ra activity concentrations. This potential exposure route and its dependence on height need to be factored into the broader land use requirements of local indigenous people when assessing potential doses to humans prior to any mining.

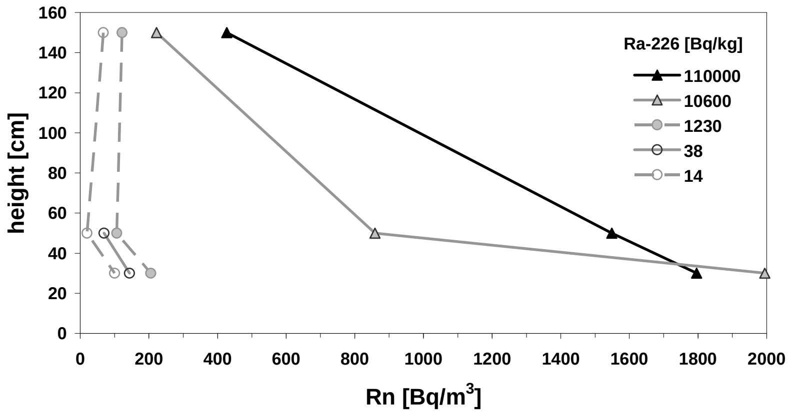

Figure 4 Radon concentration in air for various heights (30 cm, 50 cm, 150 cm) above the ground

Future work

The correlation between historical AGS data from Anomaly 2 and recent ground truthed data from the area will allow determination of average external gamma dose rates across the Ranger lease area before mining commenced at the site. The spatial resolution of the extrapolated dose rates will be limited by the resolution of the AGS, which is at least 1 ha for the 1976 AGS. However, this resolution will suffice to determine pre-mining averages across orebodies 1 and 3, and other areas on the lease. The behaviour of radon in the vicinity of Anomaly 2 has also been studied, and the results will allow the determination of doses from the inhalation of radon progeny above the pre-mining footprint at Ranger, using appropriate UNSCEAR (2000) and ICRP equilibrium and dose conversion factors, respectively.

The potential contribution from the dust inhalation pathway still needs to be established and a separate study is currently underway to quantify the resuspension of dust in naturally vegetated areas. Published resuspension factors for the region (Akber 1992) are comparatively high and need to be verified before radionuclide activity volume concentrations in air (Bq m^{3}) can be inferred from soil radionuclide activity concentrations extrapolated from the AGS survey data.

Further work is required to develop algorithms to upscale the results from the high resolution groundtruthing, to account for the motion of the aircraft as data were being acquired for the 1976 AGS .The original raw data for the 1976 survey have now been acquired for re-analysis since there was concern about artefacts that may have been introduced into the processed data set that had been originally supplied to SSD by the Northern Territory Government. Once data analysis is complete, the radiological conditions on ground around Anomalies 2A and 2B will be correlated with the pre-mining 1976 airborne signal to infer the area-wide radiological conditions at the Ranger lease area before mining commenced.

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Radon exhalation from a rehabilitated landform

A Bollhöfer & J Pfitzner

Introduction

Closure criteria for the rehabilitation of Ranger uranium mine need to incorporate radiological aspects to ensure that exposure of the public to radiation after rehabilitation of the mine is as low as reasonably achievable. As the inhalation of radon is likely to be a significant contributor to radiological dose (Supervising Scientist 2007), particularly in the vicinity of the rehabilitated landform, radon exhalation and its temporal variability need to be estimated. As the radon exhalation rate may change as the final landform evolves after rehabilitation of the site, experimental opportunities have been sought to provide data about the variation in radon emanation rates from relevant areas of the minesite. In particular, ERA's trial landform (Saynor et al 2009) provides a unique opportunity to track emanation rates over many years. The project will enable ***eriss*** and ERA to more confidently predict a long-term radon exhalation flux from a rehabilitated landform and contribute to the development of closure criteria for the site.

The aim of this project is to determine radon (^{222}Rn) exhalation flux densities for various combinations of cover types (two) and re-vegetation strategies (two) on the trial landform and to investigate long term changes in radon exhalation rates. Specifically, the ^{222}Rn exhalation from the four erosion plots (30 m × 30 m) constructed by SSD (Saynor et al 2010) will be measured over several years to investigate whether there are any temporal changes of radon exhalation, taking into account weathering of the rock, erosion and compaction effects, and the effect of developing vegetation on the landform.

Methods

Conventional charcoal canisters (or 'radon cups) are used to measure radon exhalation flux densities. The charcoal canisters used are a standard brass cylindrical design with an internal diameter of 0.070 m, depth 0.058 m and a wall thickness of 0.004 m. Details on the charcoal canister methodology are provided in Bollhöfer et al (2006) and Lawrence (2006).

Construction of the trial landform was completed late in the 2008–09 wet season. Since then, irrigation water has been regularly applied to all areas apart from a 40 m buffer strip that contains the SSD erosion plots. As soil moisture content has a substantial effect on radon exhalation, and because the irrigation water may contain significant concentrations of radium, radon exhalation flux density was measured from the four SSD erosion plots only, which were not irrigated nor affected by spray drift from the irrigation (Saynor, pers comm)

To obtain a true average radon exhalation flux density from the uneven and heterogeneous surface of the four erosion plots, radon cups were placed randomly over the surface. One experimenter would throw a bag filled with sand over his shoulder. The second experimenter would note where the bag first hit the ground, this being the selected location for charcoal cup placement. If placed on rocks, the rim of the charcoal cup was sealed using putty. This is in contrast to many other studies where radon cups are placed at 'convenient' locations where they can easily be embedded into the finer grained soil. Fine grained material exhibits higher radon exhalation flux densities than solid rock (Lawrence et al 2009). Hence, results of radon exhalation measurements can potentially be skewed if the sampling design is not random (Bollhöfer et al 2006).

The location and a description of the four erosion plots where measurements are being taken are shown on Figure 1 and in Table 1, respectively, and are further described in Saynor et al (2010). Generally, 15–20 radon cups were deployed randomly across each erosion plot and were exposed for four days in May 2009, and for three days each in September 2009, February 2010 and May 2010. The charcoal cups were collected after exposure, sealed and sent to the SSD Darwin laboratories, where they were analysed using a NaI gamma detector.

Soil samples have been collected from the surface of the erosion plots, and from the troughs and basins installed at the plots to collect run-off and sediment. The samples were size fractionated (<63 μm and 63 μm – 2 mm), and the two size fractions analysed via gamma spectrometry using HPGe gamma detectors. The methods for gamma spectrometry are described in Murray et al (1987), Marten (1992) and Esparon and Pfitzner (2010).

Progress to date

Radon cups were deployed before the trial landform was constructed to determine the radon exhalation from the substrate underlying the constructed landform. Radon flux densities from the pre-construction substrate follow a log-normal distribution with a range from 24 to 144 mBq·m^{-2}·s^{-1} and geometric mean and median both equal to 73 mBq·m^{-2}·s^{-1}. This is similar to the average (±1SD) late dry season radon flux density of 64 ±25 mBq·m^{-2}·s^{-1}, which was previously determined for the region (Todd et al 1998).

Radon exhalation flux density measurements now cover one full seasonal cycle. A plot of the results is presented in Figure 1.

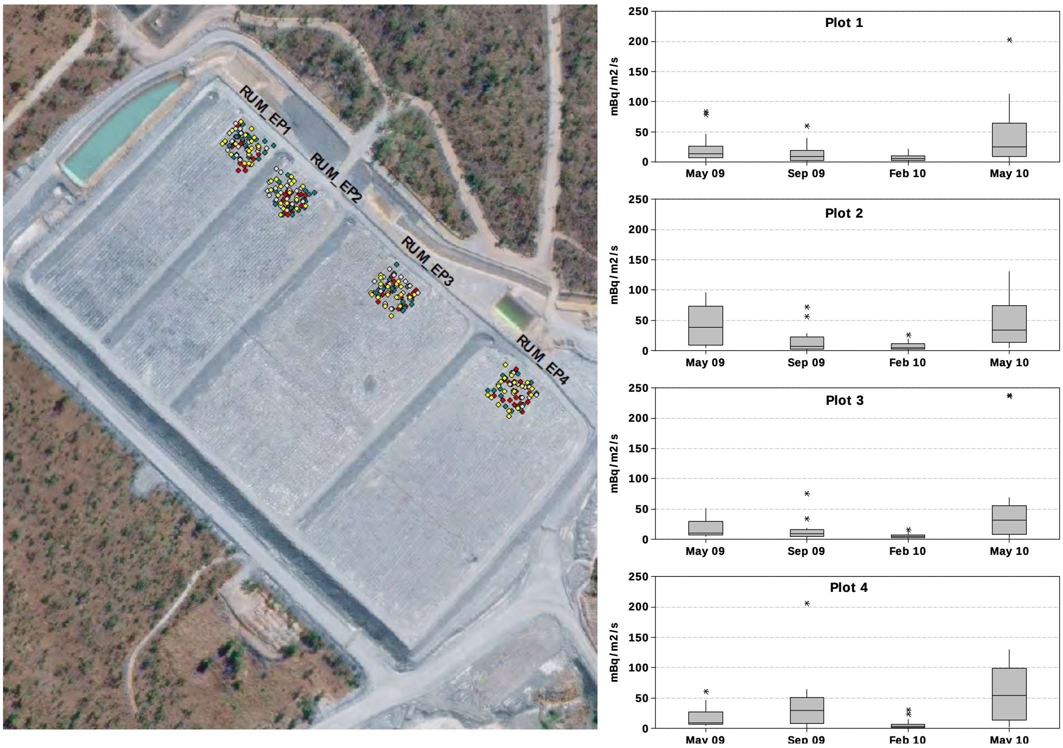

Figure 1 Locations of the radon exhalation measurements conducted in 2009 (30 April–4 May, 14–19 September) and 2010 (2–5 February, 14–17 May). The right hand figure shows the median and 1st and 3rd percentiles, respectively, of the radon exhalation flux densities measured. Whiskers show the range of the measured data, individual datapoints indicate outliers.

Generally, radon flux densities measured from the four un-irrigated erosion plots are lower than those measured on the ground prior to landform construction, most likely due to the rocky nature of the trial landform with typical rock sizes reaching up to 300 mm or more in diameter. As noted above, radon exhalation from fine grained soils is greater than that emanating from solid rock.

The radon exhalation shows a seasonal variation typical of the region (Lawrence et al 2009) with radon exhalation flux densities lower during the wet season compared to the dry season. Average wet season radon exhalation flux density (February 10) across the four plots were 7(*4*) ± 1 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ (average (*geomean*) ± SE). Dry season radon exhalation flux densities were approximately four times higher at 26(*12*) ± 5 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ and 27(*13*) ± 6 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$, respectively, for erosion plots 1 and 3; five times higher at 34(*18*)±5 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ for erosion plot 2; and 7 times higher at 46(*22*)±10 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ for erosion plot 4.

Table 1 Description of the four erosion plots and average radon flux densities measured on the surface in 2009–10

	Treatment	**^{222}Rn flux density [$mBq{\cdot}m^{-2}{\cdot}s^{-1}$]** **Arithmetic (*geometric*) average ± std error**			
		May 2009	Sep 2009	Feb 2010	May 2010
RUM_EP1	Waste rock material planted with tube stock	22(*14*) ± 11	14(*7*) ± 8	7(*4*) ± 3	43(*21*) ± 25
RUM_EP2	Waste rock material planted by direct seeding	41(*26*) ± 16	15(7) ± 9	8(*5*) ± 4	45(*28*) ± 20
RUM_EP3	30% lateritic material mixed with waste rock (2 m), direct seeding	19(*13*) ± 7	14(*9*) ± 8	5(*3*) ± 2	51(*21*) ± 35
RUM_EP4	30% lateritic material mixed with waste rock (2 m), tube stock.	18(*13*) ± 7	40(*19*) ± 32	6(*3*) ± 4	83(*42*) ± 51

To determine if there is a relationship between radon exhalation flux density at the surface and activity concentration in the soil, activity concentrations were measured in samples taken from the surface of the four erosion plots. These surface soils were taken from various locations on the erosion plots and then combined to produce a composite for analysis. The less than 2 mm fraction was analysed. In addition, eroded material was collected after a storm event on 11 November 2009. The erosion products were collected from the troughs around each erosion plot, and also from the basins (Saynor et al 2010) at the exit to each of the plots. They were then size fractionated into the <63 μm fraction and the >63 μm and < 2 mm size fraction.

As expected, soil activity concentrations were higher in the silt and clay fraction (<63 μm) than in the sand fraction (>63 μm, <2 mm) (Figure 2). Eroded sediment activity concentrations were generally lower in erosion products collected from erosion plot 1 compared with erosion plot 2, in agreement with activity concentration measurements conducted on waste rock material collected directly from the surface of the two plots. The ^{226}Ra activity concentrations measured in the surface soil samples from the two plots were 193 and 422 $Bq{\cdot}kg^{-1}$. The higher soil activity concentration partly explains the higher average dry season radon exhalation flux densities measured from erosion plot 2 compared with erosion plot 1 (both treated with waste rock only) (see Table 1).

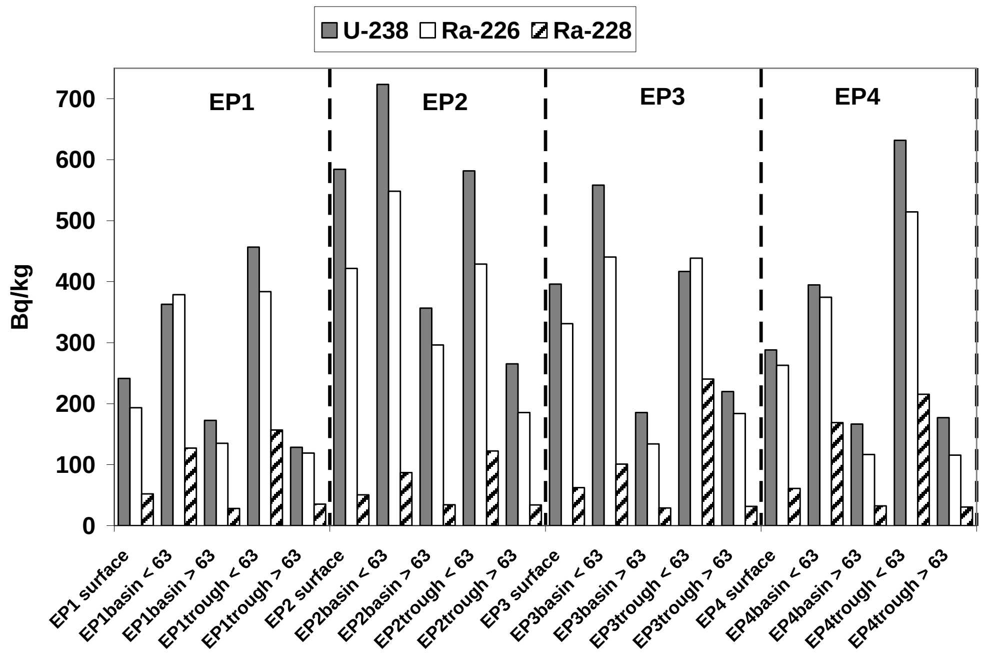

Figure 2 ^{238}U, ^{226}Ra and ^{228}Ra activity concentrations measured in soils and erosion products from erosion plots (EP) 1–4

The two erosion plots treated with 30% lateritic material mixed with waste rock (EP3 and EP4) showed little difference in ^{226}Ra activity concentrations in the <63 µm fraction of eroded material (average of 440 and 444 Bq·kg^{-1}). The >63 µm fraction from erosion plot 3 exhibited higher ^{226}Ra activity concentrations (160 Bq·kg^{-1}) compared with plot 4 (120 Bq·kg^{-1}), in agreement with activity concentration measurements of the surface soil samples of 330 and 260 Bq·kg^{-1}, respectively. The reason for the much higher dry season radon exhalation flux density from erosion plot 4 compared with the three other plots is unknown, but may be due to a higher proportion of fines as compared with erosion plot 3, or a higher soil porosity. These possibilities are currently being investigated.

Future work

Radon exhalation surveys across the four erosion plots will continue to be conducted every 4 months to investigate seasonal and long term temporal changes in radon exhalation from the trial landform. In addition, soil samples will be collected from the four erosion plots annually and radionuclide activity concentrations will be measured in the <63 µm fraction and the >63 µm, < 2 mm size fractions.

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Development of surface water quality closure criteria for Ranger billabongs using macroinvertebrate community data

C Humphrey, K Turner & D Jones

Background and aims

This paper provides a status report on the development of surface water quality closure criteria (for operations and closure) for Ranger billabongs using macroinvertebrate community data. Specifically, the study aims to quantify macroinvertebrate community structure across a gradient of water quality disturbance in the Alligator Rivers Region (ARR) so as to provide a basis for developing surface water quality closure criteria for Georgetown and Coonjimba Billabongs (Map 3).

After the Ranger mine ceases operations disturbed areas will need to be rehabilitated to a condition consistent with the values of Kakadu National Park (KNP), and ultimately to be at a standard suitable for incorporation back into the Park. Ideally, in the case of natural waterbodies, this would mean that their post-rehabilitation environmental values should be consistent with the expectations of the traditional owners of the land, and hence be consistent with those of similar, undisturbed habitats of KNP.

Following mine-site closure, there is the potential for delivery of solutes and suspended sediment from the rehabilitated mine landform. These solutes and sediments may impact on the ecological values of adjacent waterbodies if they are not maintained at appropriate concentrations. Hence, a key objective of the closure planning process will be to produce a rehabilitation design for the disturbed areas such that the delivery of solutes and suspended sediment from the disturbed footprint in the minesite catchments (eg Corridor and Georgetown Creeks) will not compromise the post closure environmental objectives for the waterbodies.

In this project, the structure of lentic macroinvertebrate communities is being used as the environmental response indicator for developing water quality closure criteria for waterbodies adjacent to the mine-site. Specifically, the work is being directed at Georgetown Billabong (GTB) which is the largest natural and still largely unimpacted waterbody in close proximity to the operational mine area. During each wet season Georgetown Billabong receives low level inputs of mine-derived solutes from Corridor Creek (Map 2).

The approach to deriving such criteria from local biological response data follows that outlined in the Australian and New Zealand Water Quality Guidelines (ANZECC & ARMCANZ 2000). Briefly, if the post-closure condition in Georgetown Billabong is consistent with similar undisturbed (reference) billabong environments of Kakadu, then the range of water quality that supports this ecological condition (as measured by suitable surrogate biological indicators) may be used for this purpose.

Summary of studies conducted between 2006 and 2009

Results of macroinvertebrate sampling in billabongs and other waterbodies

Macroinvertebrates have been collected from mainly macrophyte (water column) habitat in GTB several times, including 1978 (pre-mining), 1995, 1996 and 2006. On the first three of these occasions, the macroinvertebrate communities of this billabong consistently resembled those of reference waterbodies in the ARR, suggesting that the historical water quality regime in GTB was compatible with the maintenance of the aquatic ecosystem values of KNP.

In 2006, macroinvertebrates were collected from both macrophyte and benthic (sediment) habitats in GTB with the samples from each habitat being processed separately. Whilst the aggregated data showed essentially no difference between macroinvertebrate communities of GTB and reference billabongs, and hence were in agreement with the results from the previous surveys, when the macrophyte and benthic data were analysed separately it was found that the sediment-dwelling communities were less diverse in GTB than in reference waterbodies (Humphrey et al 2009). This raised the question as to whether the lower diversity of benthic macroinvertebrates in GTB could have been the result of accumulation of mine-derived solutes in the sediments and flagged the need to specifically investigate this issue.

Sediment U concentrations along GTB in 2007

In 2007–08, an investigation commenced to determine whether the lower benthic diversity in GTB was related to mine inputs or habitat type. In this context it should be noted that the concentration of U in GTB has always been higher than in reference waterbodies in the region. This higher value is likely the result of downstream transport of erosion products from the top of the Ranger 1 orebody in the upper catchment of GTB.

The most recent work has involved studies of sediment chemistry and sediment physical structure. Sediment samples collected from four littoral sites along the western side of GTB in 2007 (Figure 1) showed concentrations of U (mean of all samples ~42 mg/kg, range 30–63 mg/kg, n = 11) higher than U concentrations previously measured by ERA and others (Iles & Klessa 2008, Humphrey et al 2009, 2010) in GTB over the past 25 years. This raised the possibility that there could have been a substantial rise in the mean U concentration between 2001 and the present, noting that the 2007 samples were specifically collected from the western edge, rather than a mix of western-edge and central billabong locations as previously.

