supervising scientist report **201**

eriss *research summary*

2008–2009

DR Jones & AL Webb (eds)

Australian Government

Department of the Environment, Water, Heritage and the Arts

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 2008–2009 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.

Dr David R Jones – Director & Branch Head, Environmental Research Institute of the Supervising Scientist, GPO Box 461, Darwin NT 0801, Australia

Ann L Webb – Office of the Supervising Scientist, GPO Box 461, Darwin NT 0801, Australia

This report should be cited as follows:

Jones DR & Webb A (eds) 2010. ***eriss** research summary 2008–2009*. Supervising Scientist Report 201, Supervising Scientist, Darwin NT.

Example of citing a paper in this report:

Hogan A, van Dam R, Harford A, Cheng K & Turner K 2010. Effects of magnesium pulse exposures on aquatic organisms. In ***eriss** research summary 2008–2009*. eds Jones DR & Webb A, Supervising Scientist Report 201, Supervising Scientist, Darwin NT, 27–31.

The Supervising Scientist is part of the Australian Government Department of the Environment, Water, Heritage and the Arts.

Supervising Scientist
Department of the Environment, Water, Heritage and the Arts
GPO Box 461, Darwin NT 0801 Australia

ISSN 1325-1554

ISBN-13: 978-1-921069-12-3

ISBN-10: 1-921069-12-0

Printed and bound in Darwin by uniprint NT

Contents

PART 2: RANGER – REHABILITATION

KKN 2.1.1 Defining the reference state and baseline data

KKN 2.2.1 Landform design

KKN 2.2.4 Geomorphic behaviour and evolution of the landscape

KKN 2.2.5 Radiological characteristics of the final landform

KKN 2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective

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 the Environment, Water, Heritage and the Arts. ***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. A major part of its 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 uranium mining in the ARR.

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 (ARRTC – see ARRTC membership and function in Appendix 1), the Supervising Scientist, Energy Resources of Australia 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.

The KKNs comprise six thematic areas based primarily on geographic provenance (Appendix 2). The content of the research programs developed for each of these areas is assessed and reviewed annually by ARRTC in consultation with stakeholder groups.

Not all of the KKN research areas 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, or 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 Energy Resources of Australia Ltd. 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 2008–09 financial year. Much of the monitoring and research work conducted by ***eriss*** is focused on the wet season and its immediate aftermath since it is during this period that the environment is potentially at most risk from past and current uranium mining activities. By way of context the wet season rainfall of 1186 mm for 2008–09 was well below the running average of 1583 mm, with decreasing annual rainfall now having been recorded over each of three successive wet seasons (2006–07, 2540 mm; 2007–08, 1658 mm).

The U-mining-related section of the research summary has been structured under five main headings, consistent with the KKN framework:

1 Ranger – current operations

2 Ranger – rehabilitation

3 Jabiluka

4 Nabarlek

5 General Alligators Rivers Region

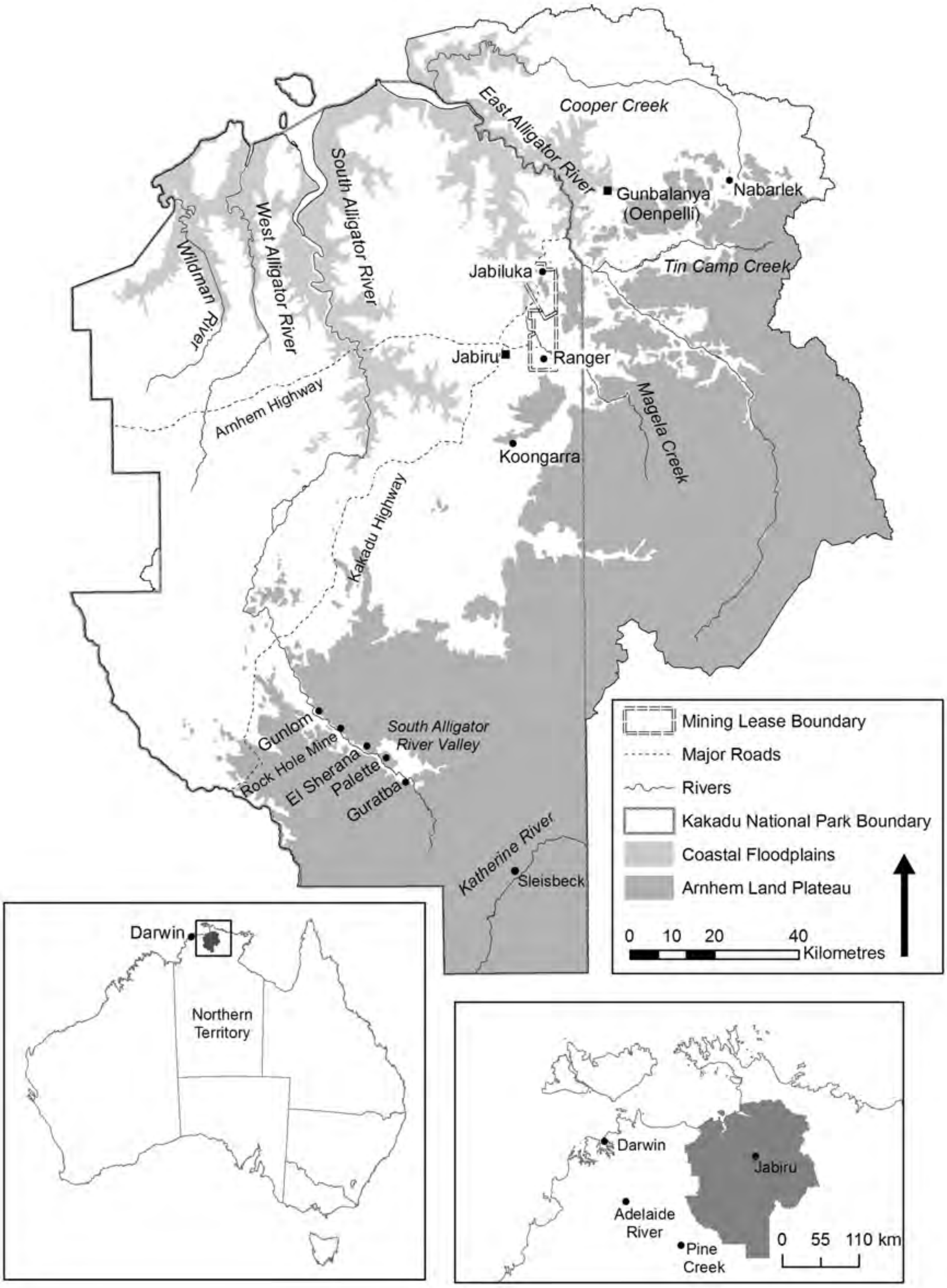
Three maps (following this Preface) provide the regional context for the locations that are referenced in the research papers. Map 1 shows Kakadu National Park and the locations of the Ranger Mine, Jabiluka project area, the decommissioned Nabarlek Mine, and the South Alligator River Valley. A schematic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

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

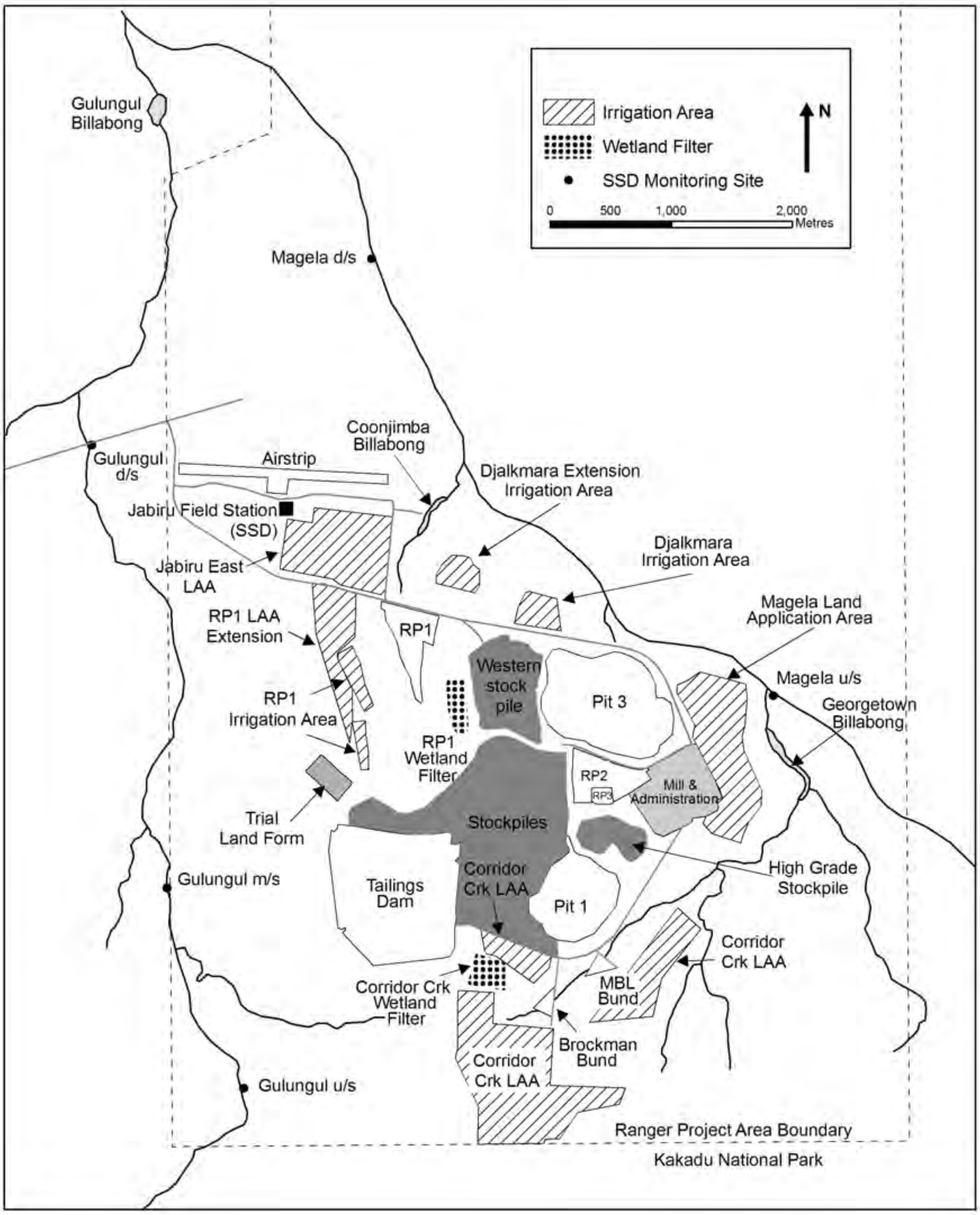
For additional information, readers are referred to the annual publications list (Appendix 3) that details all of the material published, and conference and workshop papers presented by ***eriss*** staff in 2008–09.

Dr DR Jones

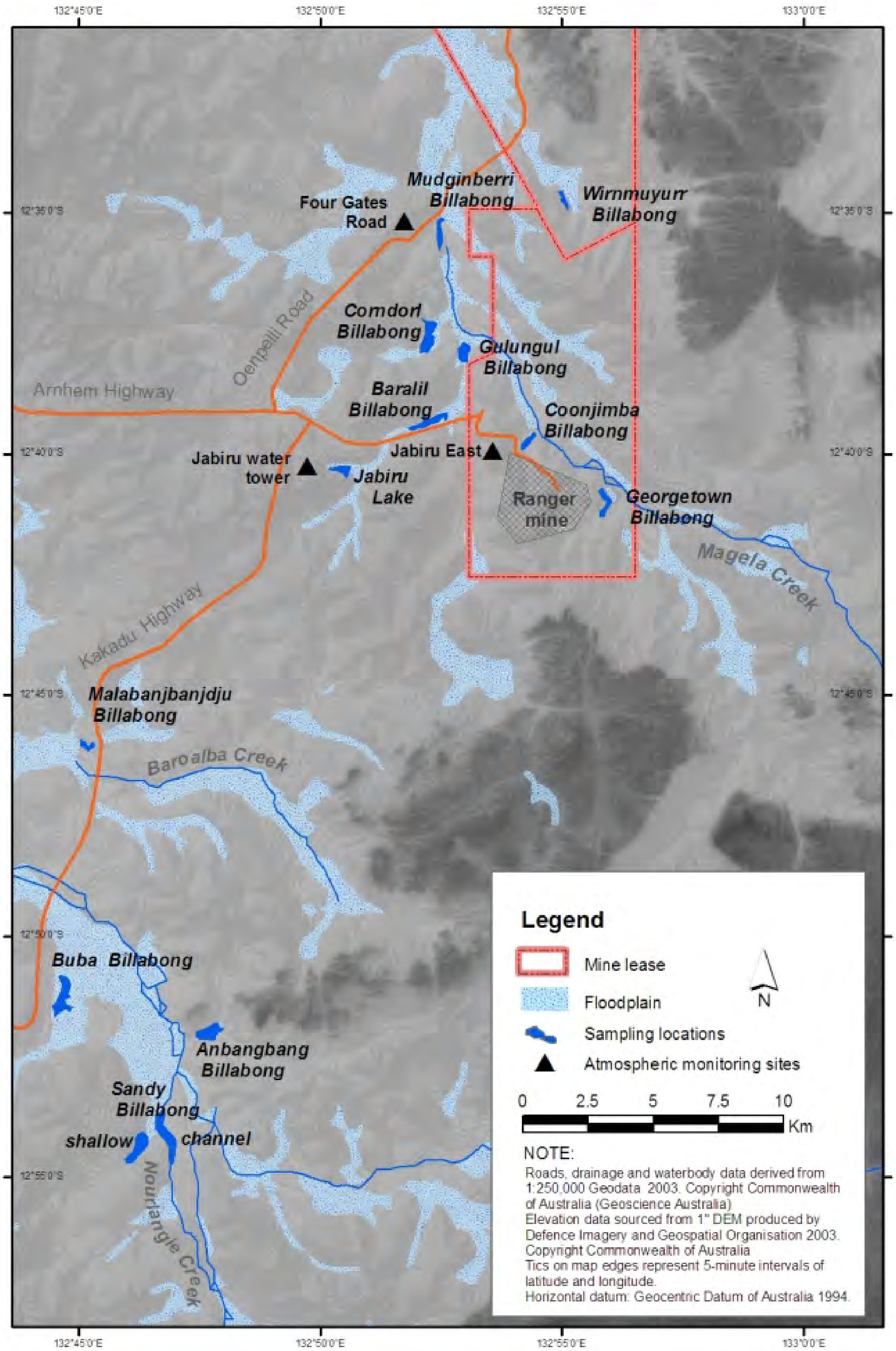
Director, Environmental Research Institute of the Supervising Scientist



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 monitoring programs

Part 1: Ranger – current operations

Contaminant pathway conceptual models for Ranger uranium mine

S Parker, R van Dam & R Bartolo

Background

In its 2004–2006 Key Knowledge Needs (KKN), ARRTC under KKN 1.2.1 stated that:

> In order to place the off-site contaminant issues at Ranger in a risk management context, a conceptual model of transport/exposure pathways should be developed. This process should include a review and assessment of the existing information on the risks of the bioaccumulation and trophic transfer (ie biomagnification) of uranium and other Ranger mining-related contaminants from all exposure pathways and including the identification of key information gaps.

To address this, a conceptual model defining the basic elements of contaminant pathways for Ranger uranium mine (Figure 1) and an associated sub-model for transport of inorganic toxicants via a direct surface water to surface water pathway (Figure 2) were progressed but not completed (van Dam & Bayliss 2006). This work built on previous conceptual modelling efforts by the Supervising Scientist (1982), Finlayson and Bayliss (2003), van Dam et al (2004) and others.

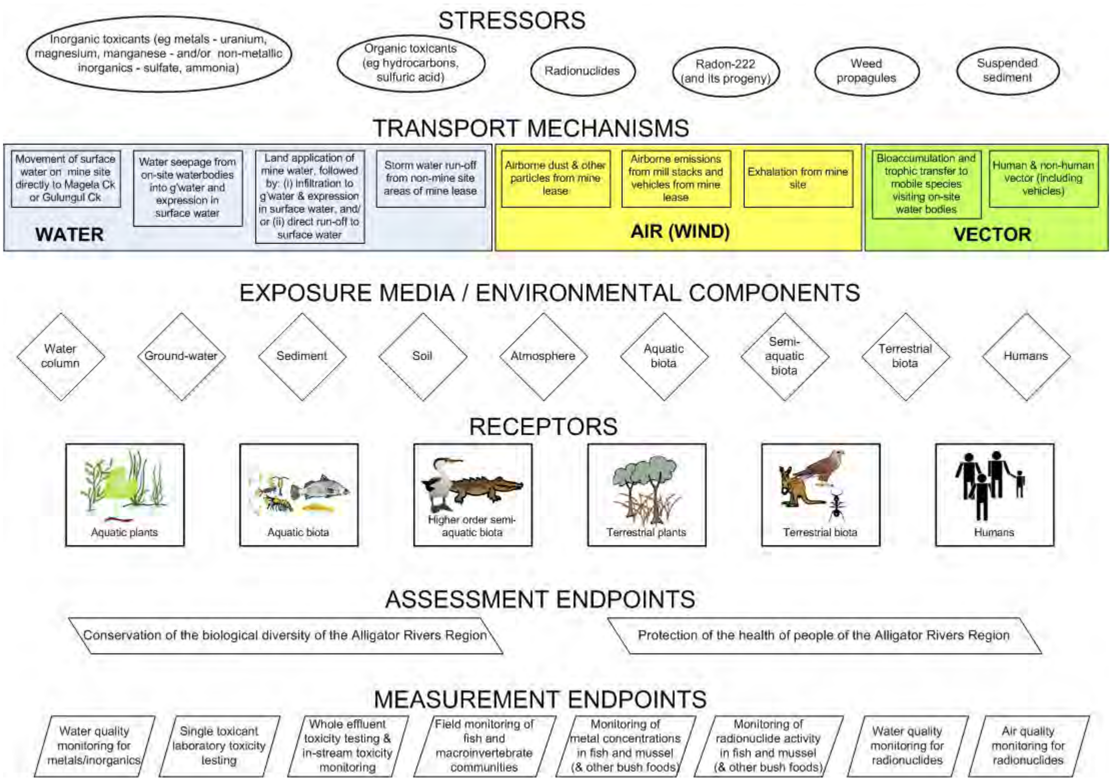


Figure 1 Basic elements of a contaminant pathways conceptual model for Ranger uranium mine (from van Dam & Bayliss 2006)

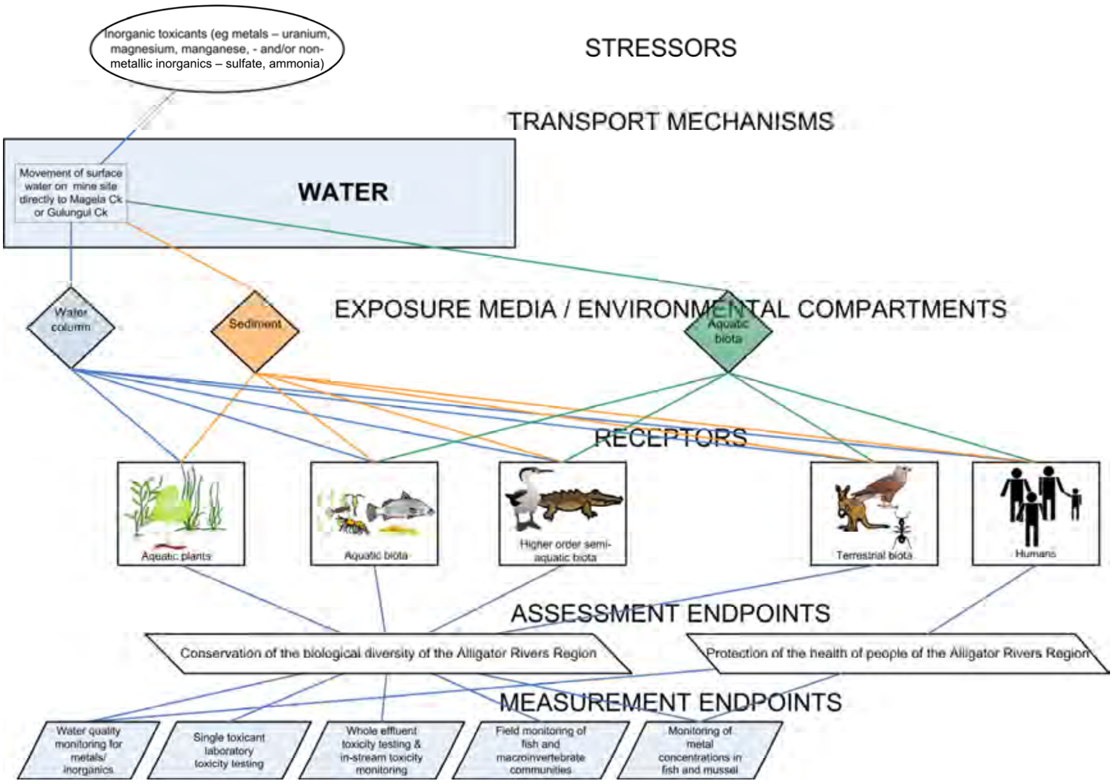


Figure 2 Conceptual model for transport of inorganic toxicants from Ranger uranium mine via a direct surface water to surface water pathway (from van Dam & Bayliss 2006)

The draft conceptual model identifies the key stressors (chemical, physico-chemical, radiological and biological contaminants) arising from Ranger and for each of these, their respective transport mechanisms off-site, affected environmental compartments, receptors, routes of exposure, types of effects and measures of effects.