Identification of outstanding sediment chemistry characterisation required for GTB

Because of (1) a substantial gap of 5 years in the collection of U sediment data between 2001 and 2007, (2) differences in locations within GTB at which samples have been collected over time, and (3) differences in methods of chemical analysis, it was unclear whether there had actually been a recent rise in U across the billabong as a whole. To address these questions, the following investigations were initiated: (i) re-analysis of U using similar methods to the 2007–08 study, in a number of GTB sediment samples collected in 2006 by ERAES (ERA Environmental Strategy), (ii) sampling of a lateral transect across the billabong in 2009 to determine the profile of U concentrations across the waterbody, (iii) different chemical analysis methods for U extraction in selected historical samples (to quantify dependence of U extraction on the various analytical methods used in the past), and (iv) the dependence of U concentration on particle size, as well as mineralogical characterisation, using selected historical samples.

Characterising physical properties of GTB sediments

The littoral sediments in GTB consist mainly of fine cracking clays, and are generally devoid of surface vegetation during the dry season when the sediment exposed around the gently sloping margins undergoes desiccation-induced cracking. Of the billabongs sampled, these conditions appeared to be unique to Georgetown and Coonjimba Billabongs, and suggest that the physical nature of the sediments may be an important factor contributing to the low diversity of benthic organisms. The very fine particle size of suspended sediments in GTB relative to other billabongs was noted by Walker and Tyler (1982). Should these sediments dry out substantially and harden when exposed in the dry season, life stages of benthic organisms adapted to seeking refuge in sediments upon exposure and drying may not be able to persist. Moreover and once re-wetted in the wet season, such sediments may not rapidly return to a sufficiently softened and yielding form for residence by sediment-dwelling organisms. To resolve this potential compaction issue, a program of measuring sediment penetration resistance (using a penetrometer) was initiated.

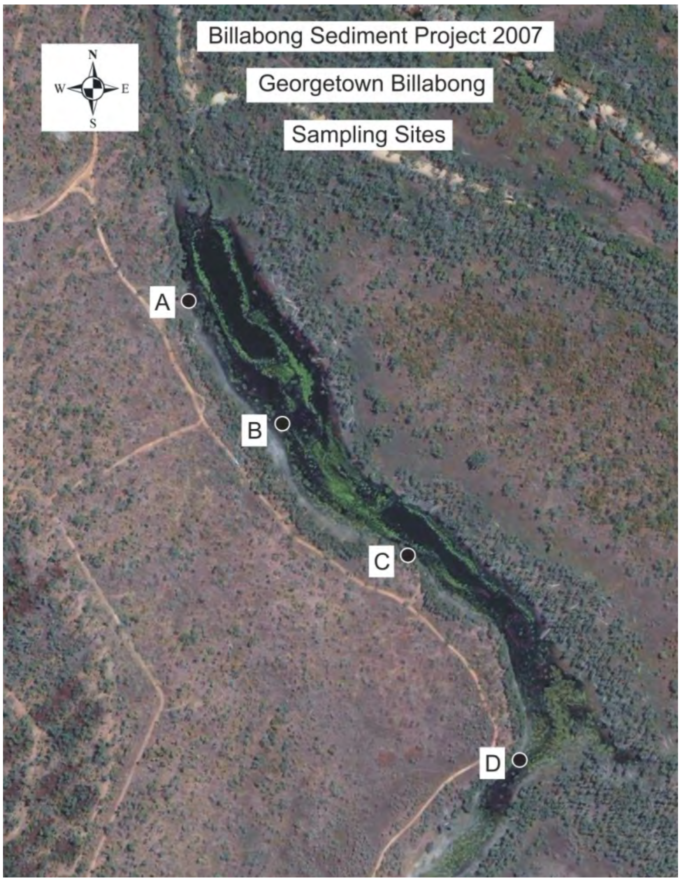

Figure 1 Map of Georgetown Billabong showing four sampling locations from which sediments were collected in August 2007 for chemical analysis

The results of these investigations, together with a dedicated field assessment of U sediment toxicity that is currently underway (see Harford et al 2010, pp 32–36 in this volume) will determine whether or not the lower benthic macroinvertebrate diversity in GTB is related to current mining activities. If unrelated to mining activities, it would confirm that the historical water record can be used to derive closure criteria that are protective of both lentic and benthic macroinvertebrate communities, notwithstanding there has been an increase in solute loads in the billabong between 1996 and the present (2010).

Progress to date

Whilst there have been some delays, because of higher priorities elsewhere in the environmental chemistry work program, in several aspects of the work program originally proposed for 2009–10, good progress has been made.

1. Processing of sediment samples collected in 2006 by ERAES

Of all the archived samples from this sampling program, only one has a matching collection location and particle size distribution common to the samples that have previously been collected and analysed by ERA and others. Processing and chemical analysis of this sample are underway.

2. Cross-billabong metal concentrations in sediment

Sediment samples were collected from GTB along a transect located at the northern (downstream) end of the billabong near the Georgetown-Magela Creek confluence (gauge board site) in May 2009. The western starting point of the transect was at location A (Figure 1), at the littoral margin on the western side, which coincided with one of ***eriss***'s macroinvertebrate sampling sites from 2006. This sampling was conducted in order to quantify the extent of cross-billabong variation of metal concentrations and thereby ascertain whether sample location was having a critical influence on the concentrations of U present in the sediment.

A single sample of the top 5 cm of sediment was collected at each location along the transect. A trowel was used in shallow waters, and an Ekman grab in deeper waters where hand collection was not possible. The < 63 µm fraction was separated for analysis since this fraction is considered to be the most relevant for biological interaction with benthic biota. The proportion of the < 63 µm fraction in each sample is shown in Table 1. A strong acid digest (nitric/perchloric) was used for consistency with previous strong acid digest methods that had been used. The results showed a substantial gradient in U concentration across GTB from highest on the western side (closest to the minesite), to lowest on the eastern side, adjacent to Magela Creek (Figure 2).

Table 1 Fraction of fine sediment (< 63 µm) in samples collected along an W-E transect (starting at A in Figure 1) of Georgetown Billabong in May 2007

Location (metres from western edge)	Proportion of sediment sample < 63 µm (%)
0	61
10	86
20	80
30	80
40	92
50	21
60	40
70	47
80	81
90	77

The eastern side of the billabong closest to Magela Creek receives greater flushing by Magela Creek during high flow events, and thus is less likely to accumulate U-enriched particles originating from further upstream in the GTB catchment. Additionally the water column would likely be more diluted towards the eastern edge resulting in lower U concentrations in the water column and hence less uptake by the sediment. Both of these mechanisms could explain the lower concentrations on the eastern side.

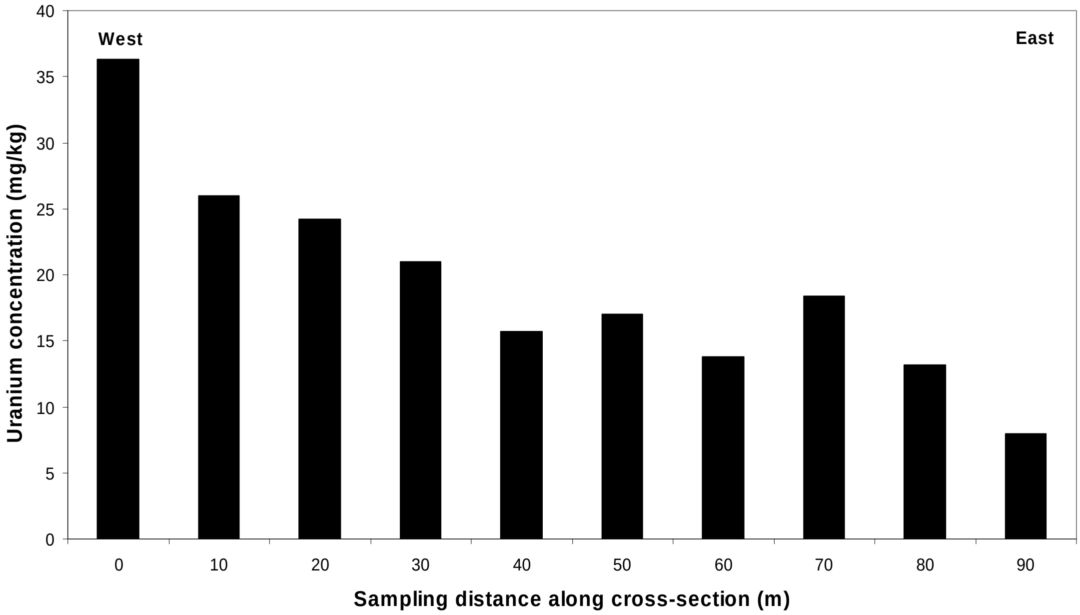

Figure 2 Uranium concentration of sediments collected along a transect in GTB (west to east) in May 2009. Uranium extracted using Nitric/perchloric digest of <63 µm fraction. A single sample was collected at each location.

The marked gradient in sediment U concentrations across GTB highlights potential problems in assessing time series data obtained from samples that have been collected at a number of different locations (eg the interannual trends reported in Humphrey et al 2009). In this context it should be noted that most of the previous samples were collected towards the western edge of the billabong at its downstream end.

It is critical that both the location in GTB, and time of year when samples were collected, are reported otherwise apparent trends could be the result of spatial artefacts created by sampling near the water's edge, given that this position depends on when in the season the sampling is done. For example, a sediment sample collected from GTB in November 1978 upstream of the transect at A referred to in Figure 1 – but in a similar central channel location across the billabong (east to west) and with similar fine silt and clay characteristics – had a U concentration of 16 mg/kg (Noller & Hart 1993). This value is very similar to the contemporary value reported for the corresponding central location (downstream), sampled in 2009 (Figure 1).

The above discussion indicates the need to conduct a dedicated sediment sampling and chemical analysis program in GTB which would include sampling across a number of transects located along the billabong. This characterisation would provide a more robust reference condition against which to quantify future changes in sediment U concentrations and for designing an effective monitoring program to detect such changes.

3. Comparison of sediment digest methods

Sediment digest methods will be compared using a common set of samples to determine if the historical data need to be 'normalised' to account for differences in sediment digest efficiency factors. If required, this normalisation should remove or substantially reduce variation in U results introduced by the use of different digest methods through time.

Samples collected from GTB in proximity to the gauge board in 2007 by SSD will be analysed for total U concentration using X-Ray Fluorescence and Gamma Spectrometry methods. U extraction efficiencies for the different acid digest extraction techniques used by SSD and ERAES through time will also be determined. This work will be carried out in early 2011.

4. Physical characteristics of sediments in billabongs

A penetrometer was used to measure the extent of compaction of exposed (dry season) littoral sediments (Figure 2). The hypothesis being tested was that billabongs dominated by fine-grained sediment (GTB and Coonjimba Billabong) would contain more compacted sediments than other sandier reference billabongs.

Figure 2 Penetrometer being used in Malabanbandju Billabong

Exposed sediment penetration studies were carried out at Sandy, Buba, Anbangbang, Malabanbandju, Wirnmuyurr, Corndorl, Gulungul, Baralil and Georgetown Billabongs and Jabiru Lake between 19th and 21st September 2010. Ranger retention ponds RP1 and RP2 and Coonjimba Billabong downstream of RP1 were measured on 14th October 2010. At each of the waterbodies, penetration was measured along transects perpendicular to the water's edge at 5 sites coinciding with the locations from which macroinvertebrate samples had been collected in May 2006 (the last time that a comprehensive macroinvertebrate survey had been undertaken for the billabongs). These data are awaiting analysis.

Pending the outcome of sediment-related studies described above, the water quality record for 2006 has been included in the database used to derive an acceptable ceiling water quality sufficient to protect the macroinvertebrate communities of GTB post-closure. The combined 1995, 1996 and 2006 data (corresponding to years for which macroinvertebrate data were obtained) have been used to derive provisional water quality closure criteria for the water column (Jones et al 2008). Of course, the meeting of numerical closure values specified for a limited range of measured water quality parameters needs to be considered in the context of providing reassurance that the aquatic ecosystem is actually remaining protected from post-closure impacts. Thus periodic post-closure surveys of macroinvertebrate communities should be part of the post-closure monitoring regime for natural waterbodies downstream of the rehabilitated Ranger 'footprint'.

Any new data arising from periodic sampling of macroinvertebrates from GTB during the operational phase will contribute to a rolling assessment and potential refinement of the currently-proposed water quality closure criteria. The results from the companion U in sediment toxicity study (Harford et al, pp 32–36 this volume) will also inform the setting of water quality criteria for GTB, ensuring water quality criteria are not relaxed to thresholds that could adversely impact upon sediment quality, by virtue of the adsorption to sediment of U initially present (at concentrations lower than that required to protect lentic macroinvertebrates) in the water column.

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Effects of fine suspended sediment on billabong limnology

C Humphrey, D Buckle & D Jones

Background and aims

This paper provides an update on a study designed to assess the effects of fine suspended sediment on billabong limnology, initially reported in Buckle et al (2010). The project is part of a portfolio of projects that are progressing the development of local water quality criteria relevant to both mine operations and mine closure.

As reported by Buckle et al (2010), the project aims to review, determine and infer the effects of fine suspended sediment on billabong limnology (physico-chemistry, primary production) using field water quality and biological effects data from both historic as well as new investigations.

The findings from the study will be used as:

i a component of a multiple lines of evidence approach to assessing effects of suspended sediment on tropical freshwater biota (complementing a current laboratory-based project developing ecotoxicological test procedures for suspended sediment – 'The effects of suspended sediment on tropical freshwater biota', Harford et al 2010); and

ii to assist with developing suspended sediment/turbidity surface water guideline criteria for both the operational and closure phases of the Ranger minesite.

Suspended sediment has been identified as an aquatic ecosystem stressor that will most likely assume greater significance in the future, as a consequence of major disturbances of surface materials occurring during rehabilitation works on the Ranger site (Harford et al 2010). Billabongs immediately downstream of Ranger are at greatest risk from erosion of fine particulate matter. For example, Humphrey (1985) and Nanson et al (1990) document high suspended solids loadings and significant sedimentation in Coonjimba Billabong as a consequence of the failure of new earthwork structures after an intense rain-storm event that occurred over the Ranger site in February 1980.

The most recent investigations have monitored the effects on billabong limnology of natural increases in turbidity that occur over the late dry season in shallow backflow billabongs adjacent to Ranger. Turbidity may disrupt a number of ecosystem functions, with one of the best-documented of these being primary (plant) production through attenuation of light. It was hypothesised that a threshold value of turbidity, above which important billabong photosynthetic functions are disrupted, may be able to be inferred from the data. This field-observed threshold would complement the findings from laboratory studies.

Data from previous limnological studies conducted in the region have been reviewed to examine possible relationships between turbidity, chlorophyll-*a* and dissolved oxygen. The field investigation conducted in Georgetown Billabong (GTB) during the dry season of 2009 (Buckle et al 2010) used in situ continuous (5 minute intervals) measurements of dissolved oxygen and turbidity, near-surface and at depth, for the period May to October/November 2009 (with the aim of encompassing the period of significant dry season increase in turbidity in the billabong). Depth profiles of water quality parameters and photosynthetically available radiation (PAR), together with surface and depth samples for chlorophyll-*a* (phytoplankton abundance) and dissolved organic carbon (produced by decay of plant matter and contributing to biological oxygen demand) determinations, were collected fortnightly over this period. The grab samples were also analysed for suspended sediment (SS) concentrations (0.45–63 µm fraction) to provide a correlation between turbidity and SS.

The major outcome from the review of historical data and analysis of data collected in 2009 was the unimodal relationship observed between near-surface chlorophyll-*a* and turbidity (Buckle et al 2010). For the data collected in 1980–81 by Humphrey and Simpson (1985), phytoplankton production appeared to be inhibited at turbidity values around 50 NTU. While the 2009 data appeared to support the historical threshold effect discerned in the 1980–81 data, the relationship was less evident due to the generally lower turbidity values observed in 2009 (only one turbidity value recorded above 30 NTU; Buckle et al 2010). Given the lack of turbidity values above 30 NTU for the 2009 dry season and the changed (lower) turbidity conditions observed since 1981 in GTB (Figure 1), Buckle et al (2010) recommended further sampling of chlorophyll-*a* and turbidity through the 2010 dry season in order to capture a higher turbidity period in the event that this occurred.

Progress during 2010

Regular grab samples of water for turbidity measurement were collected from Georgetown Billabong through the 2010 dry season. ERA turbidity data were also examined regularly to identify any increase in turbidity over 30 NTU. To have a noticeable influence on phytoplanktonic growth, sustained periods of increased turbidity would be required. Hence short (< 7 days) duration events over 30 NTU were not sampled.

Monthly turbidity data from 1980 and 1981 (Humphrey and Simpson 1985), data collected between 1982 and 2008 by Energy Resources of Australia (ERA), as well as data collected during 2009 and 2010 by ***eriss*** and ERA, are shown in Figure 1.

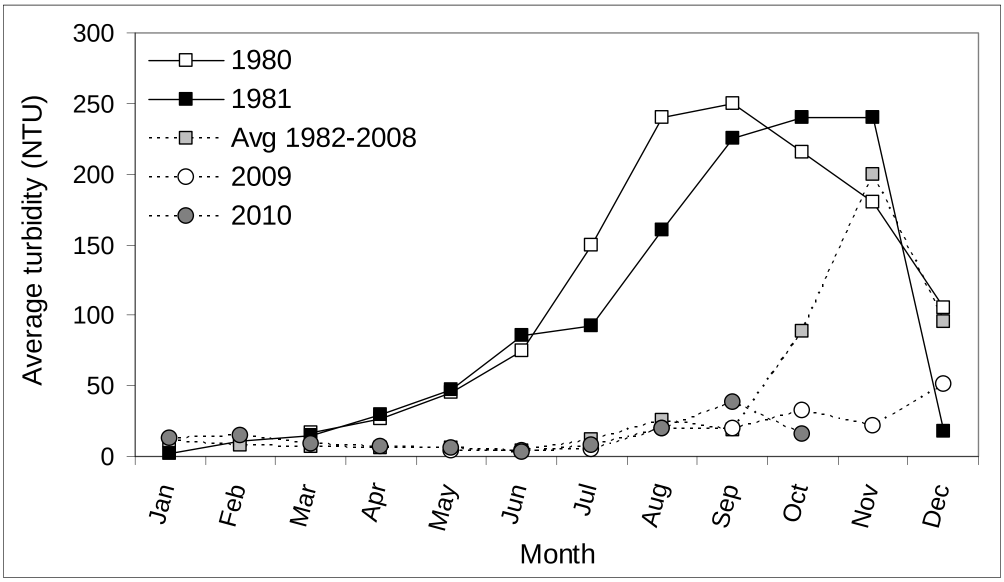

Figure 1 Average monthly turbidity (NTU) for Georgetown Billabong from 1980 to 2010. ***eriss*** data used for 1980, 1981 and 2009 (June – October). Energy Resources of Australia data used for 1982–2008 and November 2009 – November 2010.

Possible reasons for the reduction in late dry season turbidity in Georgetown Billabong surface waters between 1980 and the present were discussed in Buckle et al (2010). Suffice it to say that the relatively low late dry season turbidity that was observed in 2009 was again seen in the 2010 dry season (Figure 1). While turbidity will be again monitored in the 2011 dry season it is quite possible that turbidity levels much greater than 30 NTU in Georgetown Billabong will now be the exception rather than the rule as they were in the 1980s and 1990s. In this event the 1980s data will be the most appropriate to use to discern the effects of turbidity upon phytoplanktonic production.

Further work

Georgetown Billabong will be monitored in the 2011 dry season to check whether any sustained periods of turbidity above 30 NTU occur. In this event dissolved oxygen and turbidity profiles will be frequently measured and grab water samples will be collected for determination of chlorophyll-*a*. Determining the proportion of organic and inorganic material contributing to the suspended sediment should also be considered to provide additional insights into the effect of algal self shading.

It should be noted that the material that has been presented to date (Buckle et al (2010) and above) represents only a subset of that collected mainly in 2009. Analysis of the complete dataset is yet to be completed and may yield more insights than have been discerned to date.

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Use of vegetation analogues to guide planning for rehabilitation of the Ranger minesite

C Humphrey, G Fox, G Staben & J Lowry

Background and aims

Characterisation of plant communities from appropriate natural analogue sites will assist in selection of species for revegetation of the Ranger mine landform following decommissioning of the site. The characteristics of these communities will also assist in developing numerical performance measures (closure criteria) against which the success of the revegetation can be tracked. For the range of key vegetation community types that represent the spectrum of environments likely to be found across the rehabilitated footprint, relationships between the occurrence of such communities and key geomorphic features (eg soil type, slope, effective soil depth, etc.) of the landscape need to be identified. By identifying the key environmental features that are associated with particular vegetation community types, either (i) the conditions required to support these communities or, alternatively, (ii) the community types that best suit particular environmental conditions, may be specified for the rehabilitated landform at Ranger.

Data previously obtained by ***eriss*** and ERA Environmental Strategy (ERAES, formerly Earth Water Life Sciences) on plant communities in the ARR, including data from reference and analogue sites, have been used to identify and characterise target communities that will provide a basis for rehabilitation and subsequent post-closure monitoring. The collective data from analogue/reference sites are being used to derive plant-environment relationships with the aim of developing predictive models based upon physical and chemical input variables.

Studies conducted between 2003 and 2009

The following points summarise the vegetation surveying and modelling carried out between 2003 and 2009:

1 Gathering of plant community data by ERAES from 20 plots located in a small analogue area (400 ha) to the east of the Ranger site, the 'Georgetown analogue site' (Hollingsworth & Meek 2003), and modelling (by ERAES) of key species according to major landform characteristics (Hollingsworth et al 2007).

2 Gathering of plant community data by ***eriss*** from additional areas of the ARR, including hill slopes and lowland sites near Ranger minesite (from Brennan 2005) as well as woodland areas of low relief in the Gulungul Creek catchment (from Staben et al 2009). Multivariate classification of the combined ERAES (from 1 above) and ***eriss*** data, representing 121 sites, identified three main vegetation community types, *Melaleuca* woodland, mixed eucalypt woodland and dry mixed eucalypt woodland (Humphrey & Fox 2010).

3 Collection by ***eriss*** and ERAES in 2006 of soil physical and chemical characteristics from hill slope in the broader ARR (Brennan 2005) and Georgetown analogue sites, from 1 above. An analysis was performed to identify possible soil property-plant relationships

for these natural analogue sites. However, at best, only weak relationships were found between soil properties and vegetation communities, indicating that physical landscape and landform features were likely to be more critical determinants of community type (Humphrey et al 2009).

4 The modelling conducted by ERAES in item 1 above, did not include the full suite of soil physical and chemical characteristics that had been collected for the analogue sites. Potential problems were also identified with the landscape physical characterisation variables (eg slope angle and length) used in the modelling (Humphrey & Fox 2010). This work used a synthesised Digital Elevation Model (DEM), the source components of which are being reassessed for accuracy and applicability by ***eriss*** (see below).

Studies conducted since 2009

The three major vegetation groups identified from the 121-site data set referred to above (item 2), were all found to occur within the relatively small Georgetown analogue area that had been used by ERAES in its original classification studies (see item 1 above). Thus, rather than continuing with plant-environment modelling over the large geographical area represented by the combined ***eriss*** and ERAES ARR data set, a decision was made to use the smaller Georgetown analogue area as the basis for acquisition of more detailed future survey data required for modelling purposes. By geographically restricting the area of focus, other sources of variation that may have confounded previous comparisons (eg altitude, parent soil type, broad geographical separation) can be removed.

A workplan was presented to ARRTC at its April 2010 (25^{th}) meeting that proposed the following tasks:

1. Conduct additional plant surveys on the Georgetown analogue area to increase the number of sites represented in some of the currently under-represented classification groups in this area.
2. Further characterise, for inclusion in modelling, the soil properties of sites associated with the different vegetation classification groups represented in the analogue area. (This could include soil properties not previously measured.)
3. Produce a DEM with appropriate resolution for the analogue area and from this re-derive a full suite of landscape variables for inclusion in the terrain modelling analysis. This task is necessary because the original terrain analysis conducted by ERAES used a DEM that was likely of insufficient accuracy to capture the shallow slopes characterising the analogue area. Derivation of the descriptive physical features is especially important given that the landform design criteria contained in the ERA closure model were based on this former terrain analysis.
4. Undertake analyses of the substrate/soil chemistry data from the newly-constructed trial landform and from the various historical rehabilitation trial sites on the Ranger minesite to assess the potential of different types of mine-derived materials to sustain plant communities.
5. Using the data acquired from 1 to 4 above, assess if there are any significant relationships between plant community/species and environmental variables that can be used for predictive purposes. Recently-developed more powerful multivariate techniques will be used to reanalyse the data to determine if more subtle plant-environment relationships had been missed by previous (ie PRIMER BIOENV-based) analyses.

The additional vegetation surveys of the Georgetown analogue site in 1 above were conducted between 30th August and 5th September 2010. Fifty sites were selected, with the site locations generated randomly in ARCMap and stratified within five different 5 m contour intervals. Consistent with previous surveys, at each site, every tree and shrub within a 20 m x 20 m plot above 2 metres in height was identified to species level. Height and diameter of each stem at breast height was measured. Canopy cover and ground cover (green grass, dry grass, leaf litter, wood, bare/burnt ground and resprout) along five transects within the plot were also measured with a densitometer.

Landform terrain variables for the 50 sites will be derived from a LiDAR image acquired by ERA in November 2010 for the Georgetown analogue study area. The imagery will be used to produce a very-high resolution DEM with a vertical accuracy of 0.15 metres and a horizontal accuracy of 0.25 m in areas of clear and open terrain. The DEM will be used to produce contours over the study area at an interval of 0.5 metres. In addition to the very high resolution provided by this dataset, a further key advantage of this DEM over earlier DEMs used for ecological modelling of the study area is that the provenance and accuracy of the dataset are clearly defined. This improves the confidence in the landform parameters that will derived for ecological modelling.

Analysis of the vegetation survey data gathered in 2010 is presently underway and is expected to be completed by March 2011. At the time of preparing this report, ***eriss*** had just received the LiDAR imagery from ERA. The more comprehensive plant and landform data that will be produced from the recently completed acquisitions should provide a more robust basis for landform design, for specification of closure criteria, and for design of post-rehabilitation monitoring programs.

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Charles Darwin University seed biology research

S Bellairs[1] & M McDowell[1]

Introduction

Charles Darwin University (CDU) staff are undertaking seed biology research to optimise germination of local native species to support the rehabilitation of the Ranger minesite. The project involves collaboration between the CDU researchers and staff from ***eriss***, Energy Resources of Australia (ERA), Kakadu Native Plant Suppliers (KNPS), Earth Water Life Sciences (EWLS) (now ERA), Greening Australia and Top End Seeds (TES).

ERA is required to sustainably establish a range of local native species on rehabilitated areas at the Ranger minesite. According to the applicable Environmental Requirements for rehabilitation, the company is required to establish an environment similar to the adjacent areas of Kakadu National Park, using local native plant species similar in density and abundance to those existing in adjacent areas of the Park.

To rehabilitate the areas impacted by the mine footprint, large numbers of plants comprising a broad range of species will be required. Therefore, effective techniques will be needed to source, store and germinate seeds, whether for direct seeding or for production of tube stock. KNPS are both providing seeds and producing tube stock for current rehabilitation areas using nursery facilities in Jabiru. They are collecting seeds from the local area to produce native plants that are adapted to local conditions.

Most Australian species germinate poorly from seeds unless specific seed treatments are applied, since they have seed dormancy mechanisms that prevent or delay germination, except in response to specific cues. However, pre-treatment information is lacking for the vast majority of NT species. Very little information is known about the seed biology of the local (ARR) species, including how to optimise viability of seeds during collection, how to store the seeds, or how to overcome dormancy and germinate the seeds (Bellairs 2007, Bellairs & Ashwath 2007). Tropical flora species are likely to differ in their seed biology responses to environmental cues compared to those of other Australian flora. Therefore, although information from southern Australian studies can be used as a guide, for most species, results obtained for similar southern species are unlikely to be directly applicable. KNPS is also identifying species that are difficult to germinate as part of its nursery operations.

The aim of the project has been to investigate seed viability, germination, dormancy and storage for up to 50 species that occur on the Ranger mine lease that have been identified as potentially important for rehabilitation. The project aims to develop protocols for effective seed storage and germination.

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Approach

Seed lots are being supplied by KNPS or from Top End Seeds and Greening Australia when KNPS is unable to provide supplies of seeds. While most seed lots have been obtained from KNPS, factors such as unusual rainfall patterns and cyclones have prevented KNPS from supplying seeds during some seasons and so alternate supplies have been obtained. It is also becoming evident that collection of seeds in sufficient quantities for many of the species is and will be more difficult than originally anticipated. TES collection data indicate that seed lots for some species have not been able to be collected for periods of up to seven years in spite of being able to be collected for several years before and after.

Testing of seeds is being carried out by CDU research staff under standard conditions, in laboratory incubators, to control light, temperature and moisture variations. In this manner, the most effective treatments may be accurately identified without having to use large quantities of seeds. Factors being tested include seed viability following collection, the types of dormancy mechanism(s) present, effective treatments to overcome dormancy and seed longevity under various storage conditions.

Progress to date

This seed biology project commenced in July 2006. ***eriss***, EWLS, KNPS and CDU project staff met and chose 50 priority species based on (i) their abundance in the Georgetown analogue sites (data provided by EWLS), (ii) their difficulty in propagation (information provided by Peter Christophersen (KNPS)), and (iii) preference for perennial species not likely to create a fire risk on the rehabilitation areas. This list was reviewed in December 2008 and a major review and re-prioritisation carried following the July 2009 project meeting.

Following the July 2009 meeting, a more objective prioritisation process was carried out, based upon the ranking of relative abundance of plant species of each life-form (trees, shrubs and ground cover), and the success to date in collecting and propagating the various species based upon the collective experience of EWLS, KNPS and CDU. For this ranking and scoring approach, EWLS used species quantitative abundance data from combined EWLS and ***eriss*** vegetation analogue surveys (Humphrey et al 2007). Separately for trees and shrubs, stem density values were used to rank the species. Then each species was assessed as to whether: (i) tubestock have been used for revegetation at Ranger and Jabiluka; (ii) whether seeds have been successfully germinated by KNPS or CDU; and (iii) whether seeds were available from KNPS. Note that the assessment that germination was observed and seeds were able to be collected was applied if this occurred on at least one occasion. It does not necessarily imply that germination will occur consistently or that seeds are reliably available. Thus in future, there may well be seed biology issues to be overcome with those species which are not considered a priority for seed biology research. Nevertheless, research needs for these species have a lower priority than species for which no knowledge is available. Herb and grass species were ranked according to plant biomass (g dry mass m^{-2}) data from Brennan (2005). These species were then assessed as to whether (i) plants have been recorded as natural recruits in revegetation at Ranger (primarily in the 'heritage' revegetation site on the east of the tailings dam wall); (ii) whether seeds have been germinated successfully by KNPS or CDU; and (iii) whether seeds are currently available from KNPS.

The most obvious observation from this reprioritisation is the dearth of seed biology information available for shrubs and ground cover species compared to the greater information available for trees. Thus the focus for seed biology research has shifted to shrubs and groundcover species. However, we do not imply that that seed biology issues have necessarily been overcome for all of species that have been removed from the priority list as a result of this reassessment. For reliable future rehabilitation, further testing of seed lots of all species to be used for trials is recommended, at the very least until quantitative data are acquired for three or more seed lots.

As of November 2010, the CDU researchers have investigated the seed biology of 41 priority species. Where CDU has not been able to source seed lots of the priority species, local seeds of other species in the same genus have been obtained and tested. To date 19 other species have been investigated. Priority species supplied by KNPS include 9 seed lots from 8 species in 2006, 14 lots from 14 species in 2007, no seed lots in 2008, 8 lots from 7 species in 2009 and 13 lots from 11 species in 2010. However, very few seeds (< 350) were available for four lots in 2010. Thirty-three other seed lots of priority species have been supplied by other organisations or collected by CDU and a total of 102 seed lots have been supplied or collected. Seed collection is difficult for many of these species and obtaining seeds is made more difficult by considerable seasonal variability in production of mature seeds and by impacts of fire. Obtaining sufficient seeds for large rehabilitation areas is likely to be challenging, even with greater resourcing. Where poor availability of seed stock is identified as a problem, other commercial sources of seed from outside of the ARR might be sought although provenance issues will need to be resolved.

Meetings have been held six monthly to present data and discuss progress. Other meetings between KNPS and CDU have been held in Jabiru to maximise information and technology transfer and assist with identification of species and timing of seed collection. The third annual report was provided in December 2009 (Bellairs & McDowell 2009) updating the previous reports (Bellairs & Crawford 2007, Bellairs & McDowell 2008) and the fourth annual report will be provided in December 2010. An Access database has been developed to store the seed biology data. The research work will continue to test new species and new seed lots of existing species. CDU research training program funding has been used to carry out more detailed work on some of the species.

All of the species have been able to be germinated with appropriate treatment, but some lots have had low viability, especially for some of the grasses. In some cases morphological features that indicate viable seeds have been identified. For some of the species there is substantial reduction in viability over a year but for others there is little decline in viability across several years and thus collection over several years and storage of seed lots is an option for obtaining sufficient seeds. Some species are time consuming to treat to allow germination and the efficiency of other means of propagation could be investigated.

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Bush food concentration ratio and ingestion dose assessment database

B Ryan, C Doering & A Bollhöfer

Introduction

Customary harvesting by local people of terrestrial bush foods from former mine impacted areas is ultimately likely to become more prevalent following the rehabilitation of the Ranger uranium mine. Bush foods encompass fruits, yams and other flora and fauna. Given the radiological nature of the site the ingestion pathway needs to be taken into account as part of the post rehabilitation radiological dose assessment. This dose assessment is an essential component of the International Standards of Practice (ICRP 2007) required for the operation and rehabilitation of radiological sites, including uranium (U) mines. The results from the assessment will also provide the means for communicating to the local people any issues associated with the consumption of bushfoods obtained from the site.

While little work has been done on the uptake of radionuclides into traditional bush foods across Australia as a whole (see for example Johansen & Twining, 2010), ***eriss*** has been measuring activity concentrations of natural-series radionuclides in local bush foods, including animal and plant tissues, in the Alligator Rivers Region (ARR) for more than 25 years. Early studies up to 1997 focused on aquatic environments, resulting in a relatively comprehensive knowledge base for aquatic bush food items in the ARR (Johnston 1987; Martin et al 1995; Martin et al 1998). This knowledge base has been supplemented by more recent studies on some terrestrial bush food items such as buffalo and pig (Martin et al 1998), fruit and yams (Martin & Ryan 2004) and other terrestrial flora and fauna (Ryan et al 2005, Ryan et al 2009). The primary purpose of these data collections has been to facilitate assessment of the possible ingestion dose to local Aboriginal people and provide assurance that there is no unacceptable radiation risk from bush foods obtained from areas affected by past and present uranium mining activities in the region. The data can also be used to calculate concentration ratios, which provide a measure of extent to which radionuclides can be taken up from water or soil into the edible portions of bush foods.

The large amount of data on natural-series radionuclides in bush foods that has been accrued by ***eriss*** in the past 25 years has been stored across various media types and file locations. The aim of this project is to firstly check and confirm the reliability of individual data points and then to collate all the validated data in one database to ensure its accessibility and longevity. This database will facilitate access to the material by other researchers across the organisation and outside, and considerably streamline the conduct of site specific assessments of potential ingestion doses to humans from the consumption of bush foods.

Methods

A Microsoft Access database is being developed as a central repository for existing and future ***eriss*** data on radionuclide activity concentrations in bush foods (ie plant and animals tissues) and in associated environmental media (eg soil and water). The design of the database will support desktop investigations of regional and site specific variability in radionuclide accumulation in local bush foods by allowing calculation of concentration ratios. It will also support assessments of the potential committed effective dose to a critical group (or ‘Representative Person’) from the ingestion of natural-series radionuclides in bush foods.

Results

A fit-for-purpose database framework has been developed using ***eriss*** in-house expertise.

More than 1500 individual records containing information on radionuclide activity concentration in various plant and animal tissues, as well as in environmental media, have been retrieved from original source files, quality checked, and entered into the database. All the records have associated geospatial information to enable spatial analysis at a later date and to facilitate incorporation of the data into a bush food geospatial information system (see paper by Walden in this volume, pp 133–135).

The radionuclides that have been analysed in tissue and media samples over the past 25 years are shown in Table 1. It should be noted, however, that not all radionuclides were analysed in all samples. Ryan et al (2005) found that in terms of radiological dose for a mine rehabilitation situation ^{226}Ra, ^{210}Pb and ^{210}Po are of greater importance than the thorium and uranium isotopes. Consequently, analyses for uranium and thorium are conducted less frequently because of their lower dose conversion factors. Radioactive equilibrium of progeny radionuclides in the dietary items is assumed for dose assessment purposes, unless direct measurements of progeny are available.

Table 1 Radionuclides analysed in tissue and media samples of the Alligator Rivers Region

^{238}U-series	^{232}Th-series	^{235}U-series
^{238}U	^{232}Th	^{227}Ac
^{234}U	^{228}Ra	
^{234}Th	^{228}Th	
^{230}Th		
^{226}Ra		
^{210}Pb		
^{210}Po		

The database can be queried to return statistical information on concentration ratio (including arithmetic mean, standard deviation, range and number) for a specified combination of biota, tissue, radionuclide and site, eg buffalo-kidney-^{210}Po-Ranger. It also returns the individual records of radionuclide activity concentration in the tissue and media samples that underpin the calculation. A screenshot of the front-end of the database for concentration ratio queries is shown in Figure 1.

The concentration ratio, expressed as the ratio of radionuclide activity concentration in tissue to that in environmental media, is a key parameter used to predict the uptake of radionuclides by human and non-human biota from the environment. Human dose assessment methods use tissue-to-media concentration ratios to estimate radionuclide accumulation in the edible parts of food items. Non-human biota dose assessment methods use organism-to-media concentration ratios to predict the average whole body radionuclide uptake in plants and animals. The main purpose of measurements by ***eriss*** of radionuclide activity concentration in bush foods has been to derive tissue-to-media concentration ratios for various radionuclide/food combinations. Although this is primarily in order to assess potential ingestion dose to humans, the measurements are also potentially useful to ascertain organism-to-media concentration ratio for some non-human species. This is needed to determine radiation dose rates to plants and animals using existing biota dose assessment tools such as ERICA and RESRAD-BIOTA (Beresford et al 2010).

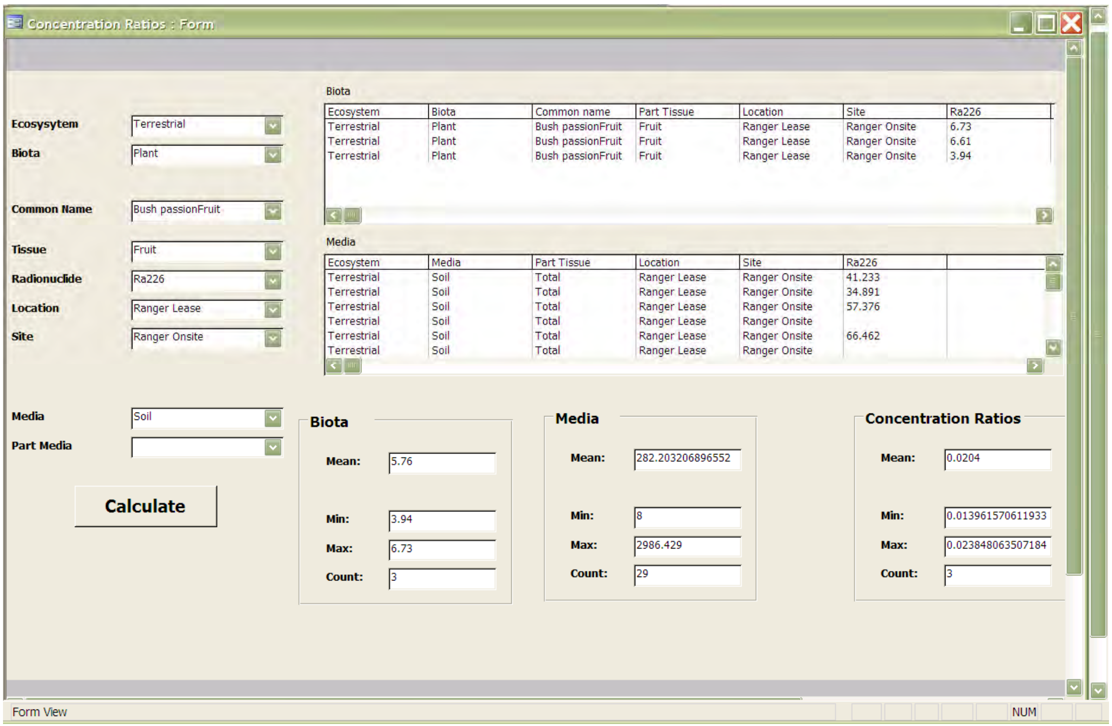

Figure 1 Screenshot of data base concentration ratio and activity concentrations [mBq·kg^{-1}] in food stuffs (*Passiflora foetida* collected on Ranger lease as an example) in the wet dry tropics

Whilst developing various ingestion dose models and scenarios pertinent to the Alligator Rivers Region it has become apparent that the models should be site specific and take account of local dietary habits, land use and the land use expectations of local people. It has also become apparent that the use of generic concentration ratios for quantifying radionuclide accumulation in food items, such as those given by the International Atomic Energy Agency (IAEA 2010) that are based largely on Northern Hemisphere species, may not be appropriate to the situation in the Alligator Rivers Region. The IAEA, for instance, suggests an average concentration ratio for radium in the tissue of freshwater fish of 4 (IAEA 2010), whereas average tissue-to-water concentration ratio for radium for fish in the Alligator Rivers Region can range from 60 (Archer fish, *Toxotes chatareus*) to 1000 (Bony bream, *Nematalosa erebi*) (Martin et al 1995). Consequently, the models used to assess ingestion dose to humans should make use of region specific concentration ratios.