In its revised 2008–2010 KKNs, ARRTC noted the work undertaken to date under KKN 1.2.1 and stated the 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'.

In recognition of the priority ARRTC and relevant stakeholders have attributed to the timely completion of the contaminant pathways conceptual models, a SSD staff member has been seconded to ***eriss*** for a period of six months to progress the project. The project is due to be completed by February 2010.

Scope

The project comprises the following two tasks:

1. developing exposure pathway sub-models for each of the identified contaminants and transport mechanisms; and
2. documenting the relevant scientific evidence for each sub-model and incorporating this into supporting narratives for each model, identifying knowledge gaps and uncertainties where possible.

A separate project will be undertaken to develop printed and computer-based communication products based on the contaminant pathways models which will assist in raising the awareness of, and providing reassurance to, relevant stakeholders regarding the actual levels of environmental and human health risks posed by each of the contaminants. The final design and functionality of the communication products and their delivery arrangements will be agreed with stakeholders once the conceptual sub-models have been completed and scientifically reviewed.

Progress and results

Senior ***eriss*** and EWL Sciences staff met on 11 September 2009 to discuss the scope of the project. EWL Sciences expressed support for the project and agreed to contribute relevant data and scientific information where possible. Draft sub-models for each of the key contaminant pathways have been prepared, based on key stressors and pathways identified by van Dam and Bayliss (2006). An internal SSD technical workshop was held on 25 September 2009 to review the draft sub-models. This was attended by ***eriss*** Program Leaders and other senior staff. The draft sub-models are being updated based on the outcomes of the workshop. Ongoing consultation with Program Leaders will be required to collate the scientific evidence and other supporting information required for the model narratives. Another workshop will be held in early 2010 to internally review the conceptual models and associated narratives. Consultation with stakeholders will also be required once the draft sub-models have been finalised.

References

Finlayson M & Bayliss P 2003. Conceptual model of ecosystem processes and pathways for pollutant/propagule transport in the environment of the Alligator Rivers Region. Discussion Paper prepared for the 11th meeting of ARRTC, 17–19 February 2003.

Supervising Scientist 1982. Submission to Australian Science and Technology Council (ASTEC). Supervising Scientist of the Alligator Rivers Region.

van Dam R, Finlayson M & Bayliss P 2004. Progress on the development of a conceptual model of contaminant pathways from Ranger uranium mine. Internal Report 474, June, Supervising Scientist, Darwin. Unpublished paper.

van Dam R & Bayliss P 2006. Development of a contaminant pathways conceptual model for Ranger mine. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 5–8.

Characterisation of contamination at land application areas at Ranger (collaborative project with EWLS)

A Bollhöfer, R Akber[1], P Lu[2], B Ryan & G Passmore[2]

Introduction

There has been ongoing stakeholder concern about the current radiological status of the Ranger land application areas (LAAs), in particular with regard to soils in the Magela LAA and their capacity to continue to adsorb radionuclides at the current rate of application. The concentration of radionuclides adsorbed in the soil may require the area to be rehabilitated at closure, based on the increase in external gamma dose rates alone. In addition there is the potential for radionuclides to be mobilised into Magela creek via erosion of the surface soil.

An EWLS report (Hollingsworth et al 2005) indicated that current application rates are likely to be sustainable in the context of the ongoing ability of the soil profile to bind radionuclides. However, this report did not address the issue of radiation doses to the public and the implications of this for the closure of the site. If levels are too high then specific rehabilitation strategies will need to be in place to minimise exposure of the public from radiation following rehabilitation of the Ranger Project Area.

A review of the Key Knowledge Needs during ARRTC 20 identified that:

> investigations are required into the storage and transport of contaminants in the land irrigation areas adjacent to Magela Creek, 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.

This risk management needs to consider current radiological issues associated with the LAAs as a result of irrigation, and provide options for the rehabilitation of the LAAs if required. These options strongly depend on estimated radiation doses to people post rehabilitation.

This project is a collaboration between Earth Water Life Sciences (EWLS), Dr Riaz Akber (*SafeRadiation*) and ***eriss***. ***eriss*** has been involved in planning and scoping the project from the early stages. A major part of the project will involve radioanalytical analyses by ***eriss*** of the different types of samples (soils, leaf litter, dust) from the Ranger LAAs and provision of assistance for the assessment of radon exhalation from the LAAs.

The aims of this project are to characterise the magnitude and extent of radiological contamination at the Ranger land application areas (LAAs) and to assist in the development of a dose model for the LAAs that can be used for rehabilitation planning.

---

[1] SafeRadiation, Brisbane

[2] Earth Water Life Sciences, Darwin

Methods

Soil samples have been collected by *SafeRadiation* and EWLS from all LAAs on the Ranger lease for measurement of radionuclide activity concentration. They have been collected at various distances from the sprinkler heads in order to determine depositional patterns and calculate the total load of radionuclides in LAA soils. Soil samples were dried and crushed, and pressed into a standard geometry for radionuclide analysis via gamma spectrometry at ***eriss***.

Leaf litter samples were also taken at various distances from the sprinkler heads and samples having the same radial distance from the sprinkler heads were combined in large plastic bags. Samples were subsequently ashed and homogenised at ***eriss*** and cast in epoxy resin for radionuclide analysis via gamma spectrometry. A subsample was sent for metal analysis via ICPMS.

Soil and leaf litter samples from the LAAs are being analysed using the ***eriss*** HPGe gamma detectors. The methods are described in Murray et al (1987), Marten (1992) and Esparon and Pfitzner (in press).

Radon exhalation was measured at the LAAs at various distances from the sprinkler heads using conventional charcoal canisters. The methods are described elsewhere (Spehr & Johnston 1983, Bollhöfer et al 2005). Surveys were conducted in July/August 2008 (dry season) and in March 2009 (wet season) by R Akber and EWLS. There was no irrigation of mine waters during and immediately prior to charcoal cup exposure. Radon cups were then analysed using the ***eriss*** NaI gamma detector.

Five passive dust collection stations were also established by *SafeRadiation* and EWLS along transects that intersect the boundaries of the Magela A and Magela B land application areas (see Map 2). The stations inside the Magela B LAA were installed such that water from the sprinklers did not directly fall on the panels. The stations are triangular in shape and approximately 2 m high. Each face of the stations has four collector panels (0.3 m height) centred at 0.3 m, 0.7 m, 1.2 m and 1.5 m above ground (Figure 1). Each of these four collector panels represents the lying down, sitting, juvenile standing and adult standing breathing zones, respectively. The stations were deployed on 16 and 17 July 2008. They were recovered on 30 September and 1 October 2008 from the Magela B land application area and 14 and 15 October 2008 from Magela A. Unfortunately some of the panels from Magela A were lost due to a bushfire. Samples were measured for total alpha activity by *SafeRadiation* in Brisbane.

Results

Soil radionuclide activity concentration

More than 200 soil samples have been analysed for radionuclide activity concentration via gamma spectrometry. The maximum measured ^{226}Ra soil activity concentration is a little above 1000 Bq kg^{-1}, and a large number of values lie in the range 100–500 Bq kg^{-1}. It was found that applied radionuclides have been generally retained in the top 5–10 cm of the soils, in agreement with earlier studies conducted in the Magela land application area (Akber & Marten 1992, Hollingsworth et al 2005). Hence the exposure pathways that depend upon the magnitude of ^{226}Ra activity concentration throughout deeper sections of the soil profile, such as external gamma exposure (most of the terrestrial gamma dose rate originates from the top 0.5 m of the soil profile) and inhalation of radon progeny (1–2 m is the typical diffusion length for radon in soil), are likely to be less significant at the LAAs than at those areas that contain waste rock up to 2000 Bq kg^{-1} to depths greater than 10 cm after rehabilitation.

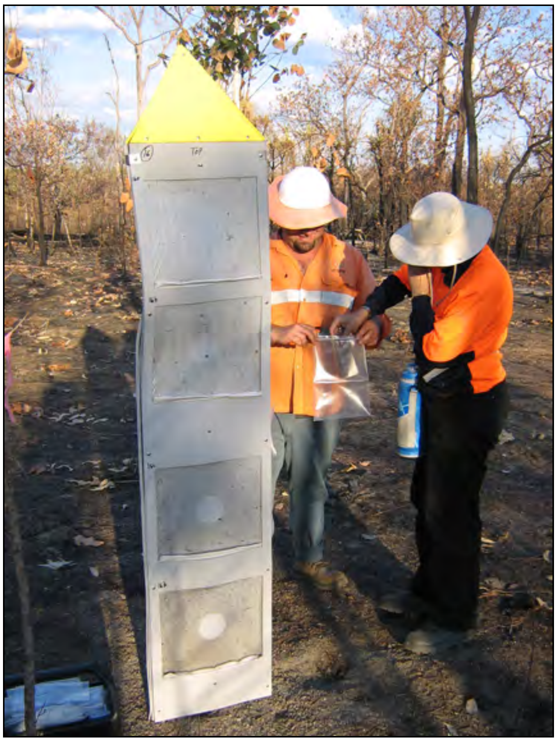


Figure 1 Passive dust sampling station

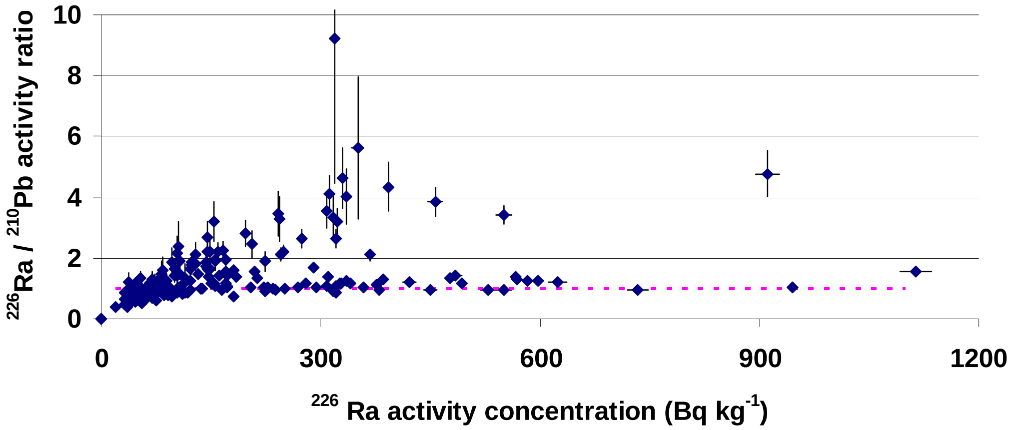


Figure 2 ^{226}Ra/^{210}Pb activity ratio plotted versus the ^{226}Ra activity concentration measured in soils from the Ranger land application areas. The dashed line indicates ^{226}Ra/^{210}Pb = 1.

Figure 2 shows the ^{226}Ra/^{210}Pb activity ratio plotted versus the ^{226}Ra activity concentration measured in the soils. For natural background soils, ^{226}Ra and ^{210}Pb are in radioactive equilibrium within the soil grains but deposition of ^{210}Pb from the atmosphere shifts the measured ^{226}Ra/^{210}Pb activity ratio to values less than one. For natural, but above background activity soils, the ^{226}Ra/^{210}Pb activity ratio should be close to one and the effect of ^{210}Pb deposited from the atmosphere on the ^{226}Ra/^{210}Pb activity ratio will be negligible. For soils subject to land application of untreated site pond water, the ratio should be greater than one as this water contains significant amounts of ^{226}Ra.. Most samples exhibit a ^{226}Ra/^{210}Pb activity ratio of $\geq$ 1 due to irrigation with these waters and most of the lower activity soils have a ^{226}Ra/^{210}Pb activity ratio < 1. It should be noted that there are some areas of relatively high ^{226}Ra and ^{210}Pb activity concentrations with ^{226}Ra/^{210}Pb activity ratios close to radioactive equilibrium, indicating that there are some areas within the LAAs that have naturally elevated

^{226}Ra activity concentrations. Ratios close to radioactive equilibrium would also be expected for areas consisting predominantly of waste rock, given the relatively high activity concentration of uranium series radionuclides (up to 2kBq kg^{-1} ^{238}U) in waste rock as compared to natural sites.

Figure 3a (left) shows the location of the soil samples collected, their ^{226}Ra/^{210}Pb activity ratio (yellow: ^{226}Ra/^{210}Pb < 0.9; red: 0.9 < ^{226}Ra/^{210}Pb < 1.1; blue: ^{226}Ra/^{210}Pb > 1.1) and a classification with regards to their ^{226}Ra activity concentration indicated by the size of the circle. Figure 3b (right) shows the same data overlaid on results from an airborne gamma survey conducted in 1976. It is apparent that soils with high ^{226}Ra activity concentration that exhibit a ^{226}Ra/^{210}Pb activity ratio of approximately 1 are located within areas that exhibited higher natural backgrounds before mining started. This is in particular obvious in samples from the Djalkmara LAA (near pit 3). The yellow circles (^{226}Ra/^{210}Pb < 0.9) are generally small in size and some of these samples are outside the zone of influence from the sprinklers.

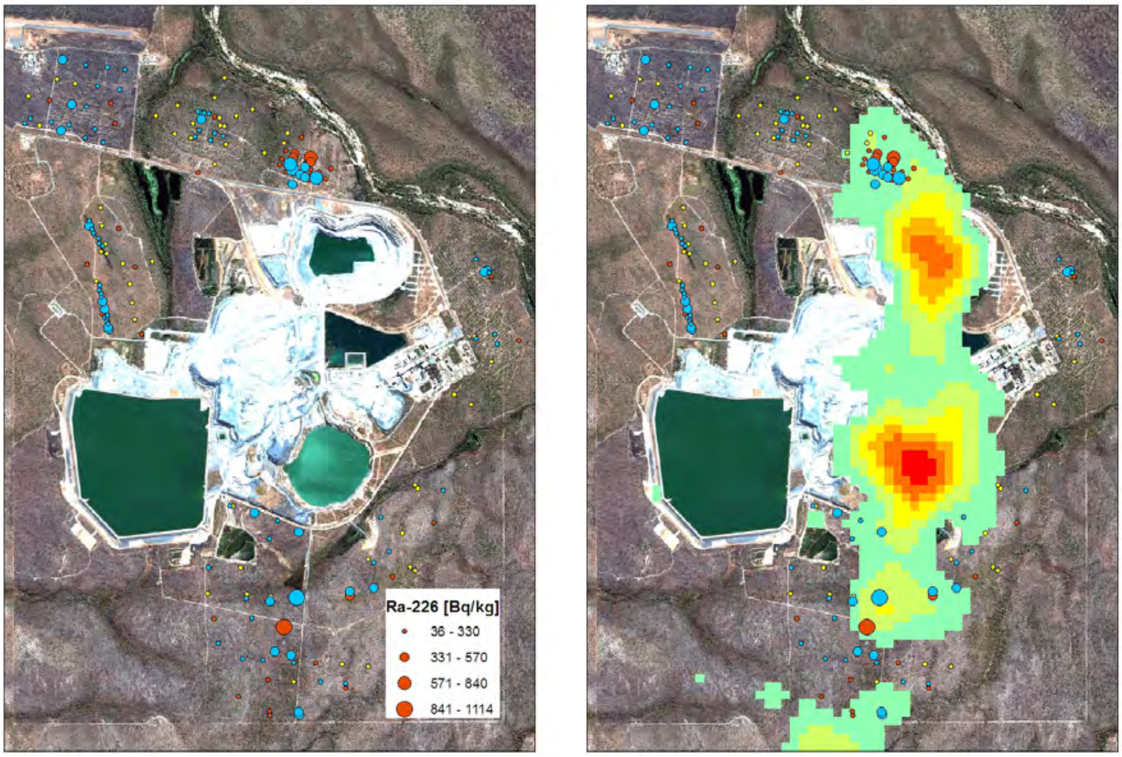


Figure 3 (a) ^{226}Ra/^{210}Pb activity ratio (yellow: ^{226}Ra/^{210}Pb < 0.9; red: 0.9 < ^{226}Ra/^{210}Pb < 1.1; blue: ^{226}Ra/^{210}Pb > 1.1) and ^{226}Ra activity concentration of the soil samples collected. (b) Data overlaid on results from an 1976 airborne gamma survey. Areas which exhibited counts per seconds in the airborne gamma survey significantly above background are indicated. Green is lowest red is highest.

The uranium activity concentration decreases approximately exponentially with distance from the sprinkler heads. This exponential decrease has been used to calculate average radionuclide activity concentrations deposited within the sprinkler halo. The overall results derived from the direct measurement of soil activities compare well with application loads calculated from historical RP2 radionuclide inventories and irrigation rates provided by ERA. The measurement of radionuclide loads in the leaf litter samples is underway and will complement the soils data set.

Preliminary analysis of the radioanalytical results produced by this study was presented by Dr Riaz Akber, consultant to Earth Water Life Sciences (EWLS), at ARRTC23 meeting, March 2009.