Hypothetical diets for local people have been established by combining information from several sources including a questionnaire developed by ***eriss*** and distributed to local Aboriginal people in 2006, information provided by a local supplier of meats to Aboriginal outstations and data gained from the ***eriss*** Kakadu bush food project over the last 13 years (Ryan et al 2008). This dietary information, combined with concentration ratios and activity concentrations, can be used to estimate the pre-mining, operational and post-rehabilitation ingestion doses to people at various locations in the ARR and surrounding regions.

Future work

The ingestion dose assessment calculation function of the database is currently being refined to allow estimates of ingestion doses to be made from existing tissue concentration ratios and model diets. It is also anticipated that an additional add-on to the database will be developed to allow retrieval of whole body-to-media concentration ratios that are needed to determine radiation doses to fauna and flora using existing biota dose assessment tools such as the ERICA tool.

Acknowledgments

Andrew Esparon for coding the database and all personnel involved with sample collection, preparation and analysis of bush foods over the last ~30 years.

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The Bushtucker database

D Walden

Introduction

For the past 30 years the concentrations of radionuclides in traditional bushfoods, known as 'bushtucker', have been measured by SSD from many locations for a wide range of species. The database currently contains in excess of 1500 records. It continues to be updated on an annual basis with data produced from routine monitoring programs and opportunistic collection (for example, from areas associated with the rehabilitation of the old mine workings in the South Alligator River Valley or provided by, or with the assistance of, local indigenous people) of samples. Although the methodology and findings have been the subject of several journal and conference papers, as well as previous SSD Annual Reports and Research Summaries, there has been no prior integration of this material in an easily accessible spatially enabled format.

Newly available spatial technologies such as the 3-Dimensional virtual Earth/Globe viewing programs (hereafter referred to as virtual globe) such as Google Earth, Arc Explorer and Arc Globe offer a means to integrate and display this complex information in a user friendly format that is able to be accessed by both the local people of the area and the wider public. **.** The virtual globe software is free for non-commercial applications and is easily downloaded from the internet, making it generally available to the community.

The virtual globe environment will allow the user to navigate around the Alligator Rivers Region using high resolution satellite imagery, and 'fly' to sampling sites to view available information. This gives the user a unique perspective of the terrain and appreciation of where the sampling sites are located relative to uranium mines, populated places and favoured bushtucker hunting and gathering sites. This approach to data presentation could be followed for other spatially-based datasets (for example, soils and vegetation) acquired by ***eriss***.

Scope of work

Three platforms are being developed – two for public viewing and a third for internal research use only. The first public viewing product is a Keyhole Mark-up Language file (KML) that contains all of the virtual globe features such as callout boxes, terrain flyovers, audio narration and internet page hyperlinks. This compressed file will be available as a standalone download from the internet or on a CD if necessary. It is simply loaded into the virtual globe program and the information tour starts automatically. KML files can also be loaded into web-based viewers such as Google Maps, Bing Maps, Yahoo Maps and Whereis.com to name a few.

The second public viewing product for non-internet users is a movie that has been created in a virtual globe environment and copied to a DVD. The movie will take the viewer on a tour of the bushtucker sampling sites with audio narration and 'pop-up' information appearing along the way.

There is a need for caution when presenting data of this nature to the public because interpretation of the results is usually complex and there is potential for confusing or misleading interpretations to be made from individual data points. In the event of data having been published previously in reports or papers, links will be provided to this reference source since they typically contain more detailed explanations and interpretations of the data.

The third 'internal use only' product would be similar to the first product but would contain all of the detailed radionuclide information. This product will enable SSD research staff to readily locate all of the available information in a spatial context and facilitate the use of the data across the Division.

Progress/results

Work on development of the KML files and construction of the internet pages is complete except for photographs of some species. These components provide the basis for all three outputs from this phase of the project. Inclusion of a narration audio (with the possibility of developing a local language version) is being investigated to enhance the quality of the presentation and help guide users through the myriad of sites, species sampled for radionuclides and other features in the presentation.

Figure 1 shows a sample Google Earth image with a callout box containing a graphic, text and links to, in this case, the catchments and major sites where bushtucker has been sampled. Subsequent links point to the species sampled (see Figure 2 for example). Other links provide information about the uranium deposits/mines in the region, a map of all sites and samples collected at them, and a virtual tour of the region and the bushtucker samples.

The dialog box displayed in Figure 2, contains icons that provide a clickable link to web pages with information on measured radionuclides, aspects of the species' biology and available information on how local Aboriginal people may catch and prepare the food. Both English and Gundjeihmi language names for the species are provided. Gundjeihmi is the major local indigenous language group.

Figure 1 Google Earth snapshot with an information callout box and customised icons

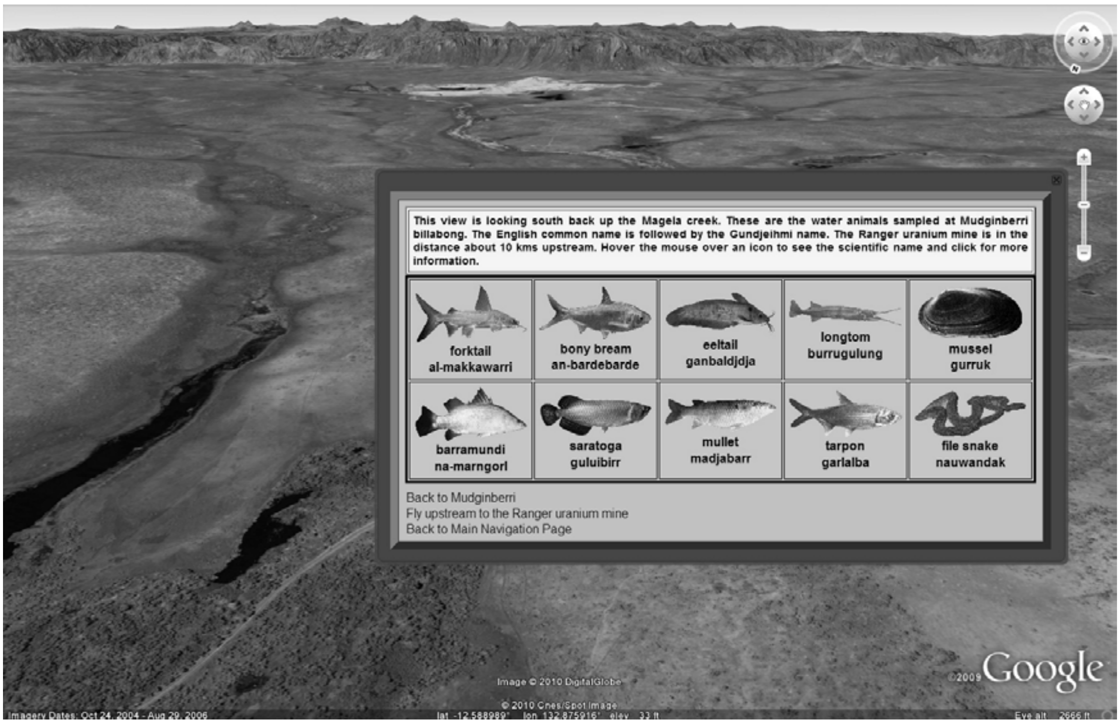

Figure 2 Another Google Earth snapshot showing a callout box with icons for the bushtucker species sampled at that site. Ranger uranium mine and the Arnhemland escarpment can be seen in the distance.

Steps for completion

1 Script the content for the narration audio.

2 Add the audio to the presentation.

3 Produce a DVD movie of the Google Earth tour.

4 Commence liaison with the Web and Intranet Management Section (WIMS) of the Department regarding the uploading of the associated bushtucker websites to the Department’s internet site.

5 Upload the KML files to the SSD internet.

Remote sensing framework for environmental monitoring within the Alligator Rivers Region

R Bartolo, G Staben, K Pfitzner, J Lowry & A Beraldo

Introduction

Broad scale characterisation of landscapes in the Alligator Rivers Region (ARR) is required to be able to place the land surface status of operating and rehabilitated minesites into a regional context. To date there is little information on landscape ecology variables (in the context of rehabilitation, close out and known risks and threats) and their scale of interaction. Application of remote sensing technologies to address this knowledge gap requires the development of a remote sensing monitoring framework. The framework will provide the basis for most efficiently and cost effectively acquiring the required data.

Progress/Results

In May 2010, a systematic remote sensing data capture, incorporating full ground control and coincident spectral data collection, was done of the Magela floodplain and Ranger uranium mine. The data capture was undertaken in collaboration with the Tropical Rivers and Coastal Knowledge research hub's Theme 5.3 project (Food webs and biodiversity: river–floodplain food web studies). Three World-View 2 images covering 730 km^2 of the Magela Creek catchment were acquired. Table 1 shows the spectral bandwidth resolution of the satellite's sensor. The spatial resolution supplied is 0.5 m for the panchromatic band and 2.0 m for the multispectral bands.

Table 1 Spectral bands for the world-view 2 sensor

Sensor band	Wavelength
Panchromatic	450–800 nm
Coastal	400–450 nm
Blue	450–510 nm
Green	510–580 nm
Yellow	585–625 nm
Red	630–690 nm
Red Edge	705–745 nm
Near-IR$_1$	770–895 nm
Near-IR$_2$	860–1040 nm

The following scene parameters and data format were requested from the supplier: nadir angle less than 20º; cloud cover threshold 0–15%; and 16 bit data format. The potential capture dates for the imagery provided were May 6, 11, 14, 22 and 25. In order to ensure all required field and calibration data were available at the time of image capture, locational positioning and spectral calibration needed to be collected. Therefore, ground targets with accurate known locations had to be deployed prior to 6 May and suitable spectral calibration targets had to be in position, with spectral characteristics measured as close as possible to the time of image acquisition.

Historically, there has been poor ground control for acquiring remote sensing data for the Magela Creek floodplain. High accuracy ground control is especially important in this case given the very low topographic relief of the area. This was achieved for the current capture by collecting 33 ground control points across the image acquisition area (Figure 1a).

Twenty-seven 3.5 x 3.5 m square silver-coloured tarpaulins were positioned on the ground prior to the image acquisition window and six image objects (features such as cross roads evident in previous image data) were selected for measurement. The centre of each of the tarpaulins and image objects were measured with a dGPS (Figure 1c) to within 12 mm x,y accuracy. These ground control points enable accurate orthorectification of the imagery.

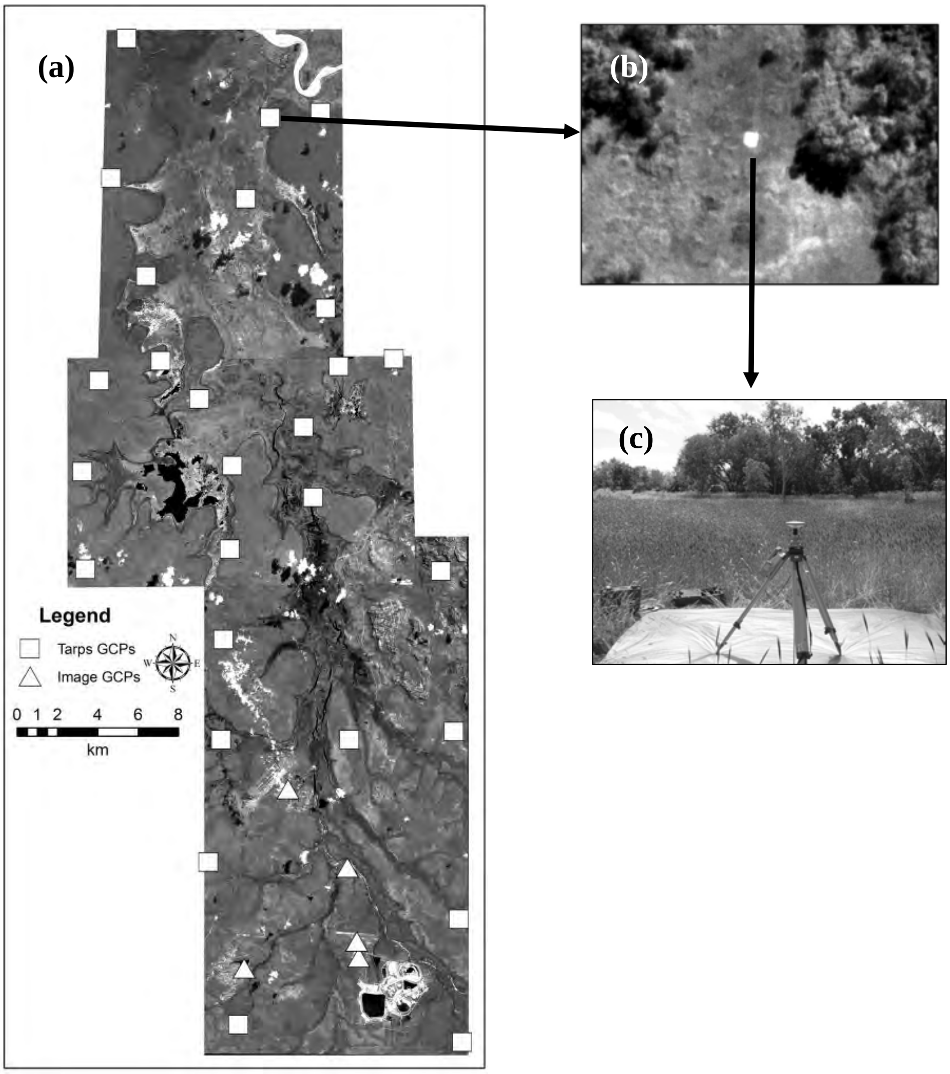

Figure 1 Distribution of Ground Control Points (GCPs) across the WV-2 imagery, (b) example of a tarpaulin GCP (site 2) captured by the WV-2 satellite in the panchromatic band, (c) collection of the dGPS data from the site 2 GCP

Atmospheric correction of satellite imagery using an empirical line method requires that high quality spectral measurements of suitable ground targets are acquired as close as possible to the time (in this case 11:15 am on May 11 and 22) of image acquisition. After testing, using laboratory measurements of reflectance spectra, the suitability of various industrial products as ground targets, four materials were chosen to represent dark and bright targets. These were: black synthetic upholstery material (2% reflectance); silver plastic weave tarpaulin (23% reflectance); white plastic weave tarpaulin (67% reflectance); and Tyvec, a building insulation product (95% reflectance). The targets needed to be sufficiently large enough to be detected in the satellite imagery (Figure 2).

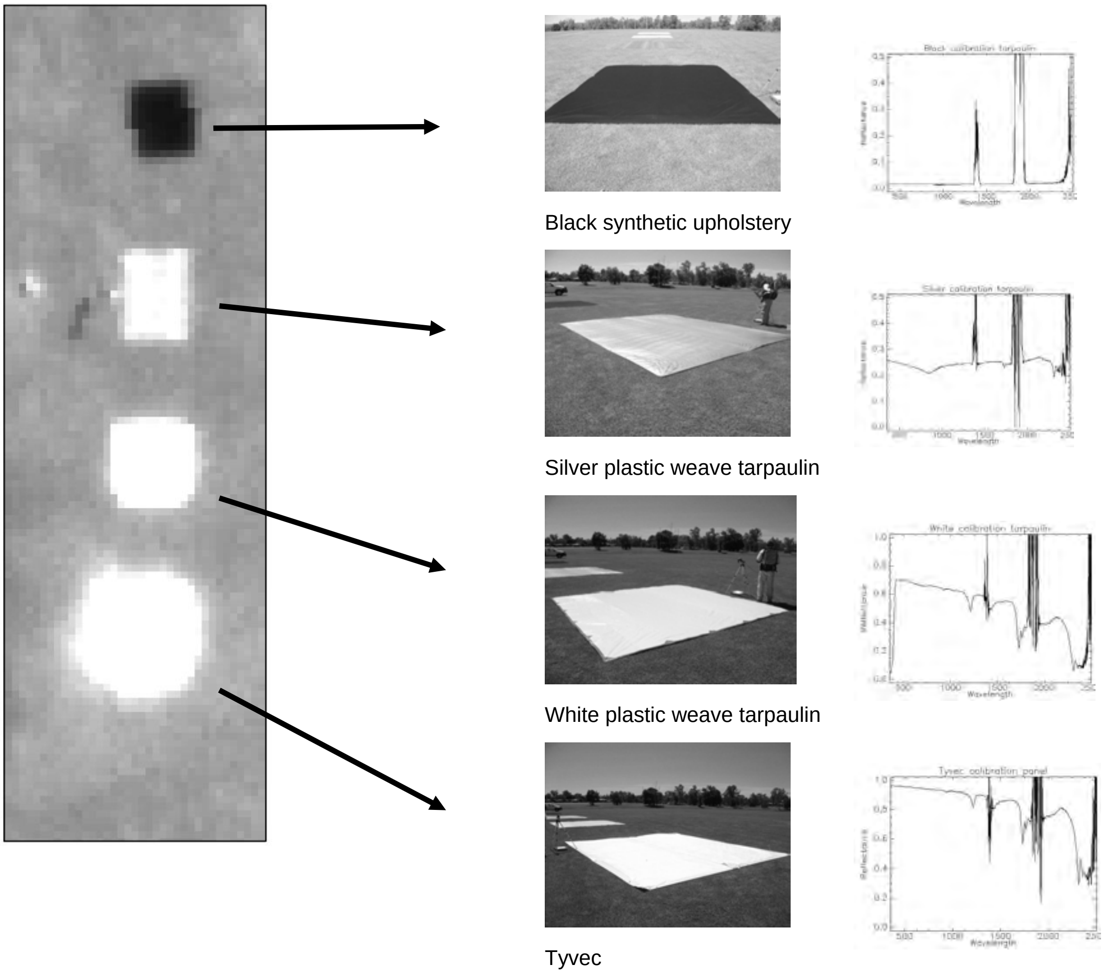

Figure 2 Left: Calibration tarpaulins as captured by the WV-2 satellite (panchromatic band, Middle: photographs of targets set up on Brockman Oval at Jabiru, Right: the spectral signal of each tarpaulin as measured by the ASD SpecPro spectrometer

Prior to the 11:15 am satellite overpass, the four target types were deployed on Brockman Oval, Jabiru. During the satellite overpass, atmospheric solar irradiance data were collected using the ASD FieldSpecPro Spectrometer. Following the measurement of the solar irradiance data, multiple measurements of each of the four ground targets were collected. To assess the accuracy of the empirical line method for calibration of the WV-2 data, spectra of various invariant targets such as deep water, bare earth and well-maintained golf greens were also collected. These spectra were measured on the day of the overpass.

The majority (95%) of the areas of the three requested scenes were captured with the specified scene parameters (nadir angle of 13.8º and total cloud cover <2%) on 11 May. The remaining 5% of one of the scenes was captured on 22 May (nadir angle of 11.6º). All 27 ground control tarpaulins were visible in the imagery. Figure 1 (b) shows an example of how ground targets appear in the imagery.

Future work

To produce a quality final product suitable for high resolution mapping of vegetation and habitat types, orthorectification of the imagery, atmospheric correction and the development of mapping applications will be required. This work will be done over the next year.

Part 3: Jabiluka

The Jabiluka site is currently in long-term care and maintenance. Several projects relating to assessment of long term changes in morphology of the stream channel in Swift Creek are currently being finalised and will be reported on in the next annual ***eriss*** Research Summary.

Part 4: Nabarlek

There are no research papers this year in the Nabarlek key knowledge needs theme. The taking over of management of the site by Uranium Equities Limited and the requirement for conduct of monitoring and progressive rehabilitation activities as part of the mine management plan have meant that the involvement of SSD has been reduced following completion of the suite of projects that had been initiated to define for stakeholders the rehabilitation status of the site.

Part 5: General Alligator Rivers Region

Assessing the success of remediation works at former uranium mining and milling sites in the South Alligator River valley

C Doering, B Ryan, A Bollhöfer, J Sellwood, T Fox & J Pfitzner

Introduction

In the 2009 dry season major works commenced in the South Alligator River valley (SARV) to rehabilitate remnants of historical uranium mining and milling activities in the area. The works included the removal of contaminated material from areas affected by historical mining (Sleisbeck, El Sherana and Palette mines) and milling (Rockhole Residues) activities. Other areas that were remediated included the old containments at the South Alligator village, the El Sherana village, and the El Sherana weighbridge. At these sites contaminated material that had previously been buried was excavated and transferred to a purpose built containment located on the site of the old El Sherana airstrip. The remediation works were finished prior to the start of the 2009–10 wet season, with approximately 20 000 m³ of material having been moved to the new containment. The locations of the main historical mining and milling sites are shown in Figure 1.

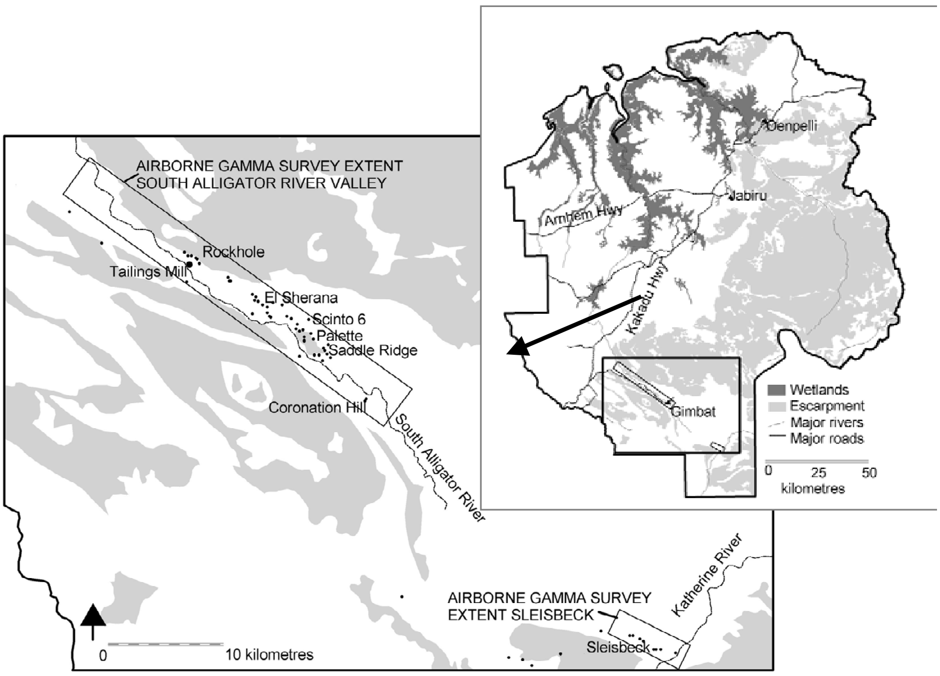

Figure 1 Alligator Rivers Region, with a detailed excerpt of the southern area showing the extent of two airborne gamma surveys conducted in 2000 and 2002, the location of known uranium anomalies (from MODAT database) and some historic mining and milling areas (Supervising Scientist 2003)

Descriptions of the historical mining and milling sites, and past remediation activities in the area, are contained in Fisher (1968), Waggitt (2004) and Bollhöfer et al (2007, 2009a&b).

The latest remediation works were carried out to address the terms of the 1996 lease agreement between the Gunlom Aboriginal Land Trust and the Director of National Parks.

This lease agreement required the Director of National Parks to implement an environmental remediation plan for all historic minesites and associated workings in the SARV. The remediation plan was managed by Parks Australia, with SSD providing specialist assistance with the radiological assessment of the sites.

The results from the radiological assessments and other investigations carried out by SSD informed the development of a remediation strategy for the SARV. The gamma dose rate guideline value to be used as the remediation threshold for historic mining and milling sites in the SARV was set at 1.25 ± 0.25 µGy·hr^{-1} (approximately 10 times higher than background levels). This guideline value was derived on the basis of being able to clearly distinguish, by field measurement, the radiological signal of mining-derived material from the regional background, whilst ensuring that likely annual doses to the public are well below the reference level (10 mSv/y) imposed by ARPANSA for remediated sites in the South Alligator River Valley.

Radiological assessments of the Rockhole Residues site, and Palette and El Sherana mines were conducted before remediation and results are published in Bollhöfer et al (2007, 2009a&b). Additional historical information on the Palette and El Sherana minesites can be found in Bollhöfer et al (2010).

This paper presents the results of gamma dose rate surveys conducted in the 2010 dry season at the Rockhole Residues, Palette and El Sherana sites (Doering et al in prep) and compares the results with those from the earlier pre-remediation survey to assess the success of the remediation works.

Methods

Gridded gamma surveys were conducted at the Rockhole Residues site and at Palette (top bench only) during July 2010, and at El Sherana (pit and access tracks, top area to the north of pit, and top areas to the south and west of pit) during September 2010. Two environmental dose rate meters of the same type were used for the surveys at each site.

The spacing of the measurements was approximately 15 m at the Rockhole Residues site, 5 m at Palette and 5–10 m at El Sherana. The geospatial coordinates – eastings and northings – of each measurement point at each site was determined by global positioning system (GPS) in the WGS84 coordinate system within Zone 53. At the Rockhole Residues site only, the visual boundary of the remediated area was walked on foot and the trace recorded using GPS.

Results

Rockhole Residues

Figure 2 shows the location of the individual measurement points and the magnitude of the calculated absorbed gamma dose rates (including the cosmic component) at the Rockhole Residues site before and after remediation. The figure also shows the visual boundary of the remediated area, overlaid on a Quickbird satellite image of the area acquired in 2004. It can be seen that absorbed gamma dose rates have been substantially reduced following remediation of the area.

The total area that was remediated at the Rockhole Residues site in 2009–10 was approximately 2.5 ha. None of the 150 dose rate measurements taken across the area in 2010 exceeded the specified radiological threshold value of 1.25 ± 0.25 µGy·hr^{-1}. Figure 3 provides a frequency and probability plot for the dose rates measured at the site. The dose frequency plot for the Rockhole Residues site (Figure 3) exhibits a lognormal distribution. From the probability plot it can be concluded, with 95% confidence, that 99.9% of the dose rates measured at the site were below the guideline value.

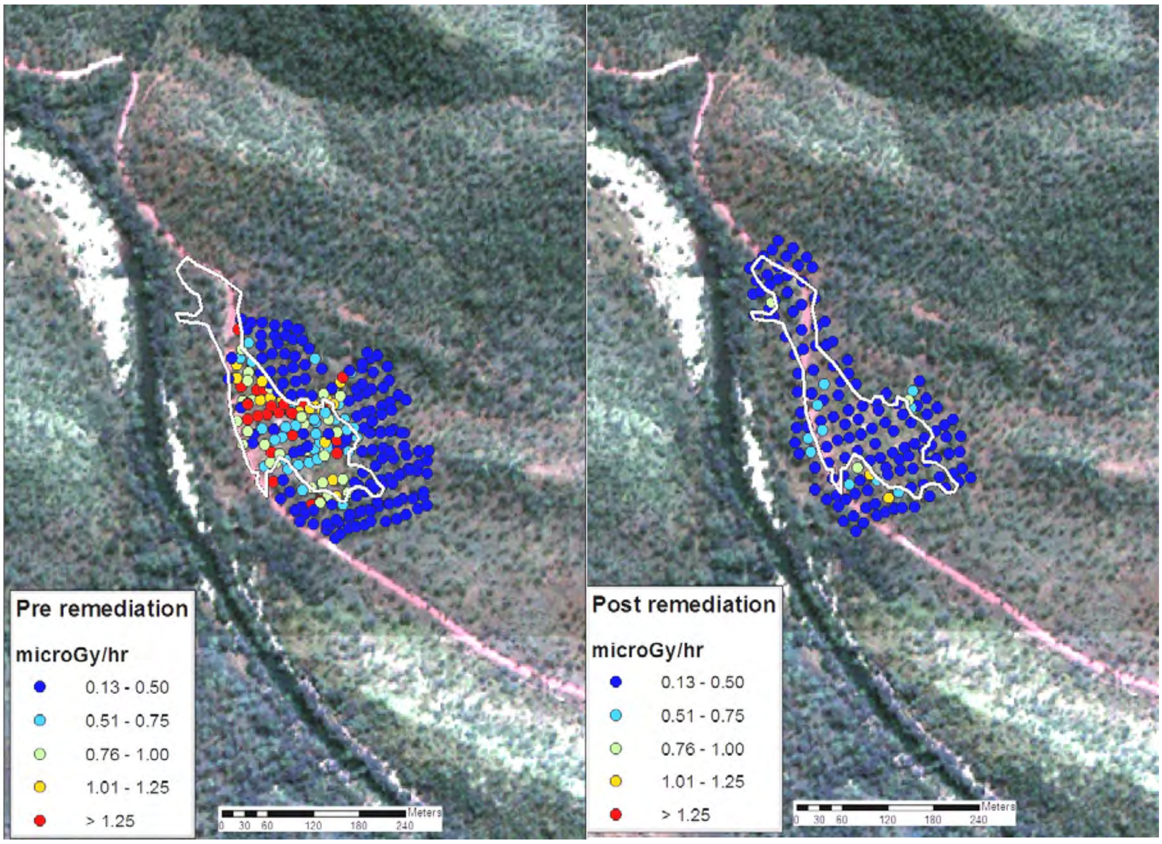

Figure 2 Pre- and post remediation gamma dose rates at the Rockhole Residues site overlaid on a Quickbird satellite image. The white line shows the approximate outline of the remediated area.

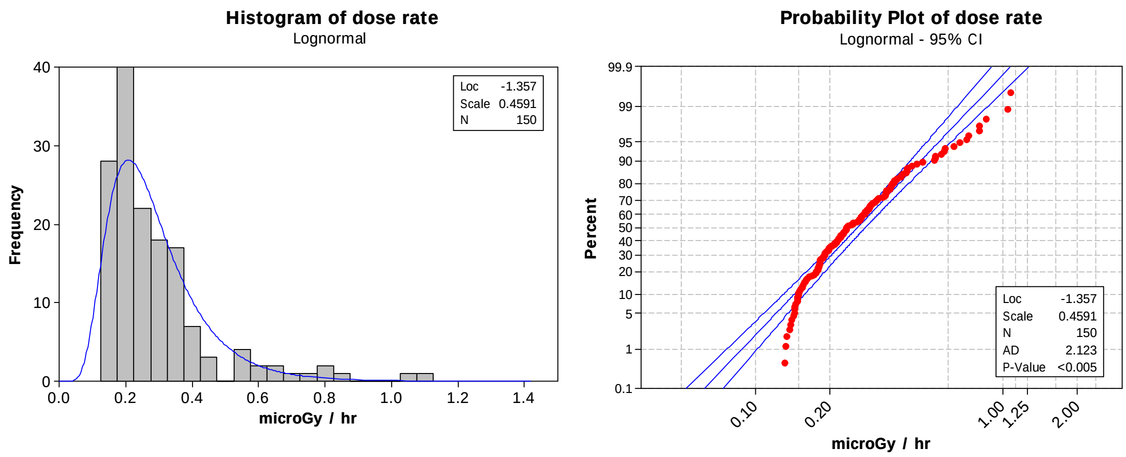

Figure 3 Histogram and probability plot of the dose rates measured at the remediated Rockhole Residues site

Palette mine

Figure 4 shows the pre- and post-remediation gamma dose rates measured at the Palette site in 2008 (Bollhöfer et al 2009b) and 2010, respectively. Whereas in 2008 the lower bench was easily accessible, the track to the lower bench was not accessible in 2010 due to the remediation works and the presence of large boulders. Consequently, only the top bench (to the west or left of image) was surveyed as it was considered unlikely that the lower bench will be accessed by people for habitation for extended periods of time.

The dose rate frequency plot for Palette (Figure 5) appears to define a lognormal distribution. The probability plot for the site indicates that 90% of the locations measured post remediation have gamma dose rates less than the guideline value of 1.25 ± 0.25 µGy·hr^{-1}, compared to only 25% pre-remediation (Bollhöfer et al 2009b). Generally, readings above 1 µGy·hr^{-1} correspond to locations near the cliff face on the top bench. There was one measurement above the guideline value, which was due to the presence of a single small piece of mineralised material (approximately 5 cm^3), believed to be pitchblende, lying on the ground surface.

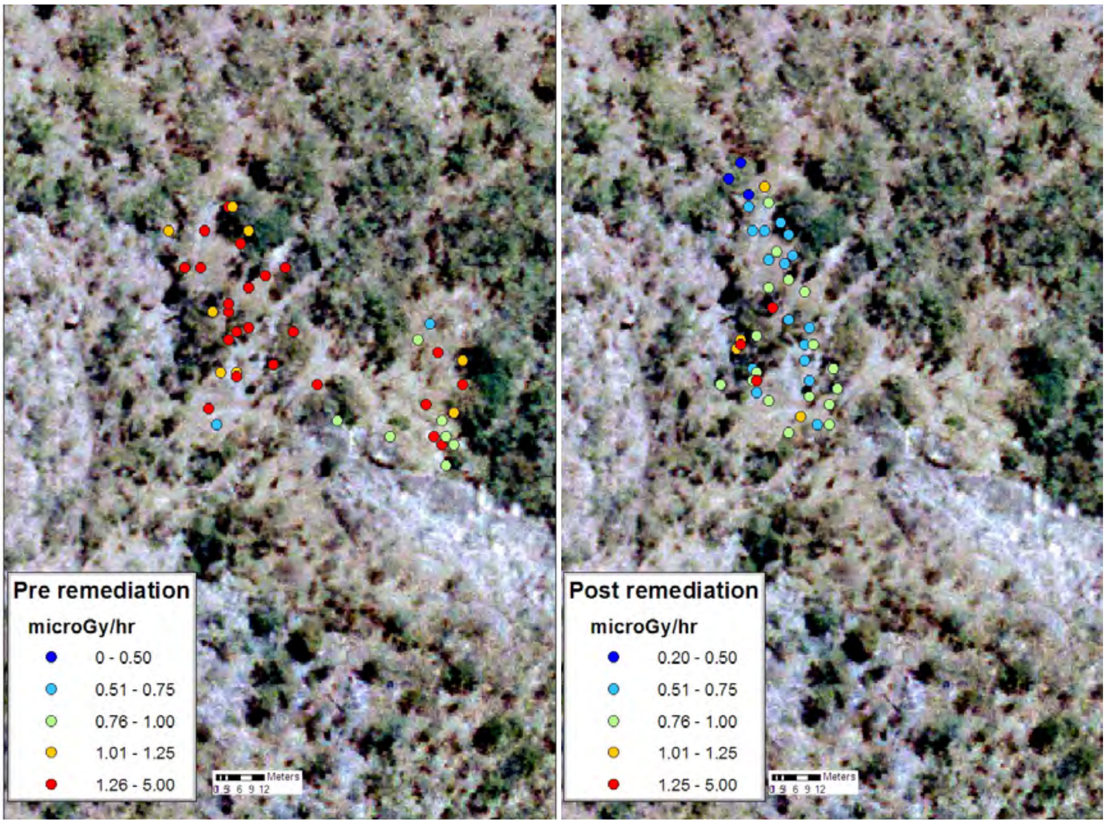

Figure 4 Measured pre- and post remediation gamma dose rates at Palette mine overlaid on an aerial photograph of the area

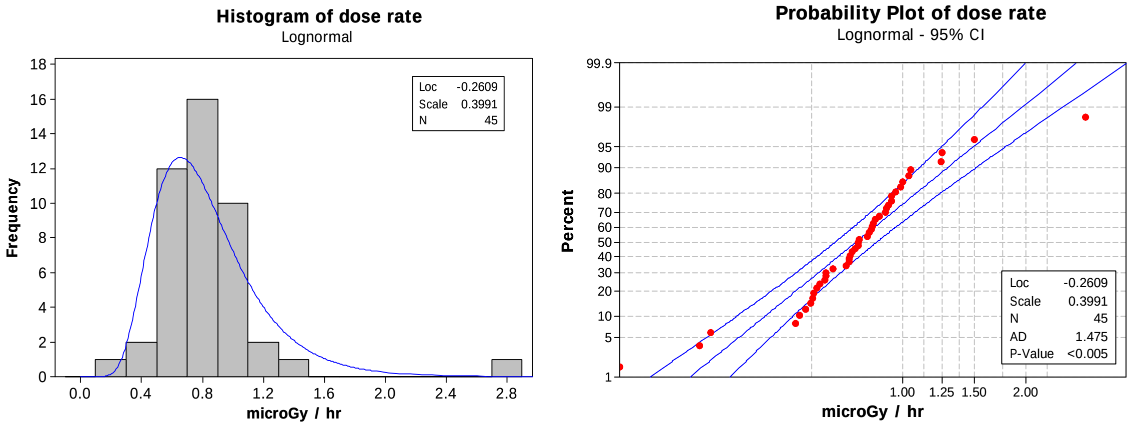

Figure 5 Histogram and probability plot of the dose rates measured at the remediated Palette mine

El Sherana mine

The highest pre-remediation dose rate on top of the El Sherana pit (to the north) in 2008 was 14 µGy·hr^{-1}, measured over a concrete pad that had supported a battery used to crush some of the high grade ore mined at the site. The next highest readings were obtained from an area without noticeable infrastructure but containing a number of rock and rubble piles (see Bollhöfer et al 2009 for details). Figure 6 compares pre- and post-remediation gamma dose rates measured at El Sherana. Gamma dose rates on top of the pit were substantially reduced after remediation. Only one reading in 2010, at 2.7 µGy·hr^{-1}, was above the guideline value and was associated with a number of small fragments of mineralised material.

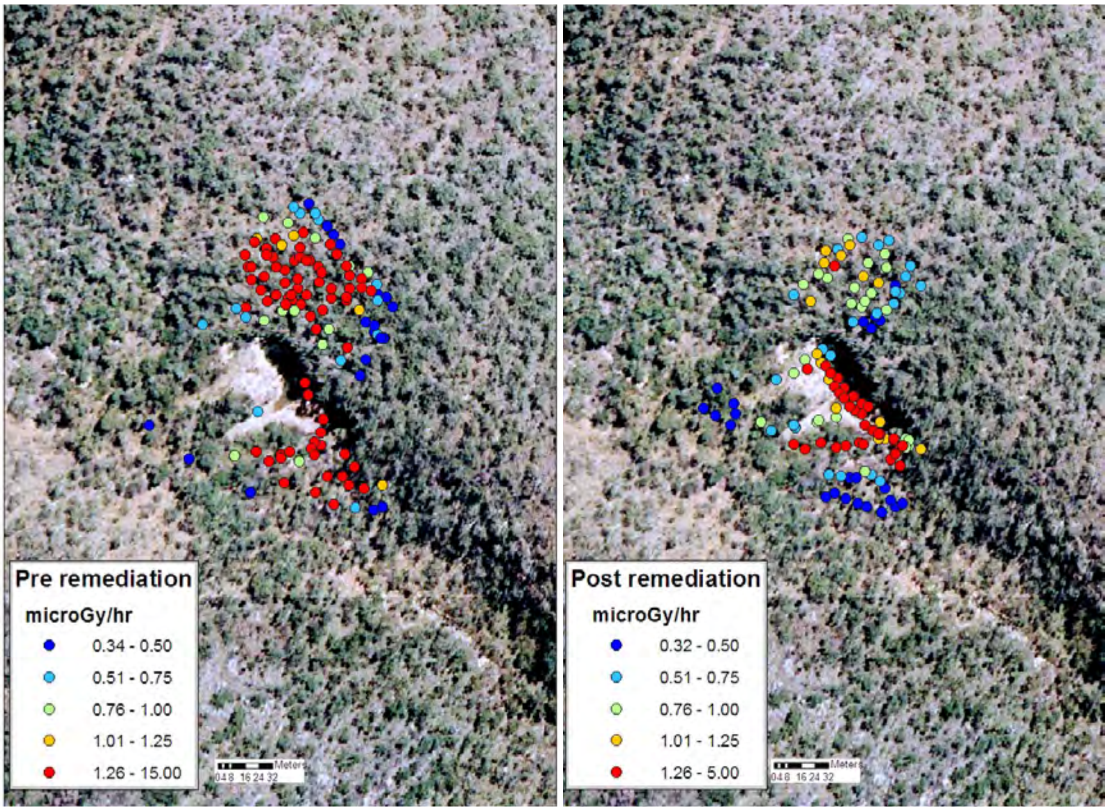

Figure 6 Measured pre- and post remediation gamma dose rates at El Sherana mine overlaid on an aerial photograph of the area

The bottom of the pit was surveyed in 2008 prior to remediation, and some areas were found to have elevated gamma signatures. These areas consisted of two waste piles and four benches in the pit wall to the southeast of the pit. The bench on top of the pit wall exhibited gamma dose rates of approximately 1 µGy·hr^{-1}. The remaining three benches, the bottom of the pit, and the two waste piles exhibited average gamma dose rates of about 2 µGy·hr^{-1} or more. In particular and prior to remediation, there was a small area of mineralisation accessible from one of the benches that exhibited absorbed gamma dose rates of above 7 µGy·hr^{-1}. As part of the remediation at El Sherana, the material from the two waste piles was shifted and pushed against the benches to the southeast of the pit, thereby covering the small area of mineralisation and reducing average gamma dose rates in the area.