Radon exhalation

Dry and wet season measurements of radon exhalation rates were conducted in 2008/09. More than 200 measurements have been finalised and a summary of the results is shown in Figure 4. In this figure radon flux densities measured in the dry season (Aug 08) and wet season (Mar 09) are shown plotted versus distance from the sprinklers at various Ranger land application areas.

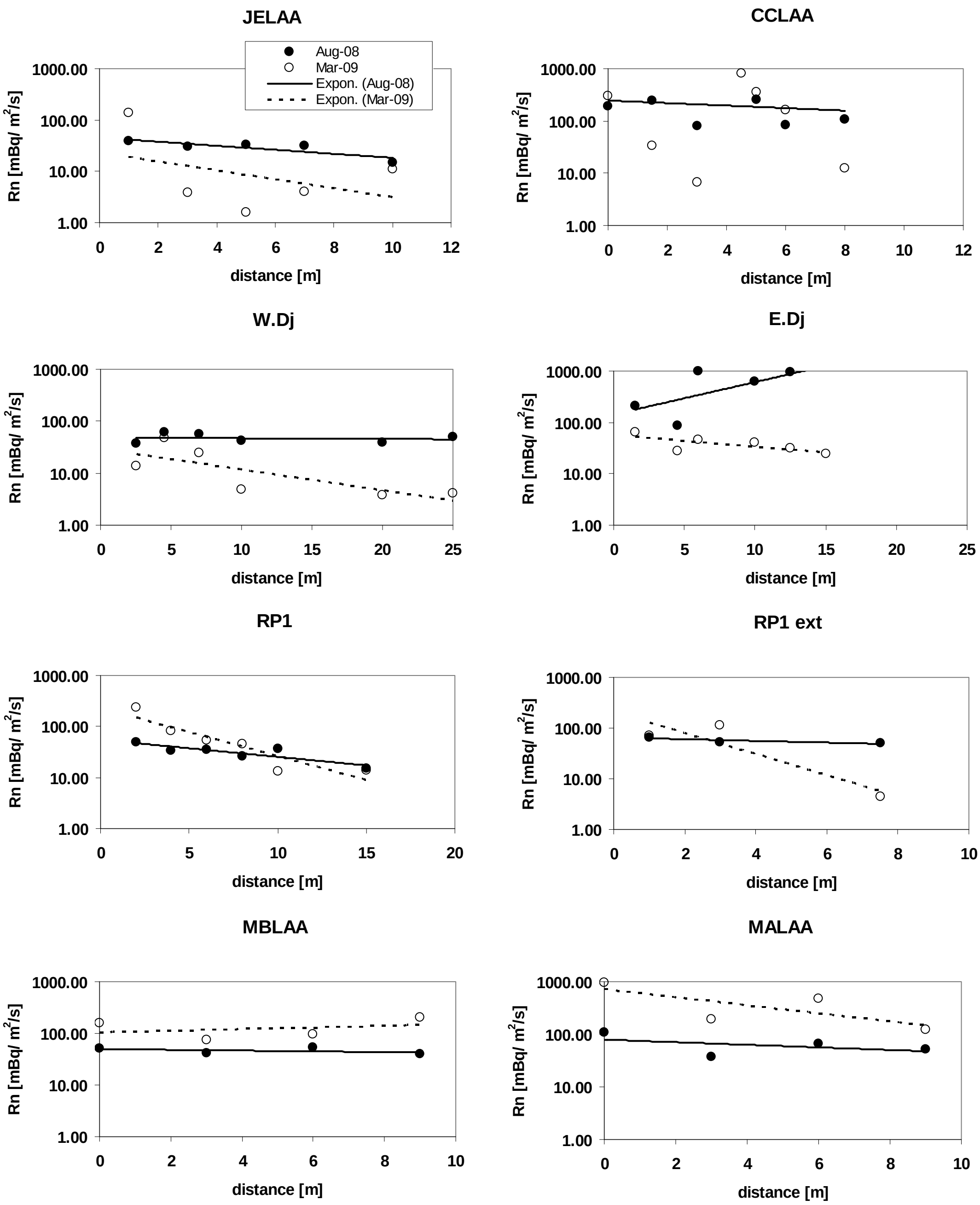


Figure 4 Radon flux densities measured in the dry season (Aug 08) and wet season (Mar 09), respectively, at various distances from the sprinklers at land application areas on the Ranger lease. The lines are exponential fits to the data.

Generally, the decrease of radon flux densities with increasing distance from the sprinklers is more pronounced during the wet season as compared with the dry. The Jabiru East (JE), Corridor Creek (CC) and Djalkmara land (W.Dj & E.Dj) application areas show higher radon flux densities during the dry season compared with the wet – due mostly to the lower soil moisture during the dry (Lawrence et al 2009). In contrast, the Magela and RP1 land application areas show higher radon flux densities in March 2009, at the end of the wet season. The reasons for these different behaviours are being investigated.

Dust

Dust samples collected by passive dust samplers have been analysed for total alpha activity by *SafeRadiation*. The analyses showed that alpha activity is generally higher in the samples closer to the ground indicating that a person sleeping may receive a higher dose from inhalation of dust than a person standing up.

Figure 5 shows the total alpha activity results from the transects in the Magela B land application area. It is apparent that there is a sharp drop of more than one order of magnitude in total alpha activity collected on the filters within the first 70 m distance outside from the LAA boundary. Total alpha activity in the breathing zone of an adult (and child) drops to about 0.01 CPM per day – a value similar to values measured at the Jabiru Field Station 4 km northwest of the transect.

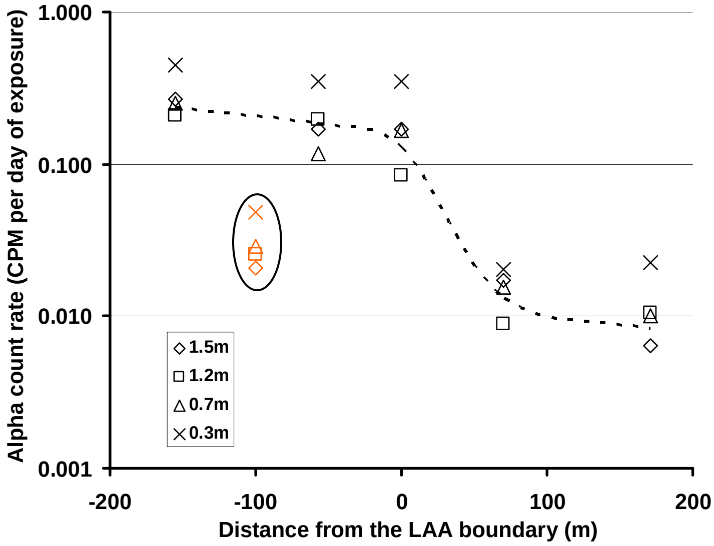


Figure 5 Total alpha activity measured in passive dust collectors along a transect in the Magela B land application area. Positive distances are outside the boundary of the LAA negative distances within. The circled values are for collectors recovered from the ***eriss*** Field Station.

Preliminary analysis of the results of this study was presented by Dr Riaz Akber at ARRTC23 meeting, March 2009.

Steps for completion

eriss will continue to provide input to the project via discussion and planning, data evaluation, interpretation and review. ***eriss*** will also continue to assist with the investigation of radionuclide transport pathways in the LAAs via the measurement of radionuclides in leaf litter samples (gamma spectrometry) and passive dust collectors (alpha spectrometry). These samples are being analysed at present and will add to the value of the present data set. Activity ratios of radionuclides collected on the passive dust collectors will provide a measure of the relative contribution of radionuclides in airborne dust due to land irrigation. ***eriss*** will also assist with the modelling of ingestion doses and provide key references and data necessary for the evaluation of this exposure pathway.

The results from this work will enable the magnitude of radiation doses from the various key exposure pathways to be determined, including the inhalation of radon progeny and dust, and via the ingestion of soil and food.

Acknowledgments

The Northern Territory Geological Survey is acknowledged for the provision of the 1976 AGS data.

References

Akber RA & Marten R 1992. Fate of radionuclides applied to soil in Ranger Uranium Mine land application area. In *Proceedings of the Workshop on Land Application of Effluent Water from Uranium Mines in the Alligator Rivers Region*. ed Akber RA, Jabiru 11–13 September 1991, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra, 139–165.

Bollhöfer A, Storm J, Martin P & Tims S 2005. Geographic variability in radon exhalation at a rehabilitated uranium mine in the Northern Territory, Australia. *Environmental Monitoring and Assessment* 114, 313–330.

Esparon A & Pfitzner J (in press). Visual Gamma – Gamma Analysis Manual. Internal Report 539, Supervising Scientist, Darwin.

Hollingsworth I, Overall R & Puhalovich A 2005. Status of the Ranger Irrigation Areas – Final Report. EWL Sciences Pty Ltd, Darwin, Australia.

Lawrence CE, Akber RA, Bollhöfer A & Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Marten R 1992. Procedures for routine analysis of naturally occurring radionuclides in environmental samples by gamma-ray spectrometry with HPGe detectors. Internal report 76, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Murray AS, Marten R, Johnston A & Martin P 1987. Analysis for naturally occurring radionuclides at environmental concentrations by gamma spectrometry. *Journal of Radioanalytical and Nuclear Chemistry*, Articles 115, 263–288.

Spehr W & Johnston A 1983. The measurement of radon exhalation rates using activated charcoal. *Radiation Protection in Australia* 1(3), 113–116.

Chronic toxicity of uranium to larval purple-spotted gudgeon (*Mogurnda mogurnda*)

K Cheng, D Parry[1], S Markich[2], A Hogan, A Harford & R van Dam

Background

During 2006–07 and 2007–08, a project was undertaken to assess the chronic toxicity of uranium (U) to the northern trout gudgeon, *Mogurnda mogurnda*.[3] The project involved the development of a 28-day larval growth (and survival) test and the subsequent use of the test to assess the toxicity of U. The key purpose of the study was to compare the U toxicity results with those of Holdway (1992) who assessed the same species but for shorter exposure durations. The previous work on this project was summarised by Cheng (2008) and Cheng et al (2008, 2009).

Two 28-d U toxicity tests were undertaken. A key finding was that U toxicity was substantially different between the two tests, with a two-fold difference between the LC50s (Table 1). Cheng et al (2009) noted that larval whole body U concentrations did not appear to explain the difference in toxicity between the tests (ie the higher toxicity in test 2 was not explainable by higher whole body U concentrations in larvae from this test). However, the whole body U concentrations may have included adsorbed as well as absorbed U and, therefore, this measurement may not have been a good indicator of U uptake. To try to further resolve the reason for the difference in U toxicity between the two tests, geochemical speciation modelling was undertaken to predict the speciation of U under the specific physico-chemical conditions of the tests.

Table 1 Summary of uranium toxicity to *M mogurnda* following 28-d exposure

Test	Endpoint	NOEC ($\mu g\ L^{-1}$ U)	IC_{10} ($\mu g\ L^{-1}$ U)	IC/LC_{50} ($\mu g\ L^{-1}$ U)
1	Survival	1400	-	2090 (1990–2200)
(June 2007)	Dry Weight	770	860 (0–1050)	>1400
	Length	770	1160 (1120–1300)	>1400
2	Survival	800	-	1070 (930–1240)
(February 2008)	Dry Weight	410	660 (600–750)	1130 (1040–1240)
	Length	410	850 (790–890)	>1200

Methods

The speciation of U in the test solutions was calculated using the HARPHRQ geochemical speciation code. Input parameters were based on physico-chemical data measured in the toxicity test solutions (Table 2). The binding of U with dissolved organic carbon (DOC) was calculated as a function of its binding with fulvic acid (FA), the major component (~90%) of

[1] Australian Institute of Marine Science (AIMS NT), Arafura Timor Research Facility, PO Box 41775 Casuarina NT 0811

[2] Aquatic Solutions International, Cammeray, NSW

[3] Project undertaken under Charles Darwin University Animal Ethics Approval Ref No. A06008

humic substances in natural fresh surface waters, using a finite mixture of simple organic ligands (Markich et al 2000). This approach has been shown to provide a very good measure of U binding with natural FA (and hence DOC) in natural Magela Creek water (NMCW) from pH 5.0–6.5 (Markich & Brown 1999, Markich et al 2000).

Table 2 Water chemistry of natural Magela Creek water used for uranium chronic toxicity tests[a]

Physico-chemical variable	Natural Magela Creek water (NMCW)	
	Test 1	**Test 2**
pH	6.58 (C), 6.68, 6.70, 6.70, 6.72, 6.72, 6.75[b]	6.26 (C), 6.22, 6.12, 5.98, 5.75, 5.73, 5.67[b]
EC (μS cm^{-1})	19–23[c]	24[c]
Alkalinity (mg L^{-1} as $CaCO_3$)	7	5
DOC (mg L^{-1})	2.1	4.2
NH_3 (mg L^{-1})	<0.5[d]	<0.5[d]
Na (mg L^{-1})	1.38	0.94
K (mg L^{-1}) [e]	0.37	0.37
Ca (mg L^{-1})	0.47	0.23
Mg (mg L)	0.74	0.27
Cl (mg L^{-1})	1.3	1.3
SO_4 (mg L^{-1})	0.20	0.20
NO_3 (μg L^{-1}) [e]	15	15
PO_4 (μg L^{-1}) [e]	7.0	7.0
Fe (μg L^{-1})	189	128
Al (μg L^{-1})	18	56
Cd (μg L^{-1})	<0.02	<0.02
Co (μg L^{-1})	0.03	0.11
Cr (μg L^{-1})	<0.1	0.17
Cu (μg L^{-1})	0.48	0.48
Mn (μg L^{-1})	0.81	5.5
Ni (μg L^{-1})	0.29	0.53
Pb (μg L^{-1})	0.05	0.13
Se (μg L^{-1})	0.20	0.20
U (μg L^{-1})	0.02	0.02
Zn (μg L^{-1})	1.58	4.5

a Unless otherwise stated, all values represent a single measurement from the batch of NMCW that was used for each of the toxicity tests.

b Values represent the mean measurements for the control (C) and increasing U treatments thereafter (n = 3).

c Values represent the range of measurement across the control and all U treatments.

d Values are based on daily measurements throughout the toxicity tests.

e Long-term (~25 years) mean values for NMCW; these were used for the speciation calculations as they were not measured as part of the present study.

Progress

Uranium speciation calculations for both toxicity tests are summarised in Table 3. In both tests, the majority of U was bound to FA/DOC. Given the higher DOC concentration in Test 2 (4.2 mg/L) compared with Test 1 (2.1 mg/L), the proportion of UO_2FA was higher in Test 2.

Whereas the proportion of UO_2FA decreased with increasing U concentration, the proportions of other key U species increased with increasing U concentration. The free uranyl ion, UO_2^{2+}, represented only a small proportion of the total U. However, in Test 2, the proportions of U as UO_2^{2+} at U concentrations of greater than 800 µg L^{-1} were at least 2.5 to 10 times greater than at similar concentrations in Test 1. The percentages of U as UO_2^{2+} at the LC_{50} concentrations for Test 1 (2090 µg L^{-1}) and Test 2 (1070 µg L^{-1}) were 0.63% and 1.26%, respectively, corresponding to UO_2^{2+} concentrations of 13.2 and 13.5 µg L^{-1}, respectively. When percentage survival of *M. mogurnda* larvae was plotted against UO_2^{2+} concentration, the responses of the larvae were very similar (ie standardised for the bioavailable UO_2^{2+} fraction) between the two toxicity tests (Figure 1).

Table 3 Calculated speciation of uranium in the two uranium chronic toxicity tests

U species	Speciation (% of total U)						
Test 1	***U (µg L⁻¹)***						
	0.02	***90***	***184***	***381***	***768***	***1397***	***3182***
UO_2^{2+}	0.38	0.34	0.40	0.58	0.69	0.68	0.55
UO_2OH^+	6.1	5.9	6.4	10	21	20	17
$UO_2(OH)_2$	0.80	1.0	1.4	1.7	2.1	2.1	2.0
UO_2CO_3	5.7	5.8	6.3	9.9	12	12	8.9
$(UO_2)_3(OH)_5^+$	<0.1	<0.1	0.22	0.38	3.0	10	24
UO_2Fa[a]	87	87	85	77	61	53	47
Test 2	***U (µg L⁻¹)***						
	0.03	***406***	***803***	***1226***	***1576***	***2070***	***2242***
UO_2^{2+}	0.10	0.24	0.56	1.7	4.4	4.9	6.2
UO_2OH^+	0.70	1.0	1.7	3.3	7.2	9.5	14
$UO_2(OH)_2$	0.07	0.10	0.24	0.80	2.0	2.3	2.7
UO_2CO_3	0.10	0.24	0.54	1.6	4.3	4.8	5.6
UO_2FA	99	98	97	93	82	78	71

a FA: fulvic acid

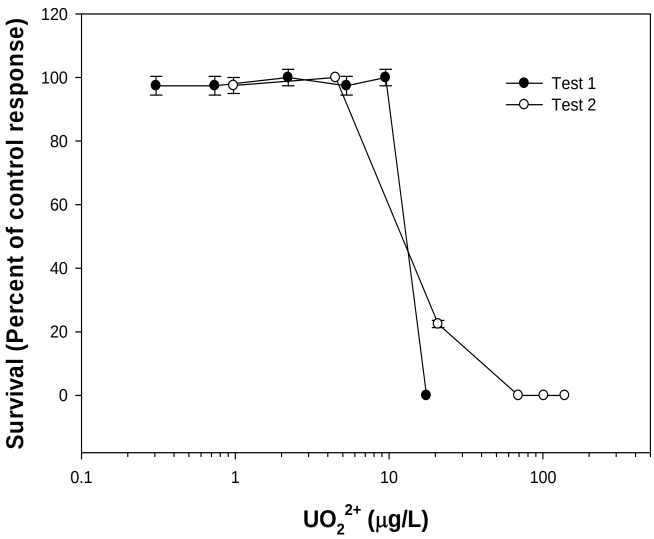


Figure 1 Mean (±SEM) survival (as a function of control survival) of *M. mogurnda* following 28 d exposure to uranyl ion, UO_2^{2+}, as calculated by HARPHRQ, for the two uranium chronic toxicity tests

The key driver of the difference in U speciation between the two tests was most likely the pH of the test solutions, which was (on average) 0.7 units lower in Test 2 (mean pH 6.0) compared to Test 1 (mean pH 6.7). The lower pH in Test 2 resulted in more bioavailable U and, therefore, higher toxicity, than at the lower pH in Test 1. The free uranyl ion is the predominant species at pH ≤5.0 and becomes less significant above pH 6.0 as complexation with hydroxides and carbonates increases (Markich 2002, Fortin et al 2007; as shown in Table 3). Dissolved organic carbon complexes with U and reduces its toxicity to freshwater biota (Markich et al 2000, Hogan et al 2005, Houston et al 2009). Although the two-fold higher DOC concentration in Test 2, compared with Test 1, is predicted to result in a higher proportion of U present as UO_2FA, this effect was not sufficient, at higher U concentrations, to counter the increase in the proportion of U as UO_2^{2+} as a result of the lower pH. Hence, the pH difference between the two tests had a much greater influence on U toxicity than did the difference in DOC concentration.

Steps for completion

This project is completed. A manuscript is currently in review with *Aquatic Toxicology*.