Figure 6 shows that post-remediation gamma dose rates were lower after removal of the two waste piles south of the pit. However, dose rates were still generally higher in the pit and along the upper small access track compared with other areas, due to remnant mineralised material. The highest dose rates (3–4 µGy·hr^{-1}) were measured immediately to the southeast of the pit, in an area that was not subject to remediation.

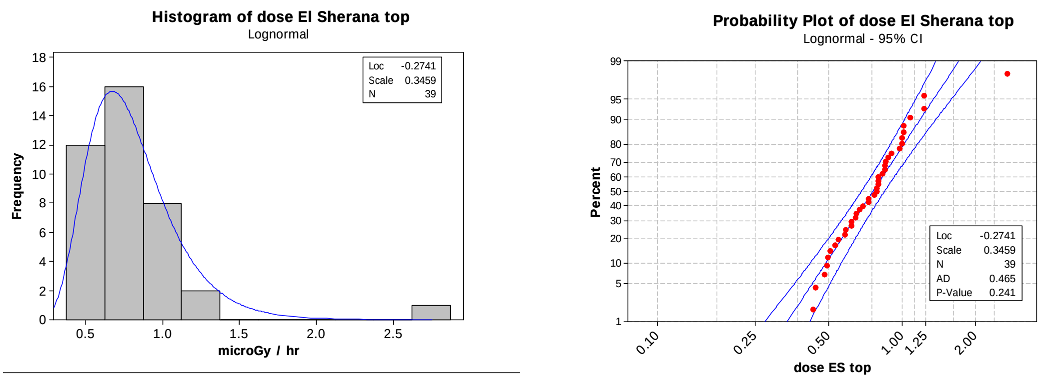

Figure 7 Histogram and probability plot of the dose rates measured on top of the remediated El Sherana mine

Figure 7 shows frequency and probability plots of the dose rates measured on top of the remediated El Sherana mine. Assuming a log-normal distribution of the absorbed gamma dose rate measured on top of the El Sherana pit, approximately 95% of the locations investigated post remediation are characterised by levels below the 1.25 ± 0.25 µGy·hr^{-1} guideline value, compared with only 40% pre-remediation (Bollhöfer et al 2009b).

The average gamma dose rate on top of the pit has been reduced substantially and typical post remediation values are 0.7–0.8 µGy·hr^{-1}. At the bottom of the pit and on access tracks not subjected to remediation, values have not changed and are typically around 1.2–1.4 µGy·hr^{-1}.

Summary

Table 1 provides a statistical summary of the pre- and post-remediation absorbed gamma dose rates measured at the Rockhole Residues site, and the remediated Palette and El Sherana mines.

It can be seen that remediation generally reduced the gamma dose rates at the remediated sites by about a factor of 2, to values well below the 1.25 ± 0.25 µGy·hr^{-1} threshold. Mean gamma dose rates within the El Sherana pit and access tracks, which were not subjected to remediation activities, are close to the guideline cleanup threshold value of 1.25 ± 0.25 µGy·hr^{-1}.

Table 1 Statistics for post- and pre-(brackets) remedation absorbed gamma dose rates measured at the Rockhole Residues, Palette and El Sherana sites

Statistic	Rockhole Residues	Palette (top bench)	El Sherana (pit and access)	El Sherana top (N of pit)	El Sherana (S and W of pit)
Median	0.23 (0.35)	0.78 (1.66)	1.27 (1.87)	0.79 (1.47)	0.40
Mean	0.29 (0.57)	0.83 (1.88)	1.39 (1.85)	0.81 (2.62)	0.44
Geo mean	0.26 (0.40)	0.77 (1.66)	1.28 (1.47)	0.76 (1.57)	0.43
Min	0.13 (0.13)	0.21 (0.56)	0.70 (0.33)	0.44 (0.34)	0.32
Max	1.08 (3.89)	2.79 (4.34)	4.07 (7.28)	2.70 (13.63)	0.76
N	150 (258)	45 (20)	60 (60)	39 (90)	23

1 Pre-remediation values for the Rockhole Residues site were calculated from Bollhöfer et al (2007), and those for Palette and El Sherana sites from Bollhöfer et al (2009b).

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Bollhöfer A, Ryan B, Fawcett M, Turner K & Jones D 2009b. Remediation of past uranium mining activities in the South Alligator River valley. In ***eriss*** research summary 2008–2009. eds Jones DR & Webb A, Supervising Scientist Report 201, Supervising Scientist, Darwin NT, 194–200.

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Waggitt P 2004. Uranium mine rehabilitation: The story of the South Alligator Valley intervention. *Journal of Environmental Radioactivity* 76, 51–66.

Research consultancies

This section contains a summary of non-uranium mining related research consultancies carried out by ***eriss*** during 2009–2010. Most of these reports are commercial-in-confidence and are not available for public release.

List of non-uranium mining related research consultancies

Boyden J 2010. Waterbirds data and related environmental datasets collated by SSD for the Northern Australia Water Futures Assessment (Ecological assets sub-project). March 2010.

Jones, DR, Humphrey C & van Dam R 2010. Review of water, sediment and biota data for Page and Lawn Hill Creeks. Commercial-in-Confidence Report for Metals and Mining Group – Century, February 2010.

Ryan B & Bradley F 2010. Preliminary report into the characterisation of groundwater at the Rum Jungle minesite: Consultancy report for Department of Resources, Energy and Tourism. Supervising Scientist, Darwin.

Ryan B, Bradley F & Bollhöfer A 2010. Final report into the characterisation of groundwater at the Rum Jungle minesite: Consultancy report for Department of Resources, Energy and Tourism. Supervising Scientist, Darwin.

Ecotoxicological assessments of discharge waters from Cosmo Howley, Pine Creek, Tom's Gully and Brocks Creek Project Areas

AJ Harford & RA van Dam

The Environmental Research Institute of the Supervising Scientist (***eriss***) conducts a limited amount of commercial work to provide scientific advice to commercial clients in northern Australia, including mining companies and government agencies. Over the 2009–10 wet season, a contract project was undertaken by the ***eriss*** Ecotoxicology Group for Crocodile Gold Australia Operations Pty Ltd (CGAO).

The objectives of the program were to:

1. derive discharge dilutions (trigger values; TVs) for mine waters from Pine Creek Mine (PCM), Tom's Gully Mine (TGM) and the Cosmo-Howley Project Area (CHPA), which can be used by CGAO to negotiate agreed discharge dilutions with the Northern Territory Government; and
2. assess the relative toxicities of ambient waters at strategic sites along Howley Creek, spanning a longitudinal gradient that receives discharges from both CHPA and the Brocks Creek Project Area (BCPA) located upstream of CHPA.

Five tropical freshwater species were used to assess the toxicities of the mine waters: a green alga (*Chlorella* sp), duckweed (*Lemna aequinoctialis*), cladoceran (*Moinodaphnia macleayi*), green hydra (*Hydra viridissima*) and fish (northern-trout gudgeon, *Mogurnda mogurnda*). The specific mine waters assessed for toxicity were: PCM – Pine Creek Process Water Dam (PCPWD) water; TGM – Evaporation Pond 2 (EP2) water; and CHPA – water from discharge site CHCK05. The toxicity of an undiluted sample of water from the Alligator Pit at BCPA was also assessed. The key outcomes from the water toxicity assessments for the three mines were the derivation of TVs and corresponding dilution ratios for each of the three mines' licensed discharges.

Ambient waters collected from five sites along Howley Creek were screened for toxicity using the above-mentioned five species. The sites were: 1. Howley Creek upstream of both mine project areas (CHCK01; upstream control); 2. BCPA tenement boundary/Alligator Pit discharge compliance point (BCSW20; influenced by discharges from BCPA and, to a lesser extent, CHPA); 3. Immediately downstream of the confluence of the main CHPA discharge point and Howley Creek (CHCK07); 4. CHPA downstream compliance point (CHCK06); and 5. ~7 km downstream of CHPA downstream compliance point (CHCK08). The results of the Howley Creek toxicity screening assessment were consistent with those from the CHPA discharge and Alligator Pit toxicity assessments, and provided additional information to CGAO on the potential effects of its operations on aquatic biota in Howley Creek.

A comparison of the Howley Creek toxicity screening results with those from the field macroinvertebrate survey undertaken in April 2010 is currently being conducted. A synthesis of the results could be used to refine and add weight to the dilution ratios derived from the laboratory ecotoxicological test work.

Surface water quality monitoring at the Rum Jungle minesite, 2008–09 wet season

DR Jones & K Turner

Introduction

The former Rum Jungle minesite is located in the wet-dry tropics of the Northern Territory of Australia and is situated approximately 100 km south of Darwin. Extensive rehabilitation works were carried out in the mid-1980s using best practice methods of the day. In recent times it has become apparent that the performance some of the rehabilitation works associated with containment of the waste rock and tailings has begun to deteriorate.

This project was commissioned under the terms of a Memorandum of Understanding (MOU) funding agreement between SSD and the Australian Government Department of Resources, Energy and Tourism (DRET) to undertake a pilot study of deployment of continuous monitoring equipment to determine seasonal loads of solutes emanating from the site (gauging station GS8150200) and from the Dysons backfilled pit and overburden dump area.

The continuous monitoring of flow and EC was complemented by fortnightly grab samples for chemical analysis collected throughout the wet season from the GS8150200 and Dysons monitoring locations. Targeted grab sampling of flow and seepage lines around the Dysons pit area was done, prior to and at the end of the wet season, to characterise the composition of source waters from this vicinity. Several soil samples were also collected to provide additional information about levels of metals and radionuclides in surface soils and cover material on and downgradient of Dysons pit.

A synoptic grab sampling survey of water quality, at a number of strategic locations in flow lines across the site, was carried out during the recessional flow period at the end of the wet season. This was done out to identify the most important contributors to contaminant loads at this time.

Full details of this project are reported in Jones and Turner (2010).

Findings

Good grab sample coverage of water quality over the wet season at both the GS 8150200 and Dysons stations was obtained by the samples collected by staff from SSD and Compass Resources. For the first time the seasonal behaviour of run off and seepage downgradient of Dysons Pit and Overburden Heap was able to be assessed. The low pH and high concentrations of metals, including U, in the seepage downgradient from Dysons pit were of especial note. In particular there is little difference in concentrations measured at the end of the dry and at the end of the wet. This behaviour suggests that this seepage represents a chronic year-round source that does not exhibit a seasonal 'flush out' profile.

Apart from the start of the wet season the dissolved concentrations of all of the metals measured, except Cu, are very similar at both the GS8150200 and Dysons monitoring locations. In the case of Cu there is a systematically higher (almost double) concentration at GS8150200 for most of the wet season. Similar behaviour is displayed by the total (ie unfiltered) metals concentration data at both locations. It may not be possible to estimate copper load from continuous EC data owing to a substantial proportion of copper being present in particulate form during the majority of the wet season.

Surface water samples were collected in drainage lines across the site as part of a parallel groundwater characterisation project. The chemical analysis results showed that the ‘primary’ seepage from both the Whites overburden heap and Dysons Pit and overburden heap is very acidic and contains high concentrations of solutes (as evidenced by EC). However, the buffering capacity of the water coming from the upper catchment lines during the wet season appears to be sufficiently high to neutralise the bulk of acidity present in the seepage, as evidenced by the close to neutral pH values downstream of the seepage inputs. The EC data indicate that inputs of solutes from the Whites overburden heap were making the most substantial contribution in April 2009 to the solute load of water from the upper catchment as it flows through the site.

Several samples of soils, surface crusts and sediment were collected from around the Dysons pit area. Of especial note is the sample of stream sediment collected from the channel about 30m downstream of the Dysons station. It contained 571 mg/kg U and 2020 mg/kg Cu in the acid extractable fraction. The highest measured chemical concentration for U in the downstream channel sample is reflected in the ^{238}U activity value for this material. The very high ^{238}U to ^{226}Ra activity ratio suggests that the U in this material may have been sourced from uranium rich (and Ra-depleted) seepage water. It is conceivable that the source of the U is the uraniferous seepage water downgradient of Dysons pit. The U in solution would likely be adsorbed on the Fe-rich soils in the catchment and then transported downstream and deposited in the slackwater location from which the sample was collected.

As result of a combination of deployment and initial equipment reliability issues, critical early wet season data were not obtained from either the Dysons or GS8150200 stations. Hence it was not possible to derive solute loads for the whole 2008/09 wet season. This is especially the case for the Dysons station where the late start (mid-February) of acquisition of continuous data was coupled with an abnormally early end to the wet season. However, it can concluded from the available water quality and stream flow data that the area upstream of the Dysons monitoring station contributes a minor proportion of the total solute lead reporting to GS8150200.

A correlation plot between EC and sulfate concentration established using grab samples can potentially be used to infer sulfate load throughout the season (by combining sulfate concentrations derived by using the EC-SO4 correlation from grab sampling with measured flow data). This will enable the period when most SO4 (conservative solute and primary metal sulfide oxidation product) is being exported from the site. However, caution should be exercised in applying this approach. Owing to changing ionic composition with pH (especially for pH below 4.5) it will likely be necessary to use separate correlation plots for early, middle and late wet season.

Recommendations for future work

1 Contracts for future monitoring deployments should be in place well in advance of wet season to ensure that system is able to be installed and debugged prior to the start of the first rains. This is especially critical to capture data pertaining to the first flush of very poor quality water.

2 The low pH and high concentrations of metals, including U, in the seepage downgradient from Dysons pit are of especial note. In particular there is little difference in concentrations

measured at the end of the dry and at the end of the wet. This behaviour suggests that this may be a chronic year-round source that does not exhibit a seasonal 'flush out' profile. Additional work needs to be done to identify the source of this seepage.

3 Synoptic scans of water quality along the site flowlines should be made several times during the wet season to identify primary locations of solute inputs.

4 Event controlled autosampling should be used to provide a primary estimate of metal loads and thence to assess the possibility of being able to use the continuous EC data to infer metal loads (especially copper) during subsequent wet seasons.

At least two complete wet seasons of continuous monitoring data will be needed to characterise the seasonal elution profile for solutes from the site.

References

Jones DR & Turner K 2010. Surface water quality monitoring at the Rum Jungle minesite, 2008–09 wet season. Internal Report 578, June, Supervising Scientist, Darwin.

Characterisation of groundwater at the Rum Jungle minesite

B Ryan, F Bradley & A Bollhöfer

Introduction

This project was undertaken as a consultancy for the Commonwealth Department of Resources, Energy and Tourism to provide a better understanding of the groundwater regime at the former Rum Jungle minesite 100 km south of Darwin in Australia's Northern Territory. Two field sampling campaigns were conducted in October 2008 (end of 2008 dry season) and April 2009 (end of 2008–09 wet season) by staff from the Supervising Scientist Division (SSD) and the Northern Territory Department of Regional Development, Primary Industry, Fisheries and Resources (DRDPIFR). Water quality parameters, dissolved metal concentrations and radionuclide levels were analysed for 24 groundwater bores and eleven surface water sites. The aims were:

- provide a contemporary snapshot of groundwater quality across the site
- delineate locations of possible contaminated groundwater plumes
- provide insight into water flow and solute transport between shallow aquifers and surface water to investigate whether groundwater adjacent to waste rock dumps contributes to solute loads from the site.

Findings

There are elevated levels of contaminants, at some locations, in both the shallow and deeper aquifers at Rum Jungle. These contaminants are being transferred from the remnant mining structures which include Whites, Intermediate and Dyson overburden heaps and Dysons Pit. The relative contribution of how much contaminant is flowing from each of these sources into the east Finniss River needs further investigation.

It would be advantageous to establish if there is any contaminant movement out of Whites and Intermediate pits as well as the tailings dam. Although the water in Whites and Intermediate pits is relatively uncontaminated it would still be prudent to sample groundwater moving from these structures as higher levels of metals are present in water at the bottom of these pits. There may also be other as yet unidentified sources (for example, dispersed amounts of fill material) contributing to the contaminant load.

Inferred directions of flow pathways for shallow groundwater, based on head levels, are based on few points and hence interpretations based on this representation of the data should be treated with some caution.

An important finding from Principle Component Analysis is that there is little seasonal influence on the quality of water in the deeper aquifers and that there is connection between the shallow aquifer and surface water.

The extent of connection between the shallow and deeper aquifer systems is still unknown. However, in this context it is noted that early studies state the deeper aquifer is confined in nature. A better understanding of the hydraulic properties of the aquifers and the effect of the complex geological structures on facilitating connections between the aquifers is needed. Extended monitoring of some of the deeper bores with electrical conductivity and water level probes may provide some insight into the dynamics of contaminant plume movement through the aquifers and additional information on hydraulic properties. . It is recommended that this monitoring be started as soon as possible so that time series level data can be obtained over at least two seasonal cycles.

A number of the bores sampled have shown some improvement in water quality over the last three decades. However, although there may have been some improvement the levels of contaminants in many of these bores are still high compared to relevant water quality guidelines. There are also some bores that have shown deterioration in water quality over the same period. The question arises as to what has caused this degradation and whether this provides evidence for an advancing contaminant plume.

Full details of the project and its findings are reported Ryan et al (2010).

Recommendations for future work

The key knowledge need for understanding changes in EC, pH and metals concentrations through time and space is an understanding of the variables that can impact these, including the strength of the source terms, location of aquifers, the lithology through which the ground water flows, and its acid buffering capacity. In order to adequately address these issues much greater definition of the physical conformations of the geological structures is needed than available from historical drill logs.

Measuring radon in surface water along the East Finniss would identify if significant amounts of deeper groundwater are expressing at the surface and contributing to the load of metals leaving the site.

It is recommended that a Stage 2 broadly-based hydrogeological characterisation program be commissioned to further analyse the material that has been presented in this report to identify locations for the installation of additional groundwater monitoring bores and to obtain additional characterisation information about aquifer properties. The additional information produced by this assessment would provide the necessary detail to develop a conceptual, and ultimately numeric, hydrogeological model for the site.

References

Ryan B, Bradley F & Bollhöfer A 2010. Final report into the characterisation of groundwater at the Rum Jungle minesite: Consultancy report for Department of Resources, Energy and Tourism. Internal Report 571, February, Supervising Scientist, Darwin. Unpublished paper.

Flood inundation mapping: Daly and Mitchell river catchments

R Bartolo, D Ward[1] & DR Jones

Introduction

The Tropical Rivers and Coastal Knowledge (TRaCK) research hub headquartered at Charles Darwin University in Darwin is one of the major components of the CERF program that was managed by DEWHA. ***eriss*** was a collaborator in the Theme 4 (Material Budgets) Project 4.1: Catchment water budgets and water resource assessment. The specific engagement was with a task focused on flood inundation mapping for the Daly and Mitchell River catchments using a combination of radar and optical satellite imagery analysis. Defining the extent of wet season inundation in floodplain and riverine environments is an important component of the annual catchment surface and groundwater budgeting process.

The aims of this sub-project were to:

1 Map the maximum inundation limit (extent) for both the Daly River and Mitchell River catchments;

2 Map the minimum inundation limit (extent) for both the Daly River and Mitchell River catchments; and

3 Provide information on the persistence of flooding in these catchments.

Determining the extent of flooding in tropical catchments using remote sensing is dependent on a number of factors: local conditions at the time of image acquisition (for example, cloud cover and flooding under vegetation); sensor selection (Optical or Synthetic Aperture Radar [SAR]); and definition of flood extent (interannual and intrannual analysis). There are a number of challenges to using remotely sensed data for such a task as mapping tropical floodplain inundation extent. In the tropics, atmospheric attenuation or interference resulting from cloud cover during the 'wet' season, and smoke from fires during the 'dry' season (Schultz & Engman 2000) are limiting factors for the use of optical sensors.

Identification of 'wettest' and 'driest' wet seasons

The wet seasons that received the highest and lowest rainfall were identified using Foley's precipitation deficit index (Foley 1957), and this analysis provided the basis for locating the years corresponding to the likely maximum and minimum interannual inundation extents of the selected catchments. Foley's precipitation deficit index is the standardized monthly mean annual precipitation over a specified lag period relative to the long-term mean annual precipitation. Fensham and Holman (1999) found that 3 years is a significant lag period for precipitation deficit to influence vegetation dynamics in Australia's tropical savannas. Hence, we calculated Foley's precipitation deficit index for a 3 year lag period. The year that matched the satellite record and data availability for both optical and L-band SAR data for mapping maximum

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inundation extent in both catchments was identified to be 2009. The year that matched the satellite record for optical data for mapping minimum inundation extent was identified to be 2005 and 2004 for the Daly River and Mitchell River catchments respectively (there were no L-band SAR data available for years that fell into the lowest rainfall wet season range).

Satellite data and image processing

Both optical (Landsat 5 TM) and Synthetic Aperture Radar (SAR – ALOS PALSAR ScanSAR) data were combined to map maximum inundation extent during the wettest wet season (2009) in the Daly River catchment floodplain complex. Landsat 5 TM data were utilised for mapping the minimum inundation extent for the driest wet season (2005) in the Daly River catchment (SAR data were not available for that particular year and time). A time series consisting of 5 images were used to assess persistence in the Daly River catchment (which is considered more important than duration of flooding for the work undertaken under Theme 5) of flooding in 2009. Floods in the Mitchell river catchment are ‘flashy’ and inundation lasts in the order of 8 to 10 weeks. In order to capture the floods on the Mitchell it was necessary to use a satellite with a return interval greater than Landsat (16 days) because of issues of occlusion due to cloud cover. The USGS Terra MODIS satellite has a daily return interval but has relatively low resolution (250 m pixels). Consequently, MODIS optical imagery was used for mapping the maximum and minimum inundation limit for the Mitchell River catchment.

A Geographic Object Based Image Analysis (GEOBIA) approach (Benz et al 2004) was used to classify the Landsat 5 TM and PALSAR ScanSAR data using Definiens Developer 8.0. Multi-resolution segmentation was conducted.

Two band ratios, based on published literature on techniques to extract water coverage from optical satellite imagery, were used to highlight open water (flooding) in the Landsat 5 TM data. The first of these band ratios, the Normalised Difference Water Index (NDWI), was developed to delineate open water in satellite imagery (McFeeters 1996). The second ratio, the Modified Normalised Difference Index (MNDWI), is based upon the principle of the NDWI but further suppresses the signal from built up areas (Xu 2006) and cleared regions in the imagery by substituting the NIR with the MIR band. For MODIS, the two highest resolution bands are the Red and NIR bands, allowing calculation of NDVI. Using measurements obtained from a combination of depth loggers and temperatures sensors located on the Mitchell floodplain for the 2009 wet season, a comparison was made between density slicing of NIR and NDVI for delineating water. NDVI performed the best and was subsequently used to delineate maximum and minimum inundation extents for the Mitchell catchment.

Results

The full range of results is not reported here. Examples of some results are provided to indicate how the data can be used for environmental management.

Daly River floodplain

Maximum inundation extent for the Daly River Floodplain Complex was mapped using both Landsat 5 TM and PALSAR ScanSAR imagery. Three flood classes were produced: open water; flooded Melaleuca; and flooded grasses and sedges. Maximum inundation extent is shown by the mapping product displayed in Figure 1. A summary of the areal extent of the flooded classes over three months during the 2009 wet season is shown in Figure 2. It should be noted that the open water class includes the near shore environment as the data used to subset imagery was the GEODATA TOPO 250K subject to inundation with a 2km buffer. Therefore, the open water class using this method is not entirely confined to the floodplain proper.

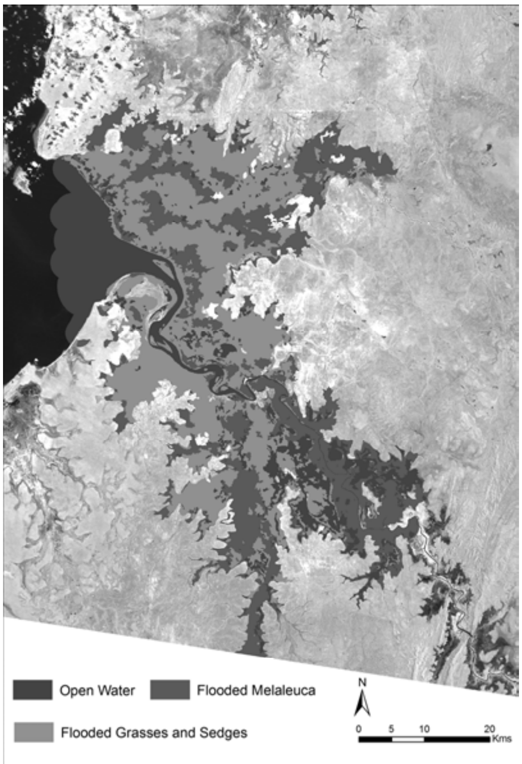

Figure 1 Maximum inundation extent during March 2009 for the Daly River Floodplain Complex

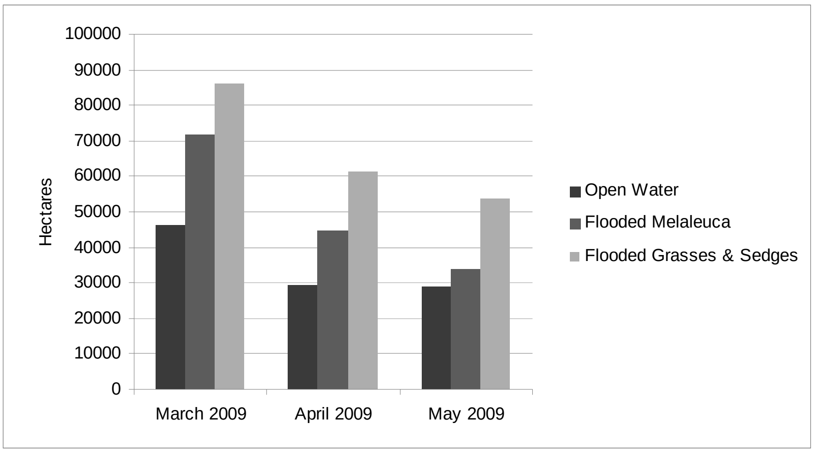

Figure 2 Summary of the areal extent of flooded classes during March–April 2009 for the Daly River floodplain complex

Mitchell River catchment

The majority of vegetation on the Mitchell river floodplain has less than 30% canopy density (Foliage Projective Cover), allowing water to be relatively easily detected using MODIS NDVI. Areas with canopy density greater than 30% were limited largely to the main river channel and subsidiary channel levees, and flood inundation was not mapped in these areas. A summary of the areal extent of flood inundation over 2 months during the 2009 wet season is shown in Figure 3.

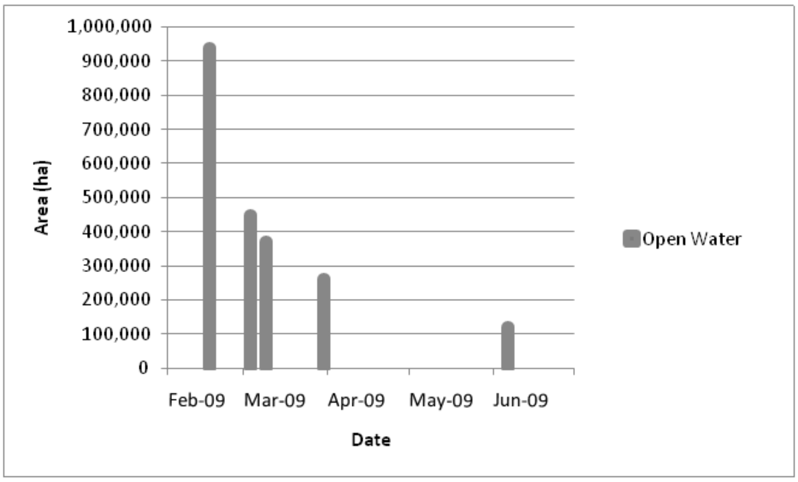

Figure 3 Summary of the areal extent of flood inundation for the 2009 wet season (with additional Landsat 5 TM water extent for June 2009) for the Mitchell River floodplain

Findings

The methodology for reliable and robust flood inundation mapping in Australia, and in particular northern Australia, is still in its developmental phase. The type of satellite sensors used are critical for mapping flood classes in northern Australia, due to ubiquitous cloud cover during the wet season, coverage of floodplains by grasses and aquatic plants etc following rain, followed by fire and associated smoke in the dry season. The effect of cloud cover and smoke in optical imagery is not the only limiting factor. Fire scars from early dry season burning (as early as May) can result in class confusion with flooded classes, in particular flooded vegetation (even with the inclusion of SAR data).

The findings from this project are that combining L-band SAR data with optical data substantially improves the ability to map flooded classes during the wet season. Flooded *Melaleuca* swamp areas are particularly well distinguished in the SAR data. The MNDWI data analysis method was found to be particularly useful in mapping open water. Combining the SRTM DEM data into the classification process may further improve classification results and this is an area for further investigation for mapping flooded classes. For optimal data classification results it is recommended that the optical and SAR data are acquired concurrently. Unfortunately with limitations in availability of satellite L-band SAR, and the available frequency of image acquisition by the SAR-capable platforms, this is not often possible.

Depending on the environmental management question to be answered, combining the three flooded classes into one flooded class for the Daly may be useful. From the isotope tracing food web work for TRaCK Theme 5.3 it was found that 30–40% of the biomass of barramundi caught in the Mitchell is coming from food sources on the floodplain during the wet season. There is evidence that this is true for other fish species as well. This indicates the importance of inundated floodplains for the maintenance of aquatic food webs. One approach to using the flood inundation mapping for management is to identify those areas of the flood plain that have the longest flood duration and that are most connected to the drainage network. In the absence of an exact inundation time series that allow estimation of inundation duration, flood inundation frequency could be used as a surrogate. Flood inundation frequency is calculated by combing all flood event captures (over any number of years) and adding them such that the frequency of flood occurrence can be determined for any point in the landscape.

Recommendations for future work

The L-band SAR imagery used in this study was the Map 1.5-G product which is provided with a level of pre-processing. The use of the Level 1.0 (raw data) would enable the Cloude Decomposition to be implemented. The Cloude Decomposition method results in data showing single, double and multiple bounce interactions with targets. The resulting data can indicate the presence of water and flooding under vegetation. However, the Level 1.0 raw data requires specialised processing software such as RAT (Radar Tools- which is open source software), available from: http://radartools.berlios.de/. There are two polarimetric images that were acquired for the Daly River floodplain which can be used to test the Cloude Decomposition method. Further investigation will also be conducted on the Fine Beam Double imagery as the cross-polarised channel (H-V) may be useful in mapping some of the flooded classes.

While cloud cover is a limitation for the optical remote sensing of floods in northern Australia it can applied to map the dynamics of water following flood events. A characteristic feature of the wet-dry tropics of northern Australia is the relative absence of cloud following the wet season. Landsat 5 TM is free and readily available and can be effectively applied to map the seasonal dynamics of water bodies following floods. One of the key advantages of optical remote sensing over microwave remote sensing is the capacity to map the constituents of the water bodies such as turbidity which has a significant influence on aquatic primary productivity.

Validation of flood mapping

The Daly River mapping is currently unvalidated, and it is recommended that ground-truthing and validation be conducted at the time of image acquisition to provide a level of accuracy (or uncertainty) on the classification results. Accuracy assessment of the MODIS food inundation mapping for the Mitchell was undertaken using data obtained from a combination of depth loggers and temperatures sensors deployed on the Mitchell floodplain over the 2009 wet season. A total of 8 depth loggers (pressure transducers) and 33 temperature sensors were deployed. Accuracy of the MODIS flood mapping ranged from 78% to 86% based on the number of correctly classified sites divided by the total number of sites.

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Further information

SSD publications on the web

http://www.environment.gov.au/ssd/publications/index.html

SSD annual reports

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http://www.environment.gov.au/ssd/publications/ssd-bibliography.html

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Appendix 2 ARRTC membership and functions

The Alligator Rivers Region Technical Committee (ARRTC) was established in 1993 following amendments to the Commonwealth *Environment Protection (Alligator Rivers Region) Act 1978*. The membership structure and functions of ARRTC were revised in 2001 in response to a recommendation by an Independent Science Panel established by the World Heritage Committee calling for the establishment of an independent scientific advisory panel to review research activities in the Alligator Rivers Region and the scientific basis for assessing mining operations.

ARRTC membership

ARRTC comprises:

- seven independent scientific members (including the Chair) nominated by the Federation of Australian Scientists and Technological Societies (FASTS) with expertise in the following disciplines:
  - Hydrology and hydrogeology
  - Radiation protection and health physics
  - Plant ecology of minesite revegetation
  - Freshwater ecology
  - Ecotoxicology
  - Geomorphology
  - Chemistry and ecological risk assessment;
- seven members representing key stakeholder interests.

ARRTC functions

The primary functions of ARRTC are:

a to consider programs for research into, and programs for the collection and assessment of information relating to, the effects on the environment in the Alligator Rivers Region of uranium mining operations in the Region;

b to keep under review programs and the carrying out of programs, referred to in paragraph (a);

c to make recommendations to the Minister for Sustainability, Environment, Water, Population and Communities on:

  i the nature and extent of research necessary to protect and restore the environment in the Alligator Rivers Region, and

  ii the most appropriate organisations to undertake the research referred to in subparagraph (i); and

d to refer to the Alligator Rivers Region Advisory Committee matters relating to programs, and the carrying out of programs, referred to in paragraph (a).

Appendix 3
Alligator Rivers Region Technical Committee Key Knowledge Needs 2008–2010: Uranium mining in the Alligator Rivers Region

As a result of the extension in mine life and the conduct of a further three years of research since the original key knowledge needs (KKNs) and timeline priorities were established, it was judged by the Alligator Rivers Region Technical Committee (ARRTC) that a revision of the KKNs should be conducted in 2007–08. This was done and a revised list of KKNs approved by the committee. For comparison, both the original (2004–2006) KKNs and the new (2008–2010) KKNs were listed in Appendix 3 of the 2007–2008 ***eriss*** research summary (SSR200). The appendix in this volume contains the 2008–2010 KKNs.

Overall objective

To undertake relevant research that will generate knowledge leading to improved management and protection of the ARR and monitoring that will be sufficiently sensitive to assess whether or not the environment is protected to the high standard demanded by the Australian Government and community.

Background

In assessing the Key Knowledge Needs for research and monitoring in the Alligator Rivers Region, ARRTC has taken into account current mining plans in the region and the standards for environmental protection and rehabilitation determined by the Australian Government. The assumptions made for uranium mining operations in the region are:

- mining of uranium at Ranger is expected to cease in about 2012. This will be followed by milling until about 2020 and final rehabilitation expected to be completed by about 2026;
- Nabarlek is decommissioned but has not reached a status where the NT Government will agree to issue a Revegetation Certificate to the mine operator. Assessment of the success of rehabilitation at Nabarlek is ongoing and may provide valuable data for consideration in the design and implementation of rehabilitation at Ranger;
- Jabiluka will remain in a care and maintenance condition for some years. ERA, the project owner, has stated that further mining will not occur without the agreement of the traditional owners; and
- grant of an exploration title at Koongarra is required under the terms of the *Aboriginal Land Rights (Northern Territory) Act 1976* before the mining company can apply for a mining title. As such, any future activity at Koongarra is subject to the agreement of the traditional owners and the Northern Land Council.

This scenario is considered to be a reasonable basis on which to base plans for research and monitoring, but such plans may need to be amended if mining plans change in the future. ARRTC will ensure the research and monitoring strategy is flexible enough to accommodate any new knowledge needs.

The Australian Government has specified Primary and Secondary environmental objectives for mining at Ranger in the Ranger Environmental Requirements. Similar standards would be expected for any future mining development at Jabiluka or Koongarra.

Specifically, under the Ranger Environmental Requirements (ERs):

> The company must ensure that operations at Ranger are undertaken in such a way as to be consistent with the following primary environmental objectives:
>
> (a) maintain the attributes for which Kakadu National Park was inscribed on the World Heritage list;
>
> (b) maintain the ecosystem health of the wetlands listed under the Ramsar Convention on Wetlands (ie the wetlands within Stages I and II of Kakadu National Park);
>
> (c) protect the health of Aboriginals and other members of the regional community; and
>
> (d) maintain the natural biological diversity of aquatic and terrestrial ecosystems of the Alligator Rivers Region, including ecological processes.

With respect to rehabilitation at Ranger, the ERs state that:

> The company must rehabilitate the Ranger Project Area to establish an environment similar to the adjacent areas of Kakadu National Park such that, in the opinion of the Minister with the advice of the Supervising Scientist, the rehabilitated area could be incorporated into the Kakadu National Park.

The ERs go on to specify the major objectives of rehabilitation at Ranger as follows:

> (a) revegetation of the disturbed sites of the Ranger Project Area using local native plant species similar in density and abundance to those existing in adjacent areas of Kakadu National Park, to form an ecosystem the long term viability of which would not require a maintenance regime significantly different from that appropriate to adjacent areas of the park;
>
> (b) stable radiological conditions on areas impacted by mining so that, the health risk to members of the public, including traditional owners, is as low as reasonably achievable; members of the public do not receive a radiation dose which exceeds applicable limits recommended by the most recently published and relevant Australian standards, codes of practice, and guidelines; and there is a minimum of restrictions on the use of the area;
>
> (c) erosion characteristics which, as far as can reasonably be achieved, do not vary significantly from those of comparable landforms in surrounding undisturbed areas.

A secondary environmental objective applies to water quality and is linked to the primary ERs. This ER states:

> The company must not allow either surface or ground waters arising or discharging from the Ranger Project Area during its operation, or during or following rehabilitation, to compromise the achievement of the primary environmental objectives.

While there are many possible different structures that could be used to specify the Key Knowledge Needs, ARRTC has chosen to list the knowledge needs under the following headings:

- Ranger – current operations;
- Ranger – rehabilitation;
- Jabiluka;
- Nabarlek; and
- General Alligator Rivers Region.

'Key Knowledge Needs 2008–2010: Uranium mining in the Alligator Rivers Region' is based on and supersedes a predecessor document, 'Key Knowledge Needs 2004–2006: Uranium mining in the Alligator Rivers Region'. KKNs 2004–2006 remained the operative set during their review and the development of KKNs 2008–2010.

While some KKNs remain essentially unchanged, others contain revised elements or are new in their entirety. Care should be exercised if using KKN numbers alone as a reference because some continuing KKNs have changed numbers in the revised document.

1 Ranger – Current operations

1.1 Reassess existing threats

1.1.1 Surface water transport of radionuclides

Using existing data, assess the present and future risks of increased radiation doses to the Aboriginal population eating bush tucker potentially contaminated by the mining operations bearing in mind that the current Traditional Owners derive a significant proportion of their food from bush tucker.

1.1.2 Atmospheric transport of radionuclides

Using existing data and atmospheric transport models, review and summarise, within a risk framework, dose rates for members of the general public arising from operations at the Ranger mine.

1.2 Ongoing operational issues

1.2.1 Ecological risks via the surface water pathway

Off-site contamination during mine operation (and subsequent to decommissioning – refer KKN 2.6.1) should be placed in a risk-based context. A conceptual model of the introduction, movement and distribution of contaminants, and the resultant biotic exposure (human and non-human) has been developed, and the ecological risks (ie probability of occurrence x severity of consequence) of some of the contaminant/pathway sub-models have been estimated. This process should be completed for all the contaminant/pathway sub-models, noting, however, that the level of effort for each needs to be proportionate to the level of concern of the issue~~.~~ It is critical that robust risk assessment methodologies are used, and that they explicitly incorporate uncertainty in both the assessment and subsequent decision making processes. Where ecological risk is significant, additional information may be required (eg. mass-balance and concentration dynamics, consideration of possible interactive effects, field data). Further, knowledge gaps preventing reasonable estimation of potential risks (ie with unacceptable uncertainty) must be filled.

The Magela floodplain risk assessment framework developed to estimate and compare mining and non-mining impacts should be revisited periodically, and updated to the current risk profile. It should be revised in the event that either **(i)** the annual monitoring program or other sources indicate that the inputs from mining have significantly increased relative to the situation in 2005, or **(ii)** an additional significant contaminant transport pathway from the minesite is identified, or **(iii)** there is a change in external stressors that could result in a significant increase in likelihood of impacts from the site.