References

Cheng KL 2008. The development and application of a 28 day larval fish toxicity test. Research thesis, BSc (Hons), Charles Darwin University, Darwin NT, Internal Report 535, June, Supervising Scientist, Darwin. Unpublished paper.

Cheng K, van Dam R, Hogan A & Parry D 2008. Chronic toxicity of uranium to larval purple-spotted gudgeon (*Mogurnda mogurnda*). In ***eriss** research summary 2006–2007*, eds Jones DR, Humphrey C, van Dam R & Webb A, Supervising Scientist Report 196, Supervising Scientist, Darwin NT, 8–10.

Cheng K, Parry D, Hogan A & van Dam R 2009. Chronic toxicity of uranium to larval purple-spotted gudgeon (*Mogurnda mogurnda*). In ***eriss** research summary 2007–2008*, eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 2–5.

Fortin C, Denison FH, Garnier-Laplace J 2007. Metal-phytoplankton interactions: Modeling the effect of competing ions (H^+, Ca^{2+}, and Mg^{2+}) on uranium uptake. *Environmental Toxicology & Chemistry* 26, 242–248.

Hogan A, van Dam R, Markich S & Camilleri C 2005. Chronic toxicity of uranium to a tropical green alga (*Chlorella* sp) in natural waters and the influence of dissolved organic carbon. *Aquatic Toxicology* 75, 343–353.

Holdway DA 1992. Uranium toxicity to two species of Australian tropical fish. *The Science of the Total Environment* 125, 137–158.

Houston M, Ng J, Noller B, Markich S & van Dam R 2009. Influence of dissolved organic carbon on the toxicity of uranium. In ***eriss** research summary 2007–2008*, eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 6–11.

Markich SJ 2002. Uranium speciation and bioavailability in aquatic systems: an overview. *The Scientific World Journal* 2, 707–729.

Markich SJ & Brown PL 1999. Thermochemical data for environmentally-relevant elements. ANSTO/E735. Australian Nuclear Science and Technology Organisation, Sydney.

Markich SJ, Brown PL, Jeffree RA & Lim RP 2000. Valve movement responses of *Velesunio angasi* (Bivalvia: Hyriidae) to Mn and U: An exception to the free ion activity model. *Aquatic Toxicology* 51, 155–175.

Amelioration of uranium toxicity by dissolved organic carbon

M Houston, 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 the toxicity of uranium (U), aluminium (Al) and arsenic (As) to freshwater biota.

With an increase in U mining, milling, use and disposal comes increased concern of the risks to human and ecological health due to U toxicity (Lesher et al 2008). Both historical and current U mining operations in tropical Australia may result in increased amounts of U in aquatic ecosystems, which, with its high capacity for solubilisation and migration in natural water (Morse & Choppin 1991), may pose a risk to aquatic biota. Uranium is of specific relevance for the Magela Creek system adjacent to the Ranger mine.

In 2007–08, the influence of a standard DOC on U toxicity to three Australian tropical freshwater species, the northern trout gudgeon (*Mogurnda mogurnda*), green hydra (*Hydra viridissima*) and unicellular green alga (*Chlorella* sp), was measured in synthetic Magela Creek water (SMCW), the composition of which is characteristic of sandy braided streams in tropical Australia. The DOC used for this work was the Suwannee River Fulvic Acid Standard I (SRFA) produced by the International Humic Substances Society (IHSS). The SRFA was selected for initial evaluation of the effects of DOC because it is an international reference material whose composition and properties has been extensively characterised (Cabaniss & Shuman 1988). A fulvic acid (FA) was selected because FAs account for a large proportion of DOC in natural fresh waters (Maurice & Namjesnik-Dejanovic 1999). The results obtained for the SRFA were summarised in Houston et al (2009).

The objective of the second phase of the U toxicity component of the PhD research program is to compare the influence on U toxicity to three tropical freshwater species of a standard riverine DOC source (Suwannee River fulvic acid) with that of a local DOC source from the Alligator Rivers Region (Sandy Billabong water).

Methods

In 2008–09, the toxicity of U to the three species listed above was assessed using DOC-rich natural water from Sandy Billabong (SBW) located in Kakadu National Park. This site was selected based on its location within the Alligator Rivers Region and for the high DOC content (10 mg/L) of the water (***eriss***, unpublished data). The DOC in this water is produced primarily by leaching of leaf litter, and release from coarse particulate matter, soil, bark and twigs (O'Connell et al 2000). Different concentrations of DOC (0, 1, 5 & 10 mg/L) in SBW were obtained by diluting SBW with SMCW, which is of very similar ionic composition to SBW but lacks DOC. Two tests, each comprising the four DOC concentrations in

[1] National Research Centre for Environmental Toxicology, The University of Queensland, Coopers Plains QLD 4108
[2] Centre for Mined Land Rehabilitation, The University of Queensland, St Lucia, QLD 4067
[3] Aquatic Solutions International, Cammeray, NSW 2062

combination with a range of U concentrations, were conducted for each species (under fixed conditions of pH, water hardness and alkalinity).

Test durations and endpoints were as follows: *M. mogurnda* – 96-h sac-fry survival; *H. viridissima* – 96-h population growth rate; *Chlorella* sp – 72-h population growth rate. For all tests, pH, dissolved oxygen and electrical conductivity were monitored daily. 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 two tests were pooled, and concentration-response relationships were determined. Physico-chemical variables were input into the HARPHRQ geochemical speciation computer model to determine the effect of DOC on U speciation to ascertain if the proportion of U bound by the DOC could account for the observed changes in U toxicity. For the initial modelling calculations presented here, the binding constant (logK) value published for SRFA (Glaus et al 2000) was also used for SBW. The speciation calculations for the SBW will be run again once the specific U binding constant for SBW FA has been determined.

To enable characterisation of the FA in SBW, 120L of SBW was transported to the Curtin Water Quality Research Centre at Curtin University, Perth. The FA fraction was isolated from the water according to the procedures outlined by the International Humic Substances Society (Thurman & Malcolm 1981). Amberlite© XAD-8 resin was used to initially adsorb the humic acids (HAs) and FAs, which through a series of adsorptions and elutions from XAD-8 resin, were concentrated and had salts removed (using a cation exchange resin). The purified concentrated solutions of HAs and FAs were then finally freeze dried to yield the solid products (approximately 5 mg and 80 mg respectively). The physico-chemical characteristics of the SBW FA were compared with those of SRFA using the following techniques:

- Elemental analysis by Chemical & Microanalytical Services (to determine % C, H, N, S & O);
- Size exclusion chromatography by Curtin Water Quality Research Centre (to determine molecular weight);
- ^{13}C NMR spectroscopy by Monash University (to compare proportions of functional groups);
- Acid/base titrations by University of Szeged, Hungary (to obtain estimates of the content of proton-binding sites)
- Fourier Transform Infra-Red spectroscopy by Environmental Chemistry University of New Mexico (to obtain a quantitative measurement of carboxylic acid content to be used to calculate the U binding constant for SBW FA).

Progress

Table 1 summarises the reduction in U toxicity that occurred with increasing concentrations of SBW and compares it with the amelioration that occurred for SRFA, all tests using SMCW as the diluting medium. Figure 1 compares linear regressions of U toxicity (expressed as IC/LC50) against DOC (expressed as the molar concentration of the FA) for each of the three test species, for SRFA and SBW. FA (M) was calculated by adjusting DOC concentration (mg/L) for the % C of each FA, converting this to g/L and dividing by the molecular weight of the FA.

Table 1 Effect of local (sandy billabong water; sbw) and standard (Suwannee River Fulvic Acid Standard i; srfa) dissolved organic carbon (doc – at 0 and 10 mg/l), on the toxicity of uranium to three local freshwater species

Species	DOC[a] (mg/L)	IC_{50}[b] (95%CL)[c]		Rate of amelioration of U toxicity (µg/L U µM FA^{-1})[f]		Difference (SRFA:SBW)
		SBW[d]	SMCW+SRFA[e]	SBW	SMCW+SRFA	
Mogurnda mogurnda[g] (northern trout gudgeon)	0	1690 (1499–1964)	1550 (1057–1961)	77	129	1.7x
	10	3093 (2829–3459)	4330 (4152–4575)			
Hydra viridissima (green hydra)	0	50 (18–81)	65 (8–85)	4	11	2.8x
	10	115 (44–191)	310 (247–491)			
Chlorella sp (unicellular alga)	0	15 (8–24)	38 (22–69)	7.5	15	2x
	10	144 (114–168)	394(248–766)			

a DOC: dissolved organic carbon, b IC_{50}:this is the concentration that results in a 50% inhibition of the test response relative to the control response; c 95% confidence limits; d SBW: Sandy Billabong water; e SMCW + SRFA: Synthetic Magela Creek water with Suwannee River fulvic acid; f Rates calculated from Figure 1 regression slopes; g For *M. mogurnda*, toxicity estimates relate to concentrations that affect percentage survival (as a % of control survival), compared with sub-lethal endpoints, such as growth and reproduction, for the other species

U toxicity was reduced approximately 10-fold for *Chlorella*, and 2-fold for *M. mogurnda* and *H. viridissima*, in SBW containing 10 mg/L DOC compared with SMCW lacking DOC. SRFA was twice as effective as SBW at reducing U toxicity for all three species (see Table 1). Preliminary geochemical speciation modelling showed that both forms of DOC resulted in the formation of similar proportions of UO_2 FA complex (Figure 2).

Characterisation of the SBW FA showed that it has properties similar to that of SRFA and many other aquatic FAs. The two FAs were found to be similar in molecular weight, elemental composition (Table 2) and their proportion of acidic (primarily carboxylate) functional groups responsible for metal complexation (Figure 3 – note that this comparison shows relative, not absolute, values). Acid/base titration also showed that the FAs have a similar proportion of proton-binding sites (not shown). Work is currently underway to quantitatively measure the carboxylic acid content of SBW FA, as this may explain why this FA reduces U toxicity to a lesser extent than SRFA.

Table 2 Comparison of molecular weight (mn) and elemental composition of Sandy Billabong fulvic acid (SBW FA) and Suwannee River fulvic acid (SRFA)

FA	Mn (d)	% C[a]	% H[a]	% N[a]	% S	% O
SRFA	856	49.9	4.6	0.6	0.33	43.6
SBW FA	1075	51.5	4.8	1.4	0.39	39.5

[a] Uncertainty associated with C, H, N measurements is ± 0.1-0.3%

Both the SRFA standard and SBW FA resulted in a reduction in U toxicity to the three freshwater species studied. At this stage the SRFA's ability to ameliorate U toxicity more than that of the SBW FA could not be explained by differences in UO_2^{2+}.FA complexation (as calculated by the speciation modelling). Using a binding constant specific for the SBW FA may however significantly alter the speciation results for the SBW tests.

Based on characterisation studies conducted so far, the characteristics of the two FAs appear to be quite similar. If final speciation modelling does not suggest a difference in the FAs capacity to complex U, the lesser degree of amelioration of U toxicity occurring in the billabong water may be due to the influence of other components present in Sandy Billabong water.

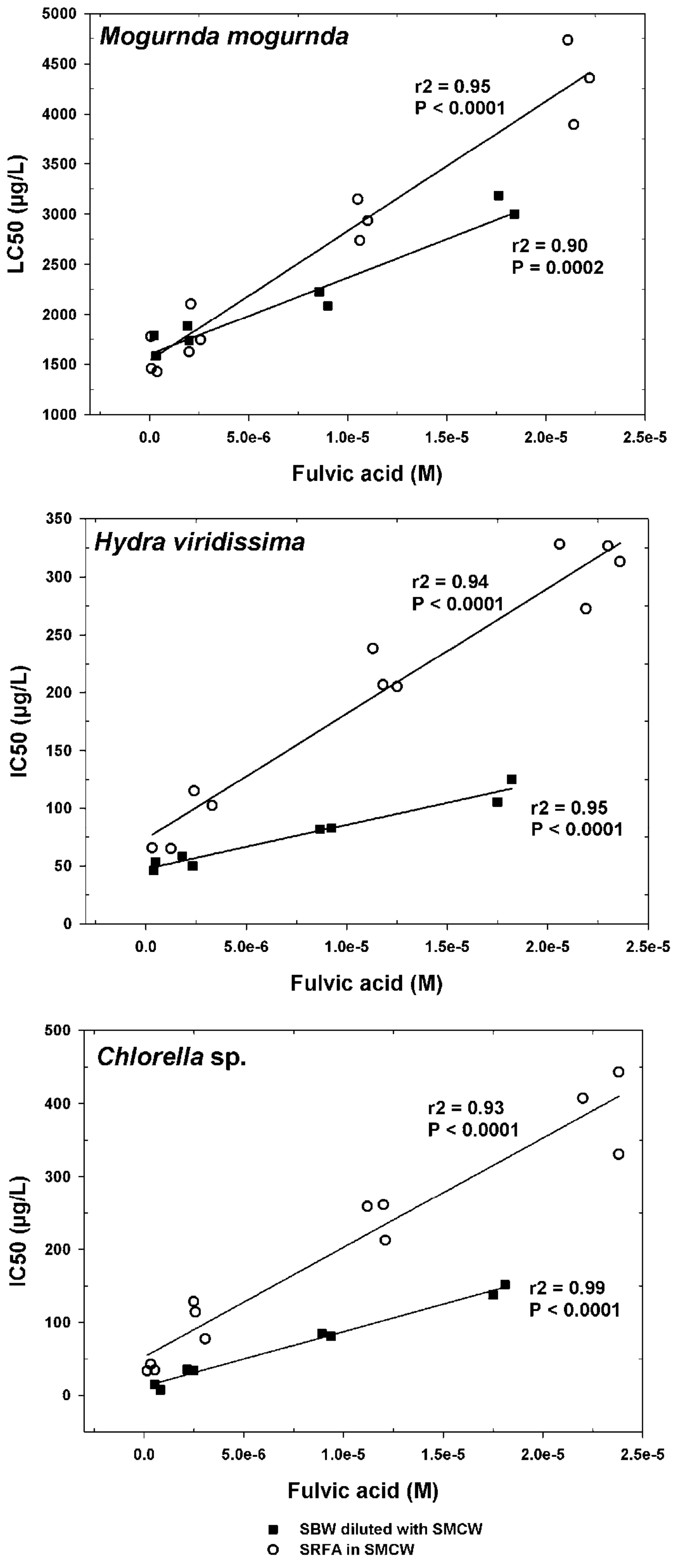


Figure 1 Effect of increasing DOC (expressed as fulvic acid concentration) on U toxicity to *Mogurnda mogurnda*, *Hydra viridissima* and *Chlorella* sp. Fitted linear regressions have been overlaid. Sandy Billabong water (SBW) diluted with synthetic Magela Creek water (SMCW); Suwannee River fulvic acid (SRFA) in SMCW.

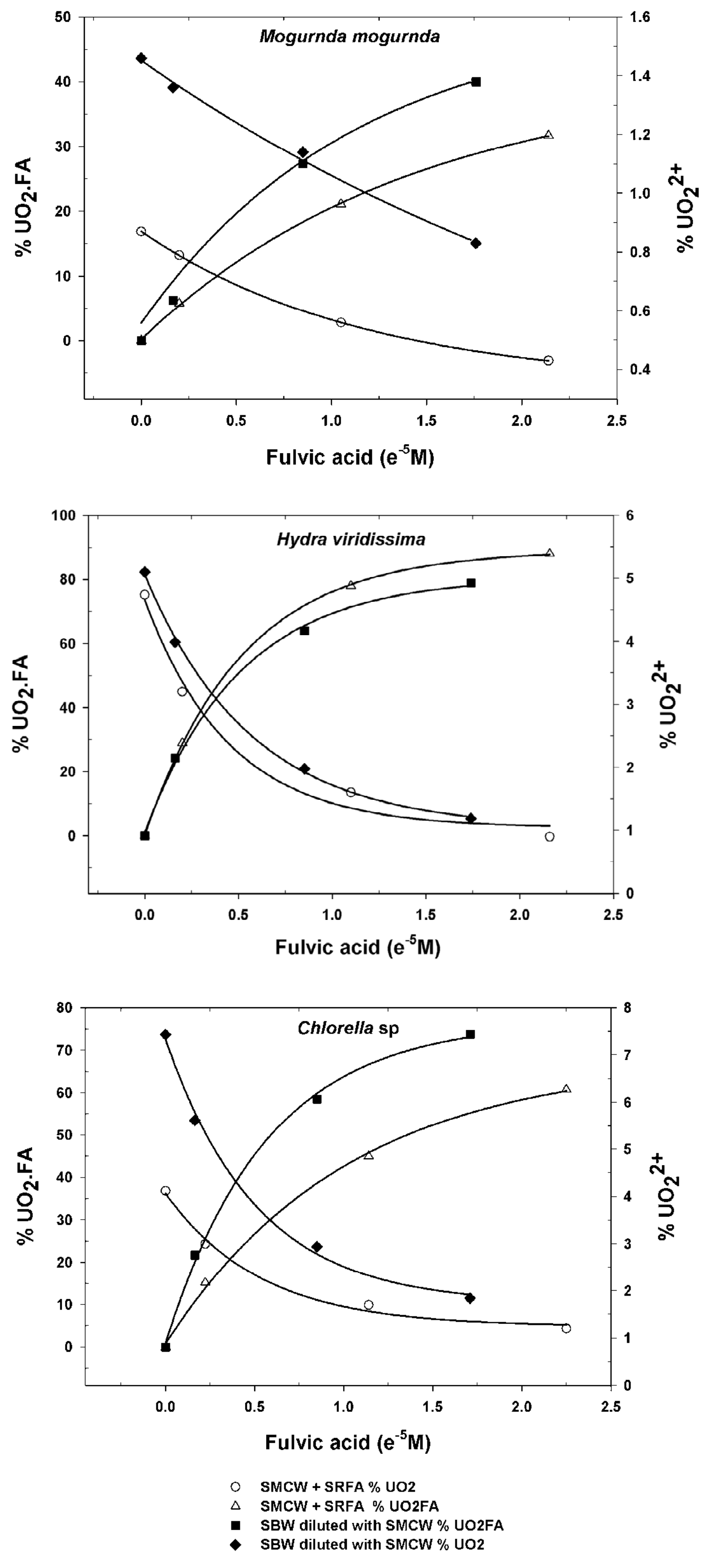


Figure 2 The proportion of UO_2^{2+} and UO_2.FA complex formed both in Synthetic Magela Creek Water with Suwannee River Fulvic Acid (SMCW + SRFA) and in Sandy Billabong Water (SBW) diluted with SMCW for *Mogurnda mogurnda*, *Hydra viridissima* and *Chlorella* sp. U speciation was calculated using HARPHRQ Version 1.02.

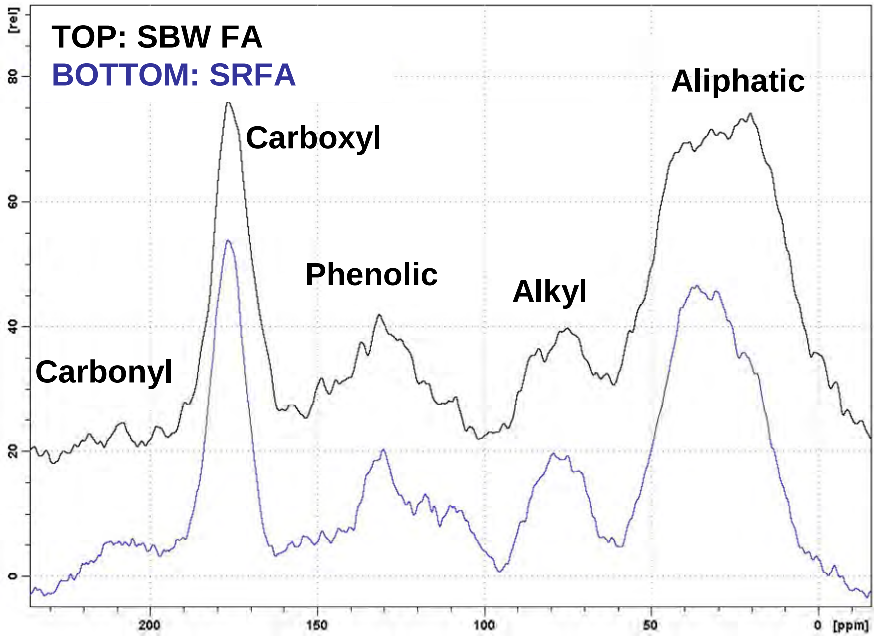


Figure 3 ^{13}C NMR results showing proportions of the major functional groups for Suwannee River Fulvic Acid (SRFA) and Sandy Billabong Water (SBW FA). Analysis conducted by Jenny Pringle, Monash University.

The evidence of attenuation of U toxicity by a local DOC source has important implications for impacted billabongs on the Ranger lease, where DOC concentrations can reach 20 mg/L, which is considerably higher than in Magela Creek (typically 1–5 mg/L). Consideration of the effects of DOC in ameliorating toxicity will be required as part of the process for deriving water quality closure criteria for U in these waterbodies.

Work will continue to derive a U binding constant for the Sandy Billabong FA for use in the speciation modelling. It may also be possible to validate the results of the speciation modelling using field flow fractionation in combination with ICPMS to measure the concentrations of uranyl ion and UO_2^{2+}.FA complex present in each of the water types at various DOC concentrations (Ranville et al 2007). U toxicity testing work will be completed using the freshwater unicellular species, *Euglena gracilis* to investigate cellular mechanisms of U toxicity. Measurements of Al toxicity are almost complete for the three species and will be completed using all four species of test organisms. The results from this work will be published in the peer-reviewed literature and documented in subsequent annual reports.

References

Cabaniss SE & Shuman MS 1988. Copper binding by dissolved organic matter: I. Suwannee River fulvic acid equilibria. *Geochimica et Cosmochimica Acta* 52, 185–193.

Glaus M, Hummel W & Van Loon L 2000. Trace metal-humate interactions. I. Experimental determination of conditional stability constants. *Applied Geochemistry* 15, 953–973.

Houston M, Ng J, Noller B, Markich S & van Dam R 2009. Influence of dissolved organic carbon on the toxicity of uranium. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 6–11.

Lesher E, Ranville J & Honeyman B 2008. Hyphenation of field flow fractionation with ICP-MS to determine Pore-Scale Uranium (VI) Speciation. Oral Presentation (presented by

Ranville) at the 5th Congress Society for Environmental Toxicology and Chemistry World Congress, Sydney, Australia, August 2008.

Maurice P & Namjesnik-Dejanovic K 1999. Aggregate structures of sorbed humic substances observed in aqueous solution. *Environmental Science & Technology* 33, 1538–1541.

Morse JW & Choppin GR 1991. The chemistry of transuranic elements in natural waters. *Review of Aquatic Sciences* 4, 1–22.

O'Connell M, Baldwin D, Robertson A. & Rees G. 2000. Release and bioavailability of dissolved organic matter from floodplain litter: influence of origin and oxygen levels. *Freshwater Biology* 45, 333–342.

Ranville JF, Hendry MJ, Reszat TN, Xie Q & Honeyman BD. 2007 Quantifying uranium complexation by groundwater dissolved organic carbon using asymmetrical flow field-flow fractionation. *Journal of Contaminant Hydrology* 91, 233–246.

Thurman E & Malcolm R 1981. Preparative isolation of aquatic humic substances. *Environmental Science & Technology* 15, 463–466.

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

K Cheng, R van Dam, A Hogan, A Harford, C Costello, D White & M Houston

Background

Over the past five years, and 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. Since 2004–05, reference toxicant control charts have been developed for four of the five routine testing species. The aims for 2008–09 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* reference toxicity test with the objective of establishing an acceptably stable reference test

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, 15 reference toxicants tests (*Chlorella* – 3; *Hydra* – 5; *Moinodaphnia* – 4; and *Mogurnda* – 3) were completed during 2008–09. Ten of these tests provided valid results, as summarised in Table 1. The associated control charts are presented in Figure 1.

A summary of the issues identified during 2008–09 for each component of the reference toxicity test program is provided below.

Chlorella sp

Only one of three *Chlorella* sp tests was valid for the period. Two of the tests were not valid with both having growth rates lower than the acceptability criterion of 1.4 ± 0.3 doublings/day (Riethmuller et al 2003). There were no problems with general test conditions (ie temperature, lighting, EC, pH and DO) and it was suspected that low nutrient concentrations were the cause of lower than normal growth. Investigations into nutrient stock solutions are on-going with more details to follow in the 2009–10 summary. The latest advice from Northern Territory Environmental Laboratories (NTEL) was to make up new stocks as needed rather than have bulk stocks that will be kept for prolonged periods. A repeated test is anticipated in the near future with newly prepared nutrient stocks to determine if this improves growth rates.

Table 1 Summary of uranium reference toxicity test results for 2008–09

Species & endpoint	Test Code	IC_{50} (µg/L)	Valid test?	Comments
Chlorella sp	966G		No	Control growth rate below criterion[b]
(72-h cell division rate)	993G	43 (33, 50)	Yes	
	1021G		No	Control growth rate below criterion[a]
Moinodaphnia macleayi	948I		No	No effect at highest conc[a]
(48-h immobilisation)	949I		No	No effect at highest conc[a]
	991I	238 (224, 253)	No	Not included in control chart[a]
	1001I	36 (31, 41)	Yes	
Hydra viridissima	962B	86 (71, 99)	Yes	
(96-h population growth)	963B	54 (18, 78)	Yes	
	969B	46 (31, 81)	Yes	
	989B	69 (47, 105)	Yes	
	1015B	53 (36, 82)[a]	Yes	
Mogurnda mogurnda	950E	1191 (974, 1456)	Yes	
(96-h sac fry survival)	984E	1685 (1258, 2256)	Yes	
	1006E	1231 (11156, 1310)	Yes	

[a] Values in parentheses represent 95% confidence limits

[b] See text for discussion

H. viridissima

All five reference toxicity tests for *H. viridissima* were valid. There are no issues associated with this protocol. The running mean IC50 is 87 µg/L U with the results from all but one of the tests lying within the upper and lower warning limits (± 2 standard deviations) of 145 and 56 µg/L U, respectively. The IC50 for test 1015B was slightly under the lower warning limit. This may have been due to sub-optimal culture stock health at the time, possibly due to stress related to acclimation to new diluent waters, which has, on occasion, affected Hydra health.

M. macleayi

Of the four reference toxicity tests for *M. macleayi*, only one was valid. The first two tests (948I and 949I) were invalid due to 100% survival of *M. macleayi* exposed to the highest concentration tested of ~80 µg/L U. The third test (991I) investigated U concentrations over a broader range (control, 5, 20, 80, 160, 320 µg/L) to determine an effect concentration. At 160 and 320 µg/L U, there was 100% and 0% survival of *M. macleayi*, respectively, resulting in an EC_{50} of 238 µg/L U. These three tests indicated a lower sensitivity of *M. macleayi* to U than previously found. In fact, in previous years there was an indication of a possible trend towards an increase in sensitivity of the laboratory culture (as reported in Cheng et al 2009).

A new $UO_2SO_4.7H_2O$ stock was prepared to determine if problems with this were responsible for the apparently lower sensitivity of *M. macleayi* to U (noting, however, that similar changes in responses were not observed for the other species). The fourth test (1001I), using the new U stock solution, resulted in a more typical response; 90% mortality was observed at 70 µg/L U and 100% mortality at 139 and 287 µg/L U, and the EC_{50} was 36 µg/L U. This species will again be closely monitored during 2009–10 to determine if there has been a change in species sensitivity.

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.

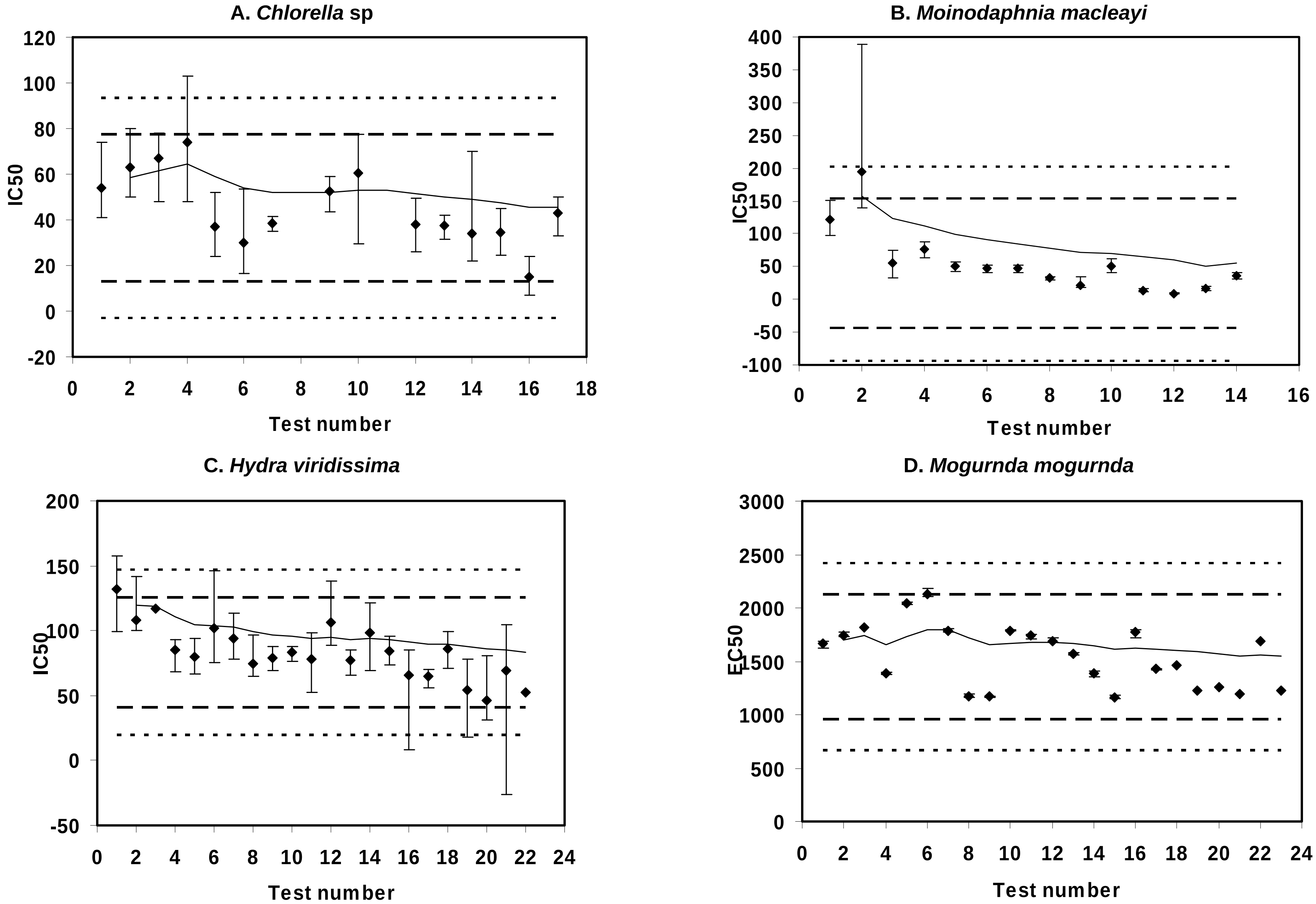


Figure 1 Reference toxicant control charts for A. *Chlorella* sp, B. *M. macleayi*, C. *H. viridissima* and D. *M. mogurnda*, as of September 2008.
Data points represent IC_{50} or EC_{50} toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: dotted lines – upper and lower 99% confidence limits of the whole data set; broken lines – upper and lower warning limits (± 2 standard deviations); unbroken line – running mean.

Reference toxicity test development for *L. aequinoctialis*

Test development for *L. aequinoctialis* is on-going. Previous growth trials using 2.5% CAAC plant growth medium (the medium used to culture this species; 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 are required to elicit a toxic response. At the time of completion of this summary, additional testing was underway to assess the suitability of using this concentration of CAAC in the reference toxicity test method.

Planned testing in 2009–10

The reference toxicity testing programs for *Chlorella* sp, *M. macleayi*, *H. viridissima* and *M. mogurnda* will continue in 2009–10, with the aim of completing at least four tests per species. In addition, further reference toxicity testing of *L.aequinoctialis* will take place to develop the control chart and to trouble-shoot the last remaining issues for this protocol. In addition to the reference toxicity test protocol, work will continue on the development of an additional endpoint, based on plant surface area or dry weight. The development of this endpoint will be done in conjunction with the reference toxicity test work.

References

Cheng K, Costello, C, van Dam R, Hogan A, Harford A & Houston M 2009. Development of a toxicity testing program for routine toxicity test species. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 25–28.

Riethmuller N, Camilleri C, Franklin N, Hogan AC, King A, Koch A, Markich SJ, Turley C & van Dam R 2003. *Ecotoxicological testing protocols for Australian tropical freshwater ecosystems*. Supervising Scientist Report 173, Supervising Scientist, Darwin NT.

van Dam R 2004. *A review of the **eriss** Ecotoxicology Program*. Supervising Scientist Report 182, Supervising Scientist, Darwin NT.

Effects of magnesium pulse exposures on aquatic organisms

A Hogan, R van Dam, A Harford, K Cheng & K Turner

Background

Acquisition of continuous water quality monitoring data in Magela Creek downstream of Ranger since the 2005–06 wet season has enabled quantification of the magnitude, duration and frequency of transient magnesium (Mg) concentrations resulting from mine water discharges (Section 3.1, Supervising Scientist annual report 2007–2008 & 2008–2009). The mine discharge signal is tracked using Electrical Conductivity (EC) as a surrogate for Mg concentration. This is possible since a strong relationship between EC and Mg concentration has been established in grab samples collected over many years for water quality analysis (see 'Surface water transport of mine-related solutes in the Magela Creek catchment using continuous monitoring techniques' pp 58–65 in this volume for full details).

The monitoring data show 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 exceedences were having adverse effects on aquatic biota, and an assessment of the toxicity of Mg under a pulse exposure regime was initiated.

At the commencement of this study, the provisional site-specific Trigger Value (TV) for Mg in Magela Creek was 4.6 mg/L Mg (van Dam et al 2008). Over the four wet seasons that Mg has been continuously monitored, the maximum duration over which this value was exceeded (ie the 'worst case' duration) was only 4 h, which, therefore, represented the duration of greatest interest. Following a subsequent revision of the provisional TV to 2.5 mg/L Mg (van Dam et al 2010)[9], the maximum exceedence duration above this lower value was 137 h (almost six days), although the vast majority of exceedences (51 out of 53), occurred for 24 h or less. Consequently, pulse exposure durations of up to 24 h are now also considered of relevance. As such, this summary paper describes the results of toxicity tests that assessed the effects of both 4 and 24 h Mg pulses on local aquatic species.

Methods

To date, an assessment of the effects of a single Mg pulse of 4 hours duration to five local species (duckweed, hydra, cladoceran, gastropod and fish) and a single 24 h pulse to three local species (duckweed, hydra and fish) has been undertaken. Test species were exposed to the Mg pulse over a range of Mg concentrations except for the fish, which, due to its relative

---

9 The trigger value was revised after it was agreed to use 10% inhibition concentrations (IC_{10}s) instead of 15% inhibition concentrations (IC_{15}s) from the individual toxicity tests for the trigger value derivation. The use of IC_{10}s was more aligned with current practices elsewhere.

insensitivity, was only exposed to a 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).

In addition, because the cladoceran test protocol involves tracking individuals from newly hatched neonate to reproducing adult, it was possible to investigate the influence of the effect of pulse timing with respect to test organism developmental stage. Consequently, an additional test was conducted for this species where the four hour pulse was administered around the time of the onset of reproductive maturity, when the juvenile cladocerans were 27 h old and developing their first brood offspring (approximately 24 h into the experiment).

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

4 h Mg pulses

Toxicity test data for each species undergoing a 4 h Mg pulse are presented in Table 1 and Figure 1. For all five species, the toxicity of a single 4 h Mg pulse at test commencement was consistently lower than when the organisms were continuously exposed to Mg. The relative toxicity across tests was determined by comparing the concentrations that caused a 50% inhibition of the test endpoint (IC50s; based on hydra or lemna growth rate and cladoceran or gastropod reproduction). Where point estimates could not be calculated (eg for the fish and the gastropod 4 h pulse test) a comparison of the different concentration responses is described.

Magnesium was approximately half as toxic to the green hydra, *Hydra viridissima*, under the 4 h pulse regime compared with the continuous exposure. No effects were observed when either the duckweed, *Lemna aequinoctialis*, or the fish, *Mogurnda mogurnda,* were exposed to a pulse of up to 4 g/L Mg. Whilst the concentration response observed for the gastropod, *Amerianna cumingi,* is questionable (with the test needing to be repeated), the snails were producing high egg numbers, similar to those of the controls, at very high concentrations (2 g/L Mg) compared to the continuous exposure IC50 for this species of 100 mg/L Mg. A 4 h Mg pulse at test commencement was approximately an order of magnitude less toxic to the cladoceran, *M. macleayi*, than the continuous exposure. However, when *M. macleayi* was exposed to the 4 h pulse at the onset of reproductive maturity, Mg was only approximately two times less toxic than the continuous exposure, indicating that the timing of the pulse is a key factor for this species.

24 h Mg pulses

The severity of response by both *L. aequinoctialis* and *H. viridissima* to a 24 h pulse was more extreme than that observed after a 4 h pulse, but still less than that following continuous exposure (Table 1 & Figure 1). Some variability was observed between the two tests undertaken for each species and, thus, repeat tests will be required to provide more confident confirmation of the initial findings. The IC50 values for the two 24 h pulse exposures of *L. aequinoctialis* were 2567 and 3144 mg/L Mg, compared to a continuous exposure IC50 of 1393 mg/L. For *H. viridissima,* the response in one 24 h pulse experiment was more aligned with a continuous exposure (continuous IC50 = 663 and 24 h pulse IC50 = 789 mg/L Mg), while the other test gave a less sensitive result, similar to a 4 h pulse (4 h pulse IC50 ~1300 and 24 h pulse IC50 = 1084 mg/L Mg). *M. mogurnda* was unaffected by a 24 h pulse concentration up to 4.1 g/L Mg. This was not unexpected given the low sensitivity of this species to Mg even under a continuous exposure regime.