1.2.2 Land irrigation

Investigations are required into the storage and transport of contaminants in the land irrigation areas particularly subsequent to decommissioning. Contaminants of interest/concern in addition to radionuclides are magnesium, sulfate and manganese. Results from these investigations should be sufficient to quantify the role of irrigation areas as part of satisfying KKN 1.2.1, and form the basis for risk management into the future.

1.2.3 Wetland filters

The key research issue associated with wetland filters in relation to ongoing operations is to determine whether their capacity to remove contaminants from the water column will continue to meet the needs of the water management system in order to ensure protection of the downstream environment. Aspects of contaminant removal capacity include (i) instantaneous rates of removal, (ii) temporal performance – including time to saturation, and (iii) behaviour under ‘breakdown’ conditions–including future stability after closure. Related to this is a reconciliation of the solute mass balance particularly for the Corridor Creek System (see KKN 1.2.5).

1.2.4 Ecotoxicology

Past laboratory studies provide a significant bank of knowledge regarding the toxicity of two of the major contaminants, uranium and magnesium, associated with uranium mining in the ARR. Further studies are scheduled to assess (i) the toxicity of manganese and, potentially, ammonia (in the event that permeate produced by process water treatment will contain potentially toxic ammonia concentrations), and (ii) the relationship between dissolved organic matter and uranium toxicity. This knowledge should continue to be synthesised and interpreted, within the existing risk assessment framework (refer KKN 1.2.1), as it comes to hand.

An additional issue that needs to be addressed is the direct and indirect effects on aquatic biota of sediment arising from the minesite. In the first instance, a conceptual model needs to be developed (building on the relevant components of the conceptual model developed under KKN 1.2.1) that describes the movement of sediment within the creek system, including the associated metal-sediment interactions and biological implications. Studies likely to arise from the outcomes of the conceptual model include:

- the effects of suspended sediment on aquatic biota;
- the relationship between suspended sediment and key metals, and how this affects their bioavailability and toxicity; and
- the effects of sediment-bound metals to benthic biota, including, initially, a review of existing information on uranium concentrations in sediments of waterbodies both on- and off the Ranger site, and uranium sediment toxicity to freshwater biota.

Whilst of relevance at present, the above issues will be of additional importance as Ranger progresses towards closure and rehabilitation (refer KKN 2.6.1). Finally, the need for studies to assess the toxicity of various mine waters (treated and untreated) in response to specific supervisory/regulatory or operational requirements is likely to continue.

1.2.5 Mass balances and annual load limits

With the expansion of land application areas and the increase in stockpile sheeting that has occurred in concert with the expansion of the footprints of the waste rock dumps and low grade ore stockpiles, it is becoming increasingly important to develop a solute mass balance for the site – such that the behaviour of major solute source terms and the spatial and temporal contribution of these sources to water quality in Magela Creek can be clearly understood. Validated grab sample and continuous data records are needed to construct a high reliability solute mass balance model.

Related to mass balance is the issue of specifying allowable annual load limits from the site – as part of the site's regulatory requirements. The technical basis for these load limits needs to be reviewed since they were originally developed decades ago. There has since been significantly increased knowledge of the environmental geochemistry of the site, a quantum increase in knowledge about ecotoxicological sensitivity of the aquatic systems and updated data on the diet profile of traditional owners.

1.3 Monitoring

1.3.1 Surface water, groundwater, chemical, biological, sediment, radiological monitoring

Routine and project-based chemical, biological, radiological and sediment monitoring should continue, together with associated research of an investigative nature or necessary to refine existing, or develop new (promising) techniques and models. A review of current water quality objectives for Ranger should be conducted to determine if they are adequate for future water management options for the whole-of-site, including the closure and rehabilitation phase (KKN 2.2.1 and KKN 2.2.2).

ARRTC supports the design and implementation of a risk-based radiological monitoring program based on a robust statistical analysis of the data collected over the life of Ranger necessary to provide assurance for Aboriginal people who source food items from the Magela Creek system downstream of Ranger.

2 Ranger – Rehabilitation

2.1 Reference state and baseline data

2.1.1 Defining the reference state and baseline data

There is a requirement to define the baseline data/reference state that existed at the Ranger site prior to development. This will inform the process of the development of closure criteria which is compatible with the Environmental Requirements. The knowledge need is to develop and perform analysis to generate agreed reference data that cover the range of pre-mining and operational periods.

2.2 Landform

2.2.1 Landform design

An initial design is required for the proposed final landform. This would be based upon the optimum mine plan from the operational point of view and it would take into account the broad closure criteria, engineering considerations and the specific criteria developed for guidance in the design of the landform. This initial landform would need to be optimised using the information obtained in detailed water quality, geomorphic, hydrological and radiological programs listed below.

Current and trial landforms at Ranger and at other sites such as Nabarlek should be used to test the various models and predictions for water quality, geomorphic behaviour and radiological characteristics at Ranger. The detailed design for the final landform at Ranger should be determined taking into account the results of the above research programs on surface and ground water, geomorphic modelling and radiological characteristics.

2.2.2 Development and agreement of closure criteria from the landform perspective

Closure criteria from the landform perspective need to be established at both the broad scale and the specific. At the broad scale, agreement is needed, particularly with the Traditional Owners and within the context of the objectives for rehabilitation incorporated within the ERs, on the general strategy to be adopted in constructing the final landform. These considerations would include issues such as maximum height of the landform, the maximum slope gradient (from the aesthetic perspective), and the presence or absence of lakes or open water. At the specific scale, some criteria could usefully be developed as guidance for the initial landform design such as slope length and angle (from the erosion perspective), the minimum cover required over low grade ore, and the minimum distance of low grade ore from batter slopes. Specific criteria are needed that will be used to assess the success of landform construction. These would include, for example, maximum radon exhalation and gamma dose rates, maximum sediment delivery rates, maximum constituent concentration rates in runoff and maximum settling rates over tailings repositories.

2.2.3 Water quality in seepage and runoff from the final landform

Existing water quality monitoring and research data on surface runoff and subsurface flow need to be analysed to develop models for the quality of water, and its time dependence, that would enter major drainage lines from the initial landform design. Options for adjusting the design to minimise solute concentrations and loads leaving the landform need to be assessed.

There is a need to develop and analyse conceptual models of mine related turbidity and salinity impacts following closure. These models could be analysed in a variety of ways~~,~~ as a precursor to the development of a quantitative model of potential turbidity and salinity impacts offsite cause by surface and subsurface water flow off the rehabilitated minesite. This analysis should explicitly acknowledge knowledge uncertainty (eg plausible alternative conceptual models) and variability (eg potential for Mg/Ca ratio variations in water flowing off the site) and explore the potential ramifications for the off-site impacts. (see also KKN 2.6.1)

2.2.4 Geomorphic behaviour and evolution of the landscape

The existing data set used in determination of the key parameters for geomorphological modelling of the proposed final landform should be reviewed after consideration of the near surface characteristics of the initial proposed landform. Further measurements of erosion characteristics should be carried out if considered necessary. The current site-specific landform evolution models should be applied to the initial proposed landform to develop predictions for long term erosion rates, incision and gullying rates, and sediment delivery rates to the surrounding catchments. Options for adjusting the design to minimise erosion of the landform need to be assessed. In addition, an assessment is needed of the geomorphic stability of the Ranger minesite with respect to the erosional effects of extreme events.

2.2.5 Radiological characteristics of the final landform

The characteristics of the final landform from the radiological exposure perspective need to be determined and methods need to be developed to minimise radiation exposure to ensure that restrictions on access to the land are minimised. Radon exhalation rates, gamma dose rates and radionuclide concentrations in dust need to be determined and models developed for both near-field and far-field exposure.

The use of potential analogue sites for establishing pre-mining radiological conditions at Ranger should be further investigated to provide information on parameters such as pre-mining gamma dose rates, radon exhalation rates, and levels of radioactivity in dust. This information is needed to enable estimates to be made of the likely change in radiation exposure when accessing the rehabilitated site compared to pre-mining conditions.

2.3 Groundwater dispersion

2.3.1 Containment of tailings and other mine wastes

The primary method for protection of the environment from dispersion of contaminants from tailings and other wastes will be containment. For this purpose, investigations are required on the hydrogeological integrity of the pits, the long-term geotechnical properties of tailings and waste rock fill in mine voids, tailings deposition and transfer (including TD to Pit #3) methods, geochemical and geotechnical assessment of potential barrier materials, and strategies and technologies to access and 'seal' the surface of the tailings mass, drain and dispose of tailings porewater, backfill and cap the remaining pit void.

2.3.2 Geochemical characterisation of source terms

Investigations are needed to characterise the source term for transport of contaminants from the tailings mass in groundwater. These will include determination of the permeability of the tailings and its variation through the tailings mass, strategies and technologies to enhance settled density and accelerate consolidation of tailings, and pore water concentrations of key constituents.

There is a specific need to address the existence of groundwater mounds under the tailings dam and waste rock stockpiles. Models are needed to predict the behaviour of groundwater and solute transport in the vicinity of these mounds and options developed for their remediation to ensure that on-site revegetation can be achieved and that off-site solute transport from the mounds will meet environmental protection objectives. Assessment is also needed of the effectiveness (cost and environmental significance) of paste and cementation technologies for increasing tailings density and reducing the solubility of chemical constituents in tailings.

2.3.3 Aquifer characterisation and whole-of-site model

The aquifers surrounding the tailings repositories (Pits 1 & 3) need to be characterised to enable modelling of the dispersion of contaminants from the repositories. This will involve geophysics surveys, geotechnical drilling and groundwater monitoring and investigations on the interactions between the deep and shallow aquifers.

2.3.4 Hydrological/hydrogeochemical modelling

Predictive hydrological/hydrogeological models need to be developed, tested and applied to assess the dispersion of contaminants from the tailings repositories over a period of 10 000 years. These models will be used to assess whether all relevant and appropriate factors have been considered in designing and constructing an in-pit tailings containment system that will prevent environmental detriment in the long term.

2.4 Water treatment

2.4.1 Active treatment technologies for specific mine waters

Substantial volumes of process water retained at Ranger in the tailings dam and Pit 1 must be disposed of by a combination of water treatment and evaporation during the mining and milling phases of the operation and during the rehabilitation phase. Research priorities include treatment technologies and enhanced evaporation technologies that can be implemented for very high salinity process water. A priority should be evaluation of the potential impact of treatment sludge and brine streams on long term tailings chemistry in the context of closure planning and potential post closure impacts on water quality.

2.4.2 Passive treatment of waters from the rehabilitated landform

Sentinel wetlands may form part of the final landform at Ranger. Research on wetland filters during the operational phase of mining will provide information relevant to this issue. Research is needed to establish the effect of wet-dry seasonal cycling on contaminant retention and release, since this aspect will influence design criteria and whether such wetlands should be maintained as ephemeral or perennial waterbodies There is also the need to assess the long-term behaviour of the physical and biotic components of the wetlands, their ecological health, and the extent of contaminant accumulation (both metals and radionuclides) in the context of potential human exposure routes.

2.5 Ecosystem establishment

2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective

Closure criteria need to be established for a range of ecosystem components including surface water quality, flora and fauna. The environmental requirements provide some guidance but characterisation of the analogue ecosystems will be an important step in the process. Consultation on closure criteria with the traditional owners has commenced and it is important that this process continues as more definitive criteria are developed.

2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites

Identification and characterisation of analogue ecosystems (target habitats) can assist in defining the rehabilitation objective and developing robust, measurable and ecologically-based closure criteria. The concept of using analogue ecosystems for this purpose has been accepted by ARRTC and the traditional owners. Substantial work has been undertaken on the Georgetown terrestrial analogue ecosystem while there is also a large body of information available on aquatic analogues, including streams and billabongs. Future work on the terrestrial analogue needs to address water and nutrient dynamics, while work on the aquatic analogue will include the development of strategies for restoration of degraded or removed natural waterbodies, Coonjimba and Djalkmara, on site.

2.5.3 Establishment and sustainability of ecosystems on mine landform

Research on how the landform, terrestrial and aquatic vegetation, fauna, fauna habitat, and surface hydrology pathways will be reconstructed to address the Environmental Requirements for rehabilitation of the disturbed areas at Ranger is essential. Trial rehabilitation research sites should be established that demonstrate an ability by the mine operator to be able to reconstruct terrestrial and aquatic ecosystems, even if this is at a relatively small scale. Rehabilitation establishment issues that need to be addressed include species selection; seed collection, germination and storage; direct seeding techniques; propagation of species for planting; fertiliser strategies and weathering properties of waste rock. Rehabilitation management issues requiring investigation include the stabilisation of the land surface to erosion by establishment of vegetation, return of fauna; the exclusion of weeds; fire management and the re-establishment of nutrient cycles. The sustainable establishment and efficiency of constructed wetland filters, reinstated waterbodies (eg Djalkmara Billabong) and reconstructed waterways also needs to be considered (see KKN 2.3.2).

2.5.4 Radiation exposure pathways associated with ecosystem re-establishment

Radionuclide uptake by terrestrial plants and animals on the rehabilitated ecosystem may have a profound influence on the potential utilisation of the land by the traditional owners. Significant work has been completed on aquatic pathways, particularly the role of freshwater mussels, and this now forms part of the annual monitoring program. The focus is now on the terrestrial pathways and deriving concentration factors for Bushtucker such as wallabies, fruits and yams. A project investigating the contemporary diet of traditional owners has commenced and needs to be completed. Models need to be developed that allow exposure pathways to be ranked for currently proposed and future identified land uses, so that identified potentially significant impacts via these pathways can be limited through appropriate design of the rehabilitation process.

2.6 Monitoring

2.6.1 Monitoring of the rehabilitated landform

A new management and monitoring regime for the rehabilitated Ranger landform needs to be developed and implemented. It needs to address all relevant aspects of the rehabilitated landform including ground and surface water quality, radiological issues, erosion, flora, fauna, weeds, and fire. The monitoring regime should address the key issues identified by the ecological risk assessment of the rehabilitation phase (KKN 2.7.1).

2.6.2 Off-site monitoring during and following rehabilitation

Building upon the program developed and implemented for the operational phase of mining, a monitoring regime is also required to assess rehabilitation success with respect to protection of potentially impacted ecosystems and environmental values. This program should address the dispersion of contaminants by surface water, ground water and via the atmosphere. The monitoring regime should address the key issues identified by the ecological risk assessment of the rehabilitation phase (KKN 2.7.1).

2.7 Risk assessment

2.7.1 Ecological risk assessments of the rehabilitation and post rehabilitation phases

In order to place potentially adverse on-site and off-site issues at Ranger during the rehabilitation phase within a risk management context, it is critical that a robust risk assessment framework be developed with stakeholders. The greatest risk is likely to occur in the transition to the rehabilitation phase, when active operational environmental management systems are being progressively replaced by passive management systems. A conceptual model of transport/exposure pathways should be developed for rehabilitation and post rehabilitation regimes and the model should recognise the potential that some environmental stressors from the minesite could affect the park and vice versa. Implicit in this process should be consideration of the effects of extreme events and climate change.

Conceptual modelling should be followed by a screening process to identify and prioritise key risks for further qualitative and/or quantitative assessments. The conceptual model should be linked to closure criteria and post-rehabilitation monitoring programs, and be continually tested and improved. Where appropriate, risk assessments should be incorporated into decision making processes for the closure plan. Outputs and all uncertainties from this risk assessment process should be effectively communicated to stakeholders.

2.8 Stewardship

The concept of Stewardship (including ownership and caring for the land) is somewhat broader and applies to all phases of, in this case, uranium mining. In this context it is considered to be the post closure phase of management of the site, ie after relinquishment of the lease. If the rehabilitation phase is successful in meeting all objectives then this stewardship will effectively comprise an appropriate level of ongoing monitoring to confirm this. Should divergence from acceptable environmental outcomes be detected then some form of intervention is likely to be required. The nature, responsibility for, and duration of, the monitoring and any necessary intervention work remains to be determined.

3 Jabiluka

3.1 Monitoring

3.1.1 Monitoring during the care and maintenance phase

A monitoring regime for Jabiluka during the care and maintenance phase needs to be implemented and regularly reviewed. The monitoring program (addressing chemical, biological, sedimentalogical and radiological issues) should be commensurate with the environmental risks posed by the site, but should also serve as a component of any program to collect baseline data required before development such as meteorological and sediment load data.

3.2 Research

3.2.1 Research required prior to any development

A review of knowledge needs is required to assess minimum requirements in advance of any development. This review would include radiological data, the groundwater regime (permeabilities, aquifer connectivity etc), hydrometeorological data, waste rock erosion, assess site-specific ecotoxicology for uranium, additional baseline for flora and fauna surveys.

4 Nabarlek

4.1 Success of revegetation

4.1.1 Revegetation assessment

Several assessments of the revegetation at Nabarlek have been undertaken; the most recent being completed by ***eriss***. There is now general agreement that the rehabilitated areas require further work. Revised closure criteria are currently being developed through the mine-site technical committee and these should be reviewed by relevant stakeholders, including ARRTC. The required works should then be completed on site with further monitoring leading to the relinquishment of the lease.

4.1.2 Development of revegetation monitoring method

A methodology and monitoring regime for the assessment of revegetation success at Nabarlek needs to be developed and implemented. Currently, resource intensive detailed vegetation and soil characterisation assessments along transects located randomly within characteristic areas of the rehabilitated landform are being undertaken. Whilst statistically valid, these assessments cover only a very small proportion of the site. Remote sensing (satellite) data are also being collected and the efficacy of remote sensing techniques for vegetation assessment in comparison to ground survey methods should continue. The outcomes of this research will be very relevant to Ranger.

4.2 Assessment of radiological, chemical and geomorphic success of rehabilitation

4.2.1 Overall assessment of rehabilitation success at Nabarlek

The current program on erosion, surface water chemistry, groundwater chemistry and radiological issues should be continued to the extent required to carry out an overall assessment of the success of rehabilitation at Nabarlek. In particular, all significant radiological exposure pathways should be identified and a comprehensive radiation dose model developed. Additional monitoring of ground water plumes is required to allow assessment of potential future groundwater surface water interaction and possible environmental effects.

5 General Alligator Rivers Region

5.1 Landscape scale analysis of impact

5.1.1 Develop a landscape-scale ecological risk assessment framework for the Magela catchment that incorporates, and places into context, uranium mining activities and relevant regional landscape processes and threats, and that builds on previous work for the Magela floodplain

Ecological risks associated with uranium mining activities in the ARR, such as current operations (Ranger) and rehabilitation (Nabarlek, Jabiluka, future Ranger, South Alligator Valley), should be assessed within a landscape analysis framework to provide context in relation to more diffuse threats associated with large-scale ecological disturbances, such as invasive species, unmanaged fire, cyclones and climate change. Most key landscape processes occur at regional scales, however the focus will be on the Magela catchment encompassing the RPA. A conceptual model should first be developed to capture links and interactions between multiple risks and assets at multiple scales within the Magela catchment, with risks associated with Ranger mining activities made explicit. The spatially explicit Relative Risk Model will be used to prioritise multiple risks for further qualitative and/or quantitative assessments. The conceptual model and risk assessment framework should be continually tested and improved as part of Best Practice. Where appropriate, risk assessments should be incorporated into decision making processes using advanced risk assessment frameworks such as Bayesian Networks, and all uncertainties made explicit. This risk assessment process should integrate outputs from KKN 1.2.1 (risks from the surface water pathway – Ranger current operations) and the new KKN 2.6.1 (risks associated with rehabilitation) to provide a landscape-scale context for the rehabilitation of Ranger into Kakadu National Park, and should be communicated to stakeholders.

5.2 South Alligator River valley rehabilitation

5.2.1 Assessment of past mining and milling sites in the South Alligator River valley

SSD conducts regular assessments of the status of minesites in the SAR valley, provides advice to Parks Australia on technical issues associated with its rehabilitation program and conducts a low level radiological monitoring program. This work should continue.

5.3 Develop monitoring program related to West Arnhem Land exploration activities

5.3.1 Baseline studies for biological assessment in West Arnhem Land

ARRTC believes there is a need to determine a baseline for (a) rare, threatened and endemic biota and (b) indicator species or groups such as macroinvertebrates in areas where advanced exploration or proposed mining projects are identified and in line with the current approvals process under the Aboriginal Land Rights Act**.**

5.4 Koongarra

5.4.1 Baseline monitoring program for Koongarra

In line with the current approvals process under the Aboriginal Land Rights Act, a low level monitoring program should be developed for Koongarra to provide baseline data in advance of any possible future development at the site. Data from this program could also have some relevance as a control system for comparison to Ranger, Jabiluka and Nabarlek.