Table 1 Toxicity of pulse exposed Magnesium compared with continuous exposure

Species	Continuous exposure	4 h pulse at test commencement[c]	4 h pulse at onset of maturity (24 h into test)	24 h pulse at test commencement[c]
	IC_{50} (95%CL)[a] mg/L Mg			
Hydra viridissima (green hydra)	663 (518–746)	1231 (1160–1252) 1393 (1363–1419)	Not applicable	789 (735-903) 1084 (1056-1100)
Lemna aequinoctialis (duckweed)	1393 (664–3207)	>4220	Not applicable	3144 (2626-3621) 2567 (2114-3379)
Moinodaphnia macleayi (cladoceran)	130 (116–144)	1180 (1070–1321) 1498 (1271–2051) 1430 (1163-1990)	305 (289–338)	To be conducted
Amerianna cumingi (gastropod)	100 (31-243)	Not calculable[d]	Not applicable	To be conducted
	LC_{50} (95%CL)[b] mg/L Mg			
Mogurnda mogurnda (fish)	4054 (4046-4063)	Not calculable[e]	Not applicable	Not calculable[e]

a IC50 = Concentration causing a 50% inhibition of the test endpoint (associated 95% confidence limits).

b LC50 = Concentration resulting in 50% lethality of the test organisms (associated 95% confidence limits).

c The results of multiple tests are reported for some species.

d Not calculable due to interrupted concentration response (Figure 1) and low overall egg numbers.

e Not calculable as only one Mg treatment was tested.

For the species tested thus far using two pulse durations the concentrations of Mg that exhibited toxic effects were much greater than the maximum concentration (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 at the onset of reproductive maturity, the concentration of 208 mg/L Mg that caused a 10% inhibition of the test endpoint (IC10; generally considered an ‘acceptable’ level of effect), was still approximately 13 times higher than the reported maximum Mg concentration.

Conclusions

The experiments completed to date show that pulse exposures of Mg of ≤24 h are generally substantially less toxic than continuous exposures over 3 to 6 days. However, the degree to which this is the case depends on the species and, for at least one species (ie *M. macleayi*), the life stage that is exposed. The Mg concentrations at which (sub-lethal) toxic effects have been observed are well in excess of those measured during pulse events in Magela Creek. However, ultimately, Mg concentrations in Magela Creek will need to be compared to pulse exposure trigger values derived from data for all the tested species, rather than toxicity values for individual species.

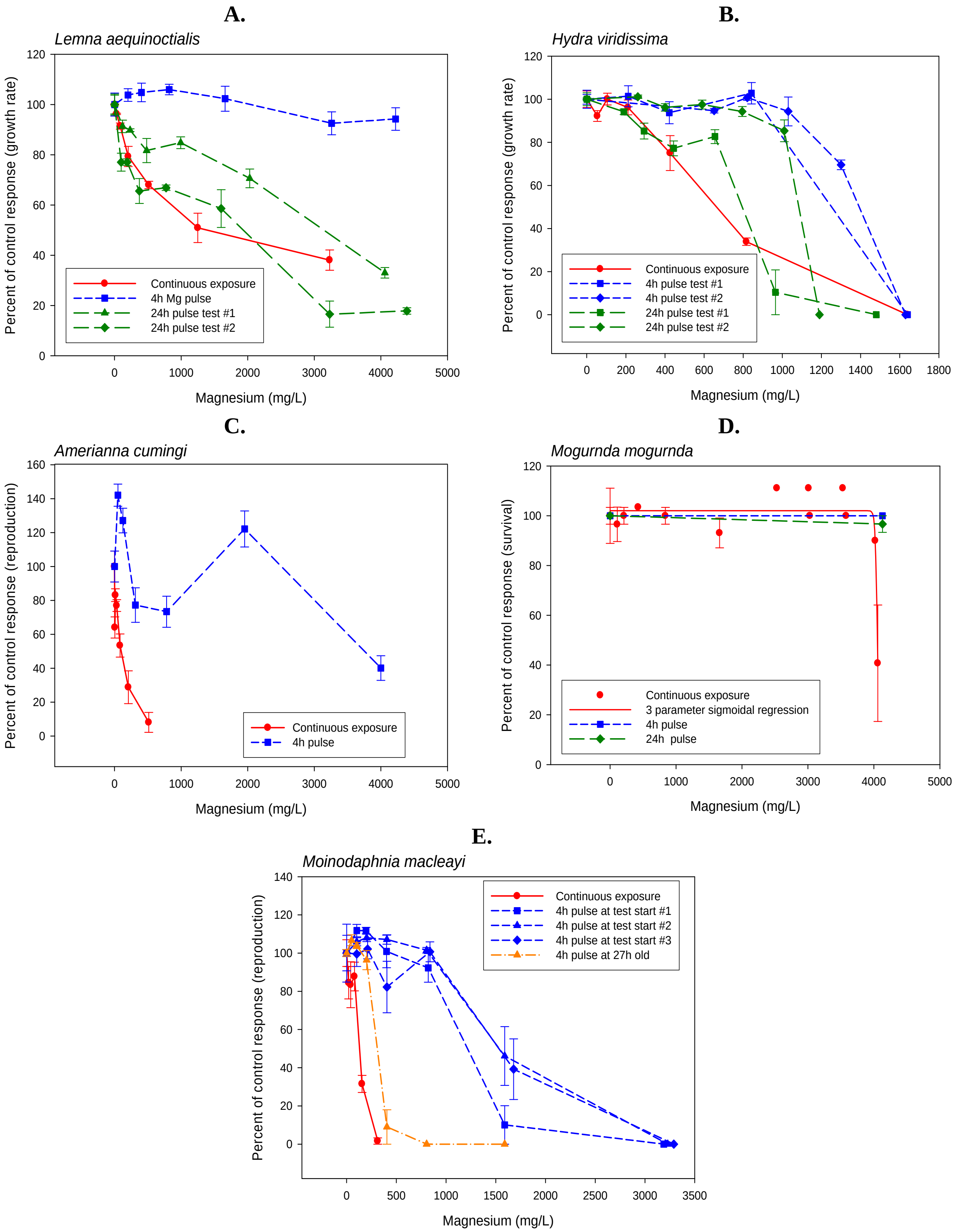


Figure 1 Toxicity of magnesium to the A. duckweed, *Lemna aequinoctialis*, B. green hydra, *Hydra viridissima*, C. gastropod, *Amerianna cumingi*, D. fish, *Mogurnda mogurnda*, and E. cladoceran, *Moinodaphnia macleayi*. Data from continuous exposure experiments are represented by a solid line while 4 h pulse data are represented by short dashed lines and 24 h pulse data by long dashed lines.

Steps for completion

At least two single 4 and 24 h pulse experiments need to be completed for all six species routinely tested at ***eriss***. Some technical challenges are envisaged with respect to pulsing the unicellular alga *Chlorella* sp, and some method development testing may be required for this species. Work will also commence soon on testing an intermediate pulse duration (eg 10 h). Once data are available for all six test species for each pulse duration, TVs will be derived for each exposure duration using the species sensitivity distribution approach.

The following phase of the research will involve testing multiple pulses given that exceedences of the 2.5 mg/L TV are so numerous that pulse frequency and recovery period between pulses are also important.

Further research into toxicokinetic modelling may also be warranted considering the expansion of the current testing program with the revision of the TV from 4.6 to 2.5 mg/L. Pulsing scenarios in the creek are likely to be much more complex and variable when compared to a benchmark of 2.5 mg/L Mg, and these scenarios may not always be matched to those tested in the laboratory. Therefore 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).

References

Ashauer R, Boxhall ABA & Brown CD 2006. Predicting effects on aquatic organisms from fluctuating or pulsed exposure to pesticides. *Environmental Toxicology and Chemistry* 25(7), 1899–1912.

Diamond JM, Klaine SJ & Butcher JB 2006. Implications of pulsed chemical exposures for aquatic life criteria and wastewater permit limits. *Environmental Science and Technology* 40, 5132–5138.

van Dam R, Hogan A, McCullough C & Humphrey C 2008. Toxicity of magnesium sulfate in Magela Creek water to tropical freshwater species. In ***eriss*** *research summary 2006–2007*, eds Jones DR, Humphrey C, van Dam R & Webb A, Supervising Scientist Report 196, Supervising Scientist, Darwin NT, 11–14.

van Dam RA, Hogan AC, McCullough C, Houston M, Humphrey CJ & Harford AJ 2010. Aquatic toxicity of magnesium sulphate, and the influence of calcium, in very low ionic concentration water. *Environmental Toxicology and Chemistry* 29(2), 410–421.

The effects of suspended sediment on tropical freshwater biota

A Harford, M Saynor, R van Dam, A Hogan & D White

Background

The issue of suspended particulate matter (SPM) as an aquatic ecosystem stressor in Magela Creek will likely assume greater significance during the decommissioning and initial rehabilitation phases of the Ranger minesite. However, there may also be an increased risk of suspended sediment in runoff from the site as a result of works associated with the proposed heap leach operation and construction of a second tailings containment facility. These new works will be located in the catchment of Gulungul Creek.

Apart from an investigation conducted by ***eriss*** on the effects of suspended sediment in a creek downstream of the Jim Jim Falls road crossing (Stowar 1997), there has been no systematic program of research to characterise the potential biological impacts of increased SPM on aquatic ecosystems in the ARR. A robust assessment of the effects of SPM is needed to provide the basis for developing operational water management triggers and closure criteria. Nationally, the issue of SPM as an ecosystem stressor has been ranked as the third highest priority for waterway management following salinity and eutrophication (Land and Water Australia 2007). In contrast to the large volume of research that has been conducted during the last four decades on dissolved contaminants, there has been little advance in water quality guidelines for SPM. This situation probably reflects the relative complexity of experimental design and degree of difficulty required to unambiguously establish the effects of suspended sediment (a heterogeneous system) compared with the protocols required to establish the effects versus concentration response for dissolved toxicants (a homogeneous aqueous system).

In reviewing the literature on the biological effects of SPM, Berry et al (2003) noted the need for more comprehensive experimental concentration-response data to better understand the responses of aquatic species within and between habitat types. A key principle of such an approach, and one that applies also to the assessment of dissolved toxicant impacts, is that both the concentration and duration of suspended sediment exposure are important determinants of the level of impact, with the frequency of exposure also being an important variable. Unfortunately, the majority of past SPM toxicity studies have reported nominal concentrations (ie not measured concentration but inferred from the mass of sediment added to test solutions) of sediment, which does not allow for an assessment of the accuracy of SPM concentrations in the tests. Additionally, there is little discussion in the literature concerning the methods of exposure of suspended sediment to the organisms (eg mixing to keep in suspension or allowing particles to settle) and the way the method of presentation affects dose to the organism.

Bilotta and Brazier (2008) noted that particle size distribution and particle chemistry are both important determinants of effects of suspended sediment. Likewise, recent developments in toxicology of particulates (specifically nanoparticles) have led to the conclusion that the physicochemical characterisation of particles is fundamental to determining appropriate ‘dose’-response models. The numerous particle characteristics that may drive the behaviour of particles in the environment and the response of an organism include particle size, particle shape, mineralogy and composition, surface area and charge, porosity and surface chemistry.

The objective of this project is to assess the effects of a SPM 'reference material' comprising the <63 μm fraction of the lateritic material that is intended to be used as a component of the surface cover layer on the rehabilitated Ranger landform. 'Lateritic material' refers to sediment (generally iron-rich) that comes from the weathered horizon of the regolith. It usually contains a high percentage of smaller particles including of clay minerals. This weathered horizon can extend to 40 m in depth below the ground surface.

It is intended to mix approximately 30% by volume of the laterite material with waste rock in the top 2 to 5 m of the rehabilitated landform. The reason that the laterite is being incorporated in the cover layer is that it contains a substantial proportion of fine-grained material that will improve the moisture-retaining characteristics of the surface layer of the landform and by inference provide a superior growth substrate. The consequences of the use of laterite will be the presence of a much more erodible component in the surface of the rehabilitated landform. Hence, there is a higher probability of fine-grained material being delivered to Magela Creek during the decommissioning and rehabilitation phases of the Ranger minesite.

In order to better understand cause-effect relationships observed for test organisms, the project will include detailed physico-chemical characterisation of the SPM 'reference material'. Concentrations (eg particles/mL) and particle size distribution of the SPM will also be measured in the toxicity test media containing the particles for the duration of the exposures. This will provide a more accurate assessment of SPM 'concentrations' and greater confidence in the concentration-response models. This project is a collaboration between the ***eriss*** Ecotoxicology and Hydrological & Geomorphic Processes (HGP) groups.

Methods

Development of <63 μm sediment separation method

A bulk sample of lateritic (weathered horizon) material was collected on 8 September 2008 from Pit 3 at the Ranger mine. Possible methods for extracting and preparing the <63 μm fraction were assessed during late 2008 to early 2009. The methods investigated were dry and wet sieving for extraction, and oven and freeze drying for preparation. Particle size distributions and particle morphology were investigated using electrozone sensing (ie Coulter counting) and microscopy techniques.

Range-finding toxicity tests

An initial <63 μm fraction that was isolated from a sub-sample of the Ranger Pit 3 laterite was used in three 'range-finding' toxicity tests. This enabled estimation of the mass of material required to conduct definitive toxicity tests and also an evaluation of the appropriateness of the standard algal (*Chlorella* sp) toxicity test Riethmuller et al (2003). *Lemna aequinoctialis* (duckweed) and *Mogurnda mogurnda* (purple-spotted gudgeon) were also used for the range-finding tests as they were suspected to be the least sensitive (based on responses to Mg and U) of the suite of species used at ***eriss***. Using a less sensitive species would yield a conservative estimate of the total mass of SPM likely to be required for future test work.

Production of the SPM Standard Reference Material (SRM)

The SPM-SRM is currently being produced from a 100 kg bulk sample of material, collected from the trial rehabilitation landform, using a process of wet-sieving, followed by a dehydration step of centrifugation and freeze drying. Although lateritic material was originally identified by Energy Resources of Australia Ltd as the preferred material for the trial landform cover, a transitional material was also used. Transitional material refers to the material that is located between the surficial lateritic material and the deeper rocks containing the uranium ore.

There was difficulty sourcing sufficient lateritic material for the construction of the trial landform and it was supplemented with the use of transitional material. Discussions on site during construction suggested that there would be a similar problem with the supply of lateritic material for the final rehabilitation and that large amounts of the transitional material could be used. Consequently, it was decided to use the transitional material for the initial toxicity tests.

Results and discussion

Development of <63 μm sediment separation method

The average mass of fines obtained from the wet sieving process was approximately 10% of the total initial weight of the bulk sample, which was an order of magnitude greater than that produced by the dry sieving process. Additionally, microscopy demonstrated that a larger proportion of fine particles were present in the wet-sieved samples (Figure 1). Thus, wet-sieving was deemed a superior method to dry sieving, especially considering that the fine particles are of most interest because they will remain in suspension for longer periods and are likely to be more biologically active than larger particles.

However, the main disadvantage with the wet-sieving method was the large volume of water required in the process. This water needed to be removed without significantly modifying the particles' physico-chemical properties. Consequently, centrifugation followed by freeze drying was identified as the most appropriate method. Electrozone analysis (using a Coulter Counter) of the wet-sieved samples showed that the majority of particles were <2 µm (spherical volume) (Figure 2a) in size, which is typically below the resolution of what this method of particle size analysis (PSA) can measure. Microscopy of the samples showed that some of the particles were flat and transparent mica particles (Figure 2b), which would present additional challenges for particle size analysis using methods that report particle size results on the basis of equivalent spherical volume. Subsequently, other methods (eg laser diffraction, sedigraphy and others) and commercial suppliers offering particles size analysis services are currently being investigated to further characterise the SRM.

Range-finding toxicity tests

The range-finding studies showed that nominal concentrations of 0.5 and 1 g/L of SPM resulted in significant reductions in survival of *M. mogurnda* and growth of *L. aequinoctialis*, respectively (Figure 3). From these results, it was estimated that 270 g of SPM-SRM would required as a minimum amount to accommodate the initially planned suite of toxicity tests, ie 4 tests for each of the 5 species, including experiments to investigate the effects of different methods of keeping particles in suspension. However, larger amounts would be required for characterisation of the material and any further toxicity tests (eg comparison with field samples and spiking the sediment with uranium). Thus, 1 kg was considered a suitable target.

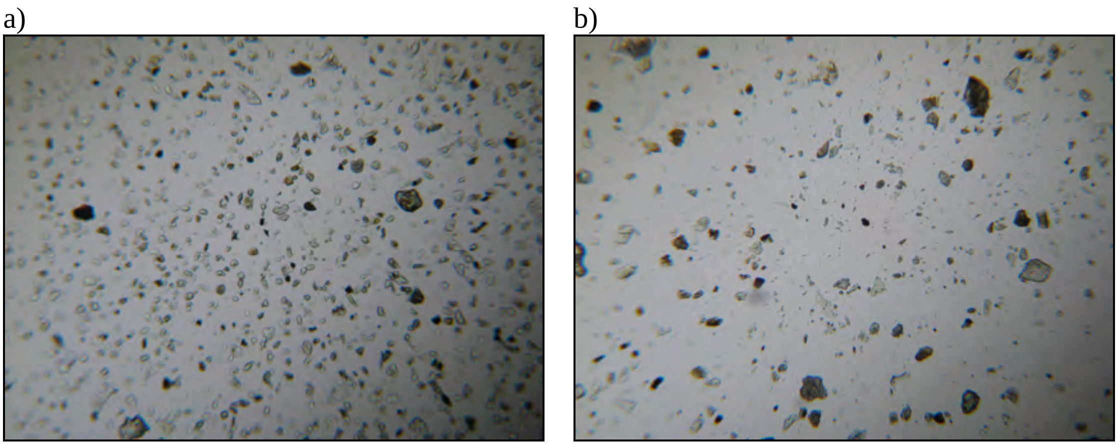


Figure 1 Images from a compound microscope (x100) showing a) a wet-sieved sample and b) a dry sieved sample. Both samples were the same mass per volume concentration.

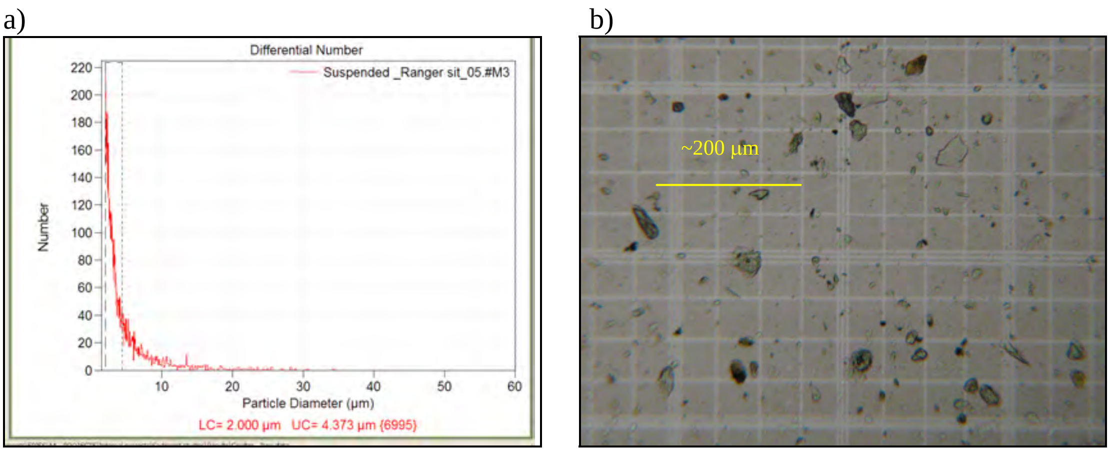


Figure 2 Laterite sample from Ranger Pit 3. Wet-sieved and diluted 1:1200 (unknown concentration). a) Particle size analysis from the Coulter counter showing the majority of particles are <2 µm b) an image of the same sample from a compound microscope.

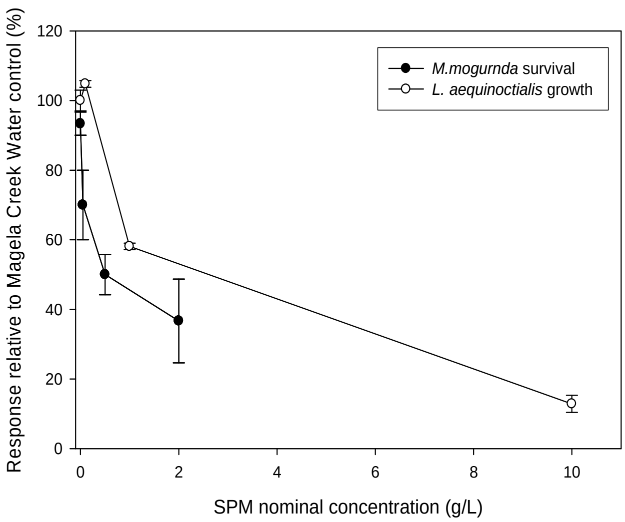


Figure 3 Effect of SPM isolated from Pit 3 on the survival of *M. mogurnda* and the growth of *L. aequinoctialis*. Data points represent the mean ± standard error of three replicates.

The initial toxicity test with *Chlorella* sp identified that electronic particle counting and light microscopy were unable to distinguish algal cells in the presence of sediment particles. Subsequently, a mercury vapour lamp (Leitz) was procured to upgrade the laboratory's microscope capability to distinguish living (auto-fluorescent) and non-living (inorganic) particles.

Production of the SPM Standard Reference Material (SRM)

The transitional waste rock material collected from the trial landform had a 10% yield of fine particles (ie <63 µm fraction). This composition of fine material was comparable with the 'laterite' material collected from Pit 3, although improvements in the isolation process may have increased the yields. Currently, 600 g of this material has been produced with an aim of producing a total of 1 kg prior to the start of the comprehensive program of toxicity testing.

Steps for completion

The current focus of this project is the production of SPM-SRM, which is proceeding steadily. Following collection of a suitable amount of this material, the physico-chemistry of the SPM-SRM will be characterised in detail. However, this material presents a number of significant challenges due to its heterogeneity and unique optical properties. Thus different methods need to be compared and measurement results validated. Consequently, a significant collaboration with the National Measurement Institute (NMI) is in the process of being developed. Potential studies will include physico-chemical characterisation of the SPM-SRM as a powder, and in Magela Creek Water and Synthetic Soft Water over the test durations, measurement of test solution concentrations, investigations into the effect of dissolved organic carbon on particle agglomeration and comparisons with field samples of interest (eg trial landform runoff).

Laboratory-based concentration-response testing for the ***eriss*** suite of six species will commence in late 2009. This may involve developing and employing novel test designs, eg fluorescence microscopy will be used for the algal assay. Further concentration-response experiments will investigate the effects of different methods (eg the use of orbital shakers) for maintaining particles in suspension and will also incorporate different exposure durations (and possibly frequencies). In addition, samples of runoff obtained from the trial landform project will be used to compare the toxicity and physico-chemistry of sediments being washed from the mine site to that of the SPM-SRM produced in the laboratory.

References

Berry W, Rubinstein N & Melzian B 2003. *The biological effects of suspended and bedded sediment (SABS) in aquatic systems: A review*. Internal Report National Health and Environmental Effects Laboratory, Office of Research and Development, United States Environmental Protection Agency, Narragansett, RI, USA.

Bilotta GS & Brazier RE 2008. Understanding the influence of suspended solids on water quality and aquatic biota. *Water Research* 42, 2849–2861.

Land and Water Australia 2007. Salt, nutrient, sediment and interactions: Findings from the national river contaminants program, http://lwa.gov.au/files/products/river-landscapes/pk071328/pk071328.pdf. Accessed: 9 September 2009

Riethmuller N, Camilleri C, Franklin N, Hogan AC, King A, Koch A, Markich SJ, Turley C & van Dam R 2003. *Ecotoxicological testing protocols for Australian tropical freshwater ecosystems*. Supervising Scientist Report 173, Supervising Scientist, Darwin NT.

Stowar M 1997. Effects of suspended solids on benthic macroinvertebrate fauna downstream of a road crossing, Jim Jim Creek, Kakadu National Park. Internal report 256, Supervising Scientist, Canberra. Unpublished paper.

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

R van Dam, C Humphrey, A Harford, S Simpson[1], K Gibb[2] & J Stauber[1]

Background

Research and monitoring of the aquatic impacts of the Ranger Uranium Mine has historically focused on water quality analysis, ecotoxicity testing and in situ monitoring of biota during mine water release, and longer-term biological monitoring of in stream macroinvertebrate and fish communities. This focus on the water column has understandably been a consequence of the fact that water is the primary transport vector for solutes released from the minesite. However, since solutes such as uranium (U) have a high affinity for sediments, sediment quality assessment and derivation of protection trigger values for sediments are aspects of aquatic ecosystem protection that also need to be considered. Such trigger values will have application both for operational water management as well as the development of sediment quality closure criteria for the site.

Internationally there has been little work done on the toxicity of U in sediments to aquatic biota, and the toxicity estimates produced by the few studies that have been published have varied by at least three orders of magnitude. This lack of sediment quality toxicity data is of some concern, not only for the local situation, but also given the projected expansion of the uranium mining industry in Australia. 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, *Chironomus crassiforceps*, was not affected by sediment U concentrations up to 5000 mg/kg dry weight. The RP1 constructed treatment wetland was the context for this work. However, two recent overseas studies reported significant (~10%) effects of U in sediment at 3 mg/kg dry weight, for the chironomid, *Chironomus riparius* (Dias et al 2008), and 600 mg/kg dry weight, for the worm, *Tubifex tubifex* (Lagauzère et al 2009), with LC50 values of 5.3 mg/kg dry weight and 2900 mg/kg dry weight, respectively.

On the Ranger lease, recent (2007) sediment U concentrations in mine-influenced waterbodies, such as Georgetown Billabong (< 45 mg/kg), are much higher than reference waterbodies (1.2–4.3 mg/kg; eg. Sandy and Buba Billabongs). While U concentrations in the sediments of Georgetown have been systematically higher than those of other natural billabongs of the region since before the start of mining (Noller & Hart 1993), the recent sediment concentrations appear to represent an increase since about 2002 (Humphrey et al 2009). It is unknown whether the concentrations in Georgetown Billabong represent a risk to sediment dwelling biota, but the work by Dias et al (2008) and Lagauzère et al (2009) demonstrates that an investigation is warranted.

In relation to the operational phase of mining, benthic macroinvertebrate communities in the mine-influenced, off-site waterbodies, Georgetown and Coonjimba Billabongs, currently exhibit lower diversity than reference billabongs. Good quality sediment U toxicity data are required to help determine whether the observed impairment is due to U in sediments or to

[1] CSIRO Centre for Environmental Contaminants Research, Lucas Heights, NSW
[2] School of Science & Primary Industries, Charles Darwin University, NT 0909

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 mine site, as well as for downstream receptor wetlands.

To address these knowledge needs, a field sediment U toxicity experiment will be undertaken, in collaboration with the CSIRO Centre for Environmental Contaminants Research and Charles Darwin University. A field experiment has several benefits over a laboratory assessment, the key ones being that a field experiment will be more time and cost effective than a laboratory approach (not requiring selection and culturing of suitable local test species, nor development of test protocols) and will be able to assess responses of whole communities of organisms.

Methods

The proposed approach will involve the deployment of U-spiked sediments (in retrievable containers) in (unimpacted) Gulungul Billabong, over the duration of a wet season. At the end of the exposure period, the extent of colonisation of macroinvertebrate, microinvertebrate and microbial communities will be measured in the control and test replicates.

The project plan comprises initial characterisation of sediment biota in the target water body (composition and densities) at the end of the 2008–09 wet season, a pilot range-finding exposure study to be conducted during the 2009–10 wet season and the main experiment to be conducted during the 2010–11 wet season. The initial characterisation study will inform the design of the pilot study which in turn will be used to evaluate/trouble-shoot the proposed approach and exposure design to be used for the main experiment.

Site characterisation

Site biological characterisation was undertaken following a sampling trip to Gulungul Billabong in April 2009. After the study site was identified, 18 sediment samples (20 × 20 × 5–10 cm) were collected along a transect of approximately 15 m at a water depth of 55–60 cm. An additional smaller sample (100 mL), for microbial community analysis (see below), was collected immediately adjacent to the larger sample. The larger samples were wet-sieved through a series of mesh screens (8 mm, 500 µm, 125 µm and 63 µm). The >8 mm fraction was discarded (after ensuring no macro-fauna was present), and the remaining three sediment fractions were collected and preserved in 90% ethanol for microinvertebrate (by Dr Russ Shiel, University of Adelaide, 125 µm and 63 µm fractions) and macroinvertebrate (by ***eriss***, 500 µm fraction) sorting, identification and data analysis. Four additional sediment samples (~50 mL) were collected along the transect for physico-chemical analysis.

Microbial analysis of the sediments involved total DNA extraction from each sediment sample. The DNA was further analysed by terminal restriction fragment length polymorphism (TRFLP) analysis in which DNA fingerprints are generated that theoretically represent each bacterial species present in the sediment. TRFLP is a low cost snapshot of diversity, and one of the reasons for incorporating it into this study was to determine if the method actually worked on billabong sediment (ie the method is often used to assess the microbiology of arable soils, so this is a relatively new application). To obtain a thorough inventory of bacterial diversity for further comparative studies and to assess the validity of the TRFLP approach, the sediment DNA was also subjected to exhaustive direct sequencing using a new technique called 454 pyrosequencing. CDU has completed the experimental work and is currently awaiting data analysis from the Australian Genome Research Facility (AGRF). Only the TRFLP results are reported here.

Pilot field study

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, pers comm), followed by 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 (latter two treatments 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 (24–35ºC) for 10 days to dry to a point where they could be transferred to the experimental containers. Sub-samples for sediment and pore water 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.

Progress/Results

Site characterisation

Sediment physico-chemistry

The total organic carbon content of the sediment was 5%. Concentrations of key metals/metalloids in whole sediment were as follows (mean ± standard deviation; n = 4): uranium – 6.2 ± 0.5 mg/kg; aluminium – 48 000 ± 4500 mg/kg; arsenic – 2.4 ± 0.3 mg/kg; copper – 32 ± 3.9 mg/kg; iron – 12 000 ± 940 mg/kg; lead – 12 ± 0.9 mg/kg; manganese – 62 ± 5.2 mg/kg; nickel – 18 ± 1.7 mg/kg; and zinc – 13 ± 1.5 mg/kg. Further physico-chemical data were unavailable at the time of completion of this summary.

Microbes

There were significant differences in microbial community level diversity (based on the DNA profiles) amongst the 18 sampling sites (ANOSIM, $p = 0.001$). When ANOSIM was run using transect distance instead of sample site as the factor, communities from the same general location tended to be more closely related ($p = 0.004$). This suggests that species composition was not homogeneous along the transect but exhibited some spatial variation. This is not particularly surprising as communities of microbiota would be expected to exhibit some spatial variation along a transect of reasonable length. This issue can be addressed by taking additional replicates at each site and/or for each sample. Moreover, experimental units in the pilot and main study will be placed along the transect in a randomly-stratified manner, to account for any systematic natural variability.

The DNA community profiling method can also be used to identify species or ‘operational taxonomic units’ by comparing DNA profiles against an international database of bacterial DNA profiles. The success of this depends on achieving matches to the database, which contains over 30 000 entries, many of which are from temperate regions. Species matches for the sample sites are shown in Table 1. They reveal plausible matches and provide insights into key biogeochemical processes in this system. Unfortunately, the database contains lists of uncultured bacteria with no additional information so these have been grouped together to give an indication of their proportional representation in the community.

The dominant genus was *Burkholderia*, which occupies remarkably diverse ecological niches, ranging from contaminated soils and water to the respiratory tract of humans (Coenye & Vandamme 2003). The most frequently occurring species was the nitrogen fixing *B. tropica*, which is associated with plant roots, and has a climatic range from temperate sub-humid to hot humid (Reis et al 2004). This genus includes the causative agent of Melioidosis *B. pseudomallei* (not found in the current study), which is commonly found in soil and water in this region (Kaestli et al 2007). *Clostridium sulfidogenes* is a recently described mesophilic bacterium isolated from pond sediment, which can degrade peptides and reduce thiosulfate and sulfur (Sallam & Steinbüchel 2009). *C. sulfidogenes* may be involved in sulfur cycling in this system. The lactic acid bacterium, *Lactobacillus ingluviei*, is a member of a genus that is commonly found associated with bird faeces and, although the literature only mentions it in association with pigeons (Nazef et al 2008), it is likely to be associated with a wide range of native bird species. The genus *Pelosinus* was only described in 2007 (Shelobolina et al 2007) and one species that occurs regularly in studies is a fermentative organism that can reduce Fe(III) in the presence of fermentative substrates such as those anions typically found in organic matter (eg formate, acetate, lactate, etc.). Members of this genus may be important to iron geochemistry in this system. The Phylum *Acidobacteria* occurred at most sites, but was not as diverse as *Burkholderia.* This Phylum could be important as the project progresses because its members are ubiquitous, diverse, and abundant in soils and sediments, and can withstand metal-contaminated, acidic, and other extreme environments, including sediments contaminated with U and other potentially toxic materials (Barns et al 2007). Little is known about members of the novel Phyla *Verrucomicrobia* or *Planctomycetes* other than they are proving to be widespread and active in the environment (Joseph et al 2003). Members have been reported from Australia and include thermoacidophilic (acid and heat tolerant) members, which use methane as their sole carbon source (methanotrophs). So far all members appear to play an important role in biogeochemical carbon cycles (Islam et al 2009).

In summary, the TRFLP worked well in that interesting and plausible species were identified, many of which are fastidious anaerobes that would probably have been overlooked in conventional cultivation techniques. The real test for this method will be whether the diversity list has any similarity to the list derived from the exhaustive sequencing exercise. The interest in using TRFLP in the longer term is that it is a very simple and inexpensive laboratory technique requiring little molecular expertise, and lending itself to high throughput, environmental-scale sample analysis.

Table 1 Community summary TRFLP information for microbes occurring in natural Gulungul Billabong sediment samples, April 2009

Species match	# OTUs[1]	Sample																													
		1a	**2a**	**3a**	**3b**	**4a**	**4b**	**5a**	**5b**	**6a**	**6b**	**7a**	**7b**	**8b**	**9a**	**9b**	**11a**	**11b**	**12a**	**12b**	**13a**	**14a**	**14b**	**15a**	**15b**	**16a**	**16b**	**17a**	**17b**	**18a**	**18b**
Burkholderia sp	51	X	X	X		X	X	X	X	X	X	X	X			X	X	X			X	X	X		X	X	X				
Clostridium sulfidogenes	1																											X		X	
Lactobacillus ingluviei	1																														X
Pelosinus sp	1																	X													
Spiroplasma sp	1										X																				
Thermus yunnanensis	1										X																				
uncultured Acidobacteria bacterium	1	X	X	X		X	X	X		X	X	X			X	X		X		X	X	X	X			X					
uncultured *Planctomyces* sp	1																													X	
uncultured *Pseudomonas* sp	1	X	X	X		X	X	X	X	X	X	X	X				X	X			X	X	X		X	X	X				
uncultured soil bacterium	2						X	X						X		X					X	X	X		X		X				
uncultured Verrucomicrobia bacterium	1										X	X											X								
uncultured Xiphinematobacteriaceae	1					X															X										
uncultured bacterium misc	70	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		X	X	X
Total no. OTUs present per site		49	48	44	1	70	52	57	42	51	67	61	40	3	3	6	44	46	1	8	62	52	80	1	51	65	66	1	15	21	20

[1] OTU: Operational taxonomic unit – representative of a genetic variant, and could represent different species and different strains of the same species.

Micro- and macroinvertebrates

Processing of the 63 and 125 μm fractions for microinvertebrates by Dr Russell Shiel was limited mainly to development of a suitable protocol for sub-sampling and extracting organisms, based upon just two or three of the 18 samples. The fine-grained (mud) sediments characteristic of backflow billabongs (such as Gulungul) present particular difficulties for sample processing, requiring meticulous separation of (often) cryptic taxa from sediment particles using fine tungsten needles. Hence sample processing is a slow task. Based upon these results and time estimates for sample processing, more substantive funding has been reserved for processing of samples arising from the pilot study to ensure timely completion of microinvertebrate sorting and identifications.

For the samples that were sub-sampled and processed, taxa lists and relative abundances were provided. Table 2 contains community summary information for these samples. (Summary estimates are minima because only a limited number of samples was processed.) Amongst all samples processed, the microinvertebrate fauna was dominated by protists (in particular, Rhizopoda, Difflugiidae – amoeboids inhabiting a test or shell) and rotifers (in particular, the Lecanidae). According to Dr Shiel, most of the fauna is characteristic of low dissolved oxygen environments. The sediment size fraction with greatest species diversity was the >63 μm fraction, with the >125 μm fraction containing a subset of taxa found in the finer fraction (data not provided here).

Table 2 Community summary information for micro- and macroinvertebrates occurring in natural Gulungul Billabong sediment samples, April 2009. (Samples were between 2000-4000 cm^3 volume of sediment.)

Aquatic organism group	Community summary	Sample fraction		
		63–125 μm	125–500 μm	>500 μm
Macroinvertebrates	Total no. taxa recorded (minimum)	–	–	32
	Ave. taxa number per sample (min)	–	–	8.4
	Ave. total abundance per sample	–	–	335
Microinvertebrates	Total no. taxa recorded (min)	34	21	–
	Ave. taxa number per sample (min)	18	12	–
	Ave. total abundance per sample	182 500	17 660	–

For macroinvertebrates contained in the >500 μm fraction, preserved samples were sub-sampled and sorted under a stereo microscope in the laboratory. All 18 samples were processed, with community summary information for these samples shown in Table 2. In the case of macroinvertebrates, summary estimates are minima because the data are summarised at family-level only for the purposes of this report. Taxa numbers and abundances in the sediments were low, reflecting possibly the fine-grained sediment particles (restricting habitat availability) and low dissolved oxygen environment characteristic of billabong waters at the end of the 2008–09 wet season. Total abundance of macroinvertebrates varied between 82 and 685 organisms per sample (mean = 335, SD=183). The faunal assemblage was dominated by worms, mites, midge larvae and microcrustaceans (Table 3). Relative to typical macroinvertebrate datasets, the data from the full 18 samples demonstrated relatively low variation across the transect.

Table 3 Macroinvertebrate taxa occurring in natural Gulungul Billabong sediment samples, April 2009, ranked by decreasing abundance in samples. Only taxa for which mean number of organisms per sample exceeded one are provided. (Samples were between 2000–4000 cm^3 volume of sediment.)

Taxon	Common name	Ave no. per sample
Nematoda	Nematodes (unsegmented worms)	189.2
Oligochaeta	Segmented worms	44.7
Oribatida	Water mites	23.8
Chironomidae	Non-biting midge larvae	15.7
Copepoda	Copepods (microcrustaceans)	9.4
Ostracoda	Ostracods (microcrustaceans)	9.2
Cladocera	Water fleas (microcrustaceans)	8.6
Coleoptera larvae (unident)	Beetle larvae	7.9
Acarina	Water mites	6.1
Ceratopogonidae	Biting midge larvae	3.3
Aranae	Aquatic spiders	3.0
Planorbidae	Freshwater snails	1.9
Chaoboridae	Phantom midge larvae	1.9
Leptoceridae	Caddisfly larvae	1.8
Collembola	Springtails	1.6
Chironomidae	Non-biting midge pupae	1.5
Caenidae	Mayfly larvae	1.1

Pilot study

Sediment physico-chemistry results for the pilot study were unavailable at the time of completion of this summary.

Conclusions

As noted above, the results of the initial site characterisation are being used to inform aspects of the design of the pilot and definitive experiments. They have identified a need to carefully consider the timing of sediment retrieval so as to maximise macroinvertebrate diversity (ie earlier late wet season retrieval when DO levels are higher). The data also serve as a baseline for natural sediments, and have indicated that such fine-grained sediments, DO issues aside, are not optimal for macrobenthic colonisation. This has emphasised the importance of also measuring other components of the sediment biota (microinvertebrates and microbes). Initial microbial analysis based on TRFLP was found to be informative, and identified numerous bacteria of interest. Finally, the data on the variability of the biological communities will be used to optimise the experimental design for the definitive study (eg through power analysis), although the data from the pilot study will be most useful for this purpose.

Steps for completion

- *Pilot study (mid 09 – ~May 10):* Dec 09 – deployment of sediment (in containers) in Gulungul Billabong; April 2010 – retrieval of containers from study site; May–July 2010 – biological and physico-chemical data collection and analysis.

- *Main experiment (mid 10 – ~May 11)* – full scale study to assess the toxicity of sediment-bound U to benthic biota. This will include: June-July 2010 – design of experiment; Aug 2010 – collection of test sediment; Aug-Dec 2010 – spiking and equilibration of test sediment; Dec 2010-May 2011 – deployment of sediment (in containers) in Gulungul Billabong through wet season (possibly with periodic sampling to assess U loss from sediments); May-Aug 2011 – biological and physico-chemical data collection and analysis; and July-Dec 2011 – write up.

References

Barns SM, Cain EC, Sommerville L & Kuske CR 2007. *Acidobacteria* Phylum sequences in uranium-contaminated subsurface sediments greatly expand the known diversity within the Phylum. *Applied and Environmental Microbiology* 73, 3113–3116.

Coenye T & Vandamme P 2003. Diversity and significance of *Burkholderia* species occupying diverse ecological niches. *Environmental Microbiology* 5, 719–729.

Dias V, Vasseur C & Bonzom J-M 2008. Exposure of *Chironomus riparius* larvae to uranium: Effects on survival, development time, growth, and mouthpart deformities. *Chemosphere* 71, 574–581.

Humphrey C, Turner K & Jones D 2009. Developing water quality closure criteria for Ranger billabongs using macroinvertebrate community data. In ***eriss*** *research summary 2007–2008*, eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 130–135.

Islam T, Jensen S, Johanne Reigstad L, Larsen Ø & Birkeland N-K 2009. Methane oxidation at 55°C and pH 2 by a thermoacidophilic bacterium belonging to the *Verrucomicrobia* phylum. *Proceedings of the National Academy of Sciences* 105, 300–304.

Joseph SJ, Hugenholtz P, Sangwan P, Osborne CA & Janssen PH 2003. Laboratory cultivation of widespread and previously uncultured soil bacteria. *Applied and Environmental Microbiology* 69, 7210–7215.

Kaestli M, Mayo M, Harrington G, Watt F, Hill J, Gal d & Currie BJ 2007. Sensitive and specific molecular detection of *Burkholderia pseudomallei*, the causative agent of Melioidosis, in the soil of tropical northern Australia. *Applied and Environmental Microbiology* 73, 6891–6897.

Lagauzère S, Terrail R & Bonzom J-M 2009. Ecotoxicity of uranium to Tubifex tubifex worms (Annelida, Clitellata, Tubificidae) exposed to contaminated sediment. *Ecotoxicology & Environmental Safety* 72, 527–537.

Nazef L, Belguesmia Y, Tani A, Prévost H & Drider D 2008. Identification of lactic acid bacteria from poultry feces: Evidence on anti-*Campylobacter* and anti-*Listeria* activities. *Poultry Science* 87, 329–334.

Noller BN & Hart BT 1993. Uranium in sediments from the Magela Creek catchment, Northern Territory, Australia. *Environmental Technology* 14, 649–656.

Peck MR, Klessa DA & Baird DJ 2002. A tropical sediment toxicity test using the dipteran *Chironomus crassiforceps* to test metal bioavailability with sediment pH change in tropical acid-sulfate sediments. *Environmental Toxicology & Chemistry* 21, 720–728.

Reis VM, Estrada-de los Santos P, Tenorio-Salgado S, Vogel J, Stoffels M, Guyon S, Mavingui P, Baldani VLD, Schmid M, Baldani JI, Balandreau J, Hartmann A &

Caballero-Mellado J 2004. *Burkholderia tropica* sp. nov., a novel nitrogen-fixing, plant-associated bacterium. *International Journal of Systematic and Evolutionary Microbiology* 54, 2155–2162.

Sallam A & Steinbüchel A 2009. *Clostridium sulfidogenes* sp. nov., a new mesophilic, proteolytic bacterium isolated from a pond sediment, able to reduce thiosulfate and sulfur. *International Journal of Systematic and Evolutionary Microbiology* 59, 1661.

Shelobolina ES, Nevin KP, Blakeney-Hayward JD, Johnsen CV, Plaia TW, Krader P, Woodard T, Holmes DE, VanPraagh CG & Derek RL 2007. *Geobacter pickeringii* sp. nov., *Geobacter argillaceus* sp. nov. and *Pelosinus fermentans* gen. nov., sp. nov., isolated from subsurface kaolin lenses. *International Journal of Systematic and Evolutionary Microbiology* 57, 126–135.

Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer, R Cahill, R Thorn, J Pfitzner & A Esparon

Introduction

The inhalation pathway has previously been identified as the main contributor to public dose from the Ranger mine site 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 two airborne 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 (ICRP) is 1 milli Sievert (mSv) per year and 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 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 (see Map 3). RDP and LLAA concentrations in the air are measured monthly and the results are compared with those from ERA's atmospheric radiological monitoring program.

Results

Radon pathway

Figure 1 shows the quarterly RDP data from Jabiru Water Tower, Jabiru East and the Mudginberri Four Gates Road Radon Station measured by ***eriss*** from early 2003 to September 2009. Two Environmental Radon Decay Product Monitors (ERDM) are now used for RDP monitoring. The instrument located at Mudginberri Radon Station logs data continuously, whereas the second instrument is moved between sites at Jabiru and Jabiru East.

A two sample t-test shows there is no statistically significant difference ($p = 0.974$) between the RDP concentrations measured at Jabiru Water Tower and Mudginberri Four Gates Road Radon Station, the latter of which is considered a background site. Average RDP concentration measured from July 2003 to June 2009 at the Mudginberri and Jabiru Water Tower site is 0.046 $\mu J/m^3$. The Jabiru East values are significantly higher ($p = 0.003$) and the average is 0.072 $\mu J/m^3$. RDP concentrations at Jabiru East 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.

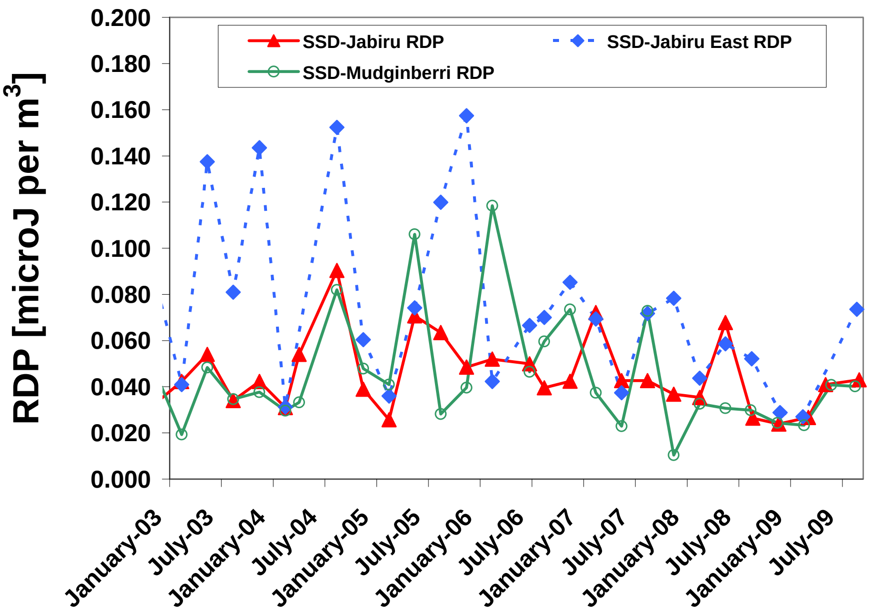


Figure 1 Radon decay product concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

In Jabiru, most of the mine origin radon has dispersed, and variations in 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 the results from this study have been published (Lawrence et al 2009).

ERA estimates the mine origin RDP using a wind correlation model and calculates the exposure via the radon pathway. Table 1 shows the annual averages for the radon decay product concentrations measured by ***eriss***, and reported by ERA, at Jabiru and Jabiru East, and the calculated total annual doses from RDP inhalation. This dose calculation assumes an 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 2008 is very small and is only about 3% of the public dose constraint.

Table 1 Average radon decay product concentrations (ERA 2008, in brackets) at Jabiru, Jabiru East and Mudginberri, and associated total and mine derived annual doses received at Jabiru, between 2006 and 2008

		2006	2007	2008
	Jabiru East	0.066 (0.071)	0.064 (0.059)	0.046 (0.033)
RDP concentration [µJ/m³]	Jabiru	0.046 (0.039)	0.049 (0.038)	0.038 (0.037)
	Mudginberri	0.075	0.036	0.029
Total annual dose [mSv] Jabiru		0.44 (0.38)	0.47 (0.37)	0.37 (0.36)
Mine derived dose [mSv] at Jabiru[a]		0.003	0	0.010

[a] predicted from wind field model

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 early 2003 to July 2009. In 2007, permanent dust samplers were installed at Jabiru and the Mudginberri Four Gates Rd radon stations, to allow for regular sampling of LLAA at these locations.

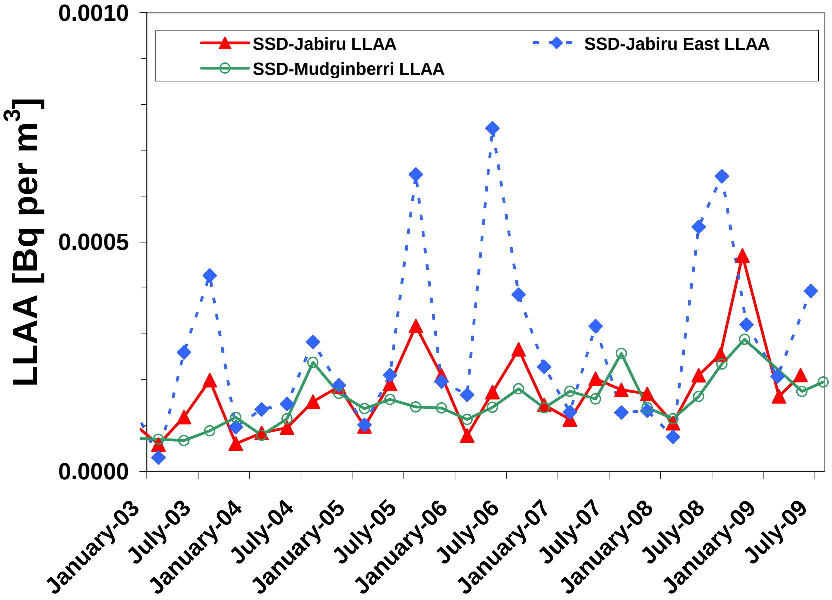


Figure 2 Long lived alpha activity concentration measured at Jabiru, Jabiru East and Mudginberri

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 early 2003 to June 2009 at Jabiru, Jabiru East and Mudginberri are 0.00018, 0.00028 and 0.00016 Bq·m^{-3}, respectively. There is no statistically significant difference between the Mudginberri and Jabiru sites ($p = 0.316$).

The total annual dose from inhalation of dust is calculated using a dose conversion factor for 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 was 11 µSv per annum in Jabiru for 2008, and only a small fraction (ie ~2 µSv for a person working in Jabiru East and living in Jabiru) of that dose would be mine-related (Bollhöfer et al 2006).

Steps for completion

The routine monitoring of dust and radon progeny will continue at Jabiru, Mudginberri Four Gates Road Radon Station and Jabiru East. Permanent dust samplers have now been installed at the three monitoring sites. Continuous RDP monitors have been acquired and tested at Mudginberri Four Gates Road Radon Station, and two additional units will be permanently deployed at the Jabiru Water Tower and the Jabiru Field Station in 2009.

Summary

Monitoring of radon and dust exposure pathways over the past 7 years has shown that the only significant contribution to radiological exposure of the public at Jabiru via inhalation is the inhalation of radon decay products. Although the contribution from the mine site has been shown consistently to be much less than the public dose constraint of 0.3 mSv per year and is of no concern according to current best practice standards, atmospheric monitoring will continue to provide re-assurance to the public that the risk from inhalation of mine derived radionuclides remains very low.

References

Bollhöfer A, Honeybun R, Rosman KJR and Martin P 2006. The lead isotopic composition of dust in the vicinity of a uranium mine in northern Australia and its use for radiation dose assessment. *Science of the Total Environment* 366, 579–589.

ERA 2009. Radiation Protection and Atmospheric Monitoring Program, Report for the Year Ending 31 December 2008. Energy Resources of Australia Ltd Ranger Mine, Jabiru NT.

Lawrence CE, Akber RA, Bollhöfer A and Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Martin P 2000. Radiological impact assessment of uranium mining and milling. PhD thesis. Queensland University of Technology, Brisbane.

Supervising Scientist 2007. *Annual Report 2006–2007*. Supervising Scientist, Darwin.

UNSCEAR 2000. *Source and effects of ionising radiation, vol 1–2*. United Nations Scientific Committee on the Effects of Atomic Radiation. United Nations, NY.

Zapantis A 2001. Derivation of the dose conversion factor for the inhalation of uranium ore dust considering the effects of radon loss. *Radiation Protection in Australasia* 18, 35–41.

Monitoring of radionuclides in groundwater at Ranger

B Ryan

Introduction

Groundwater samples are collected from Ranger uranium mine on an annual basis by the Northern Territory Department of Resources (DOR; formerly known as Department of Regional Development, Primary Industry, Fisheries and Resources). These samples are analysed for a suite of dissolved metals and radionuclides by the Supervising Scientist Division (SSD) and for metals and major ions by DOR. The results produced contribute to the understanding of the temporal and spatial variability in particular of uranium and radium in groundwater at the mine.

The data collected over the years assist in providing insights into contaminant sources and groundwater transport processes. More importantly, the contemporary and historical data from SSD, DOR and Earth Water Life Sciences (EWLS) will contribute to the production of an agreed pre-mining baseline dataset, which will in turn help develop closure criteria for groundwater quality and provide the basis for validating hydrogeological modelling of the backfilled pits and other rehabilitated structures.

The groundwater monitoring program will need to be continued during and following the mine's rehabilitation to assess the rehabilitation success and the integrity of the pits as tailings repositories.

Methods

Radionuclide activity and metal concentrations are measured in groundwater samples collected annually by DOR at the end of the dry season from selected bores around the site. Heavy metal concentrations are determined via a combination of ICP-MS and ICP-OES methods. Radionuclides of interest are radiochemically separated from the bulk water samples, and measured via alpha spectrometry. Radiochemical methods are published in Martin and Hancock (2004) and Medley et al (2005).

As thorium and lead are particle reactive and readily adsorbed and removed from solution, it is not expected that either of these metals will migrate significant distances through the groundwater aquifers. Consequently, the priority list for the more mobile U-series radionuclides that may contaminate the groundwater comprises: ^{238}U, ^{234}U and ^{226}Ra. These radioisotopes are the focus of the groundwater monitoring program. The $^{234}U/^{238}U$ activity ratio in particular may be a useful tracer to discriminate natural ($^{234}U/^{238}U$ generally > 1) from mine related ($^{234}U/^{238}U \approx 1$) uranium in groundwater (Ryan & Bollhöfer 2007).

Results

All the groundwater data that have been collected at Ranger by the Supervising Scientist Division over the last 25 years have been validated and checked for quality. A summary of the bores that have been monitored is presented in Table 1. The number of years for which data are available are listed in the table, although it should be noted that not all analytes were measured every year, and sometimes only a limited suite of dissolved metals were analysed. A full version of the table will be contained in an Internal Report to be produced on Ranger groundwater data collected by SSD.

Ranger bore histories and bore logs have been acquired from EWLS in 2009 and this information will be used to enhance the knowledge and understanding of the Ranger groundwater sampling sites. Lithological and stratigraphical information from these logs will allow the standardising of geological descriptions. The screen depths and hence the aquifers the bores draw from will be able to be assigned to each hole. All of this information will be placed on the SSD Ranger groundwater GIS database which is being developed to facilitate the review and examination of the data.

Table 1 A summary of all historical groundwater data collected by SSD 1984–2008

Bore	Collection years	^{234,238}U	^{210}Po	^{226,228}Ra	^{230,232}Th	^{227}Ac	^{210}Pb	^{40}K	^{137}Cs	ICPMS
79/1	85–96	x		x	x		x			
79/2	85–89	x		x	x		x			
79/6A	89–92	x		x	x	x	x	x	X	
79/9	89–96	x		x	x		x	x		
83_1	03–06	x		x						x
83_1 Deep	06–07	x		x						x
B11	Sep-06–07	x		x						x
C1SHALLOW	03	x		x						
MBH	03	x		x						
MBL	03	x		x						
MC24	03	x		x						
MC27	03–04	x		x						
MC27 Deep	Sep-06	x		x						
OB1A	89–08	x		x	x	x	x	x	X	x
OB2A	88–02	x		x						
OB4A	88–01	x		x	x	x	x	x	X	x
OB6A	88–02	x	X	x	x	x	x	x	X	x
OB7A	89–02	x		x	x	x	x		X	
OB9A	88 -99	x	X	x	x	x	x	x	X	x
OB10A	89–02	x	X	x	x	x	x	x		
OB11A	85–97	x	X	x	x	x	x	x	X	
OB12A	89–96	x		x	x	x	x			
OB13A	85–98	x	X	x	x	x	x		X	
OB15	88–96	x	X	x	x	x	x	x	X	
OB15A	96–97	x								
OB16	85–97	x	X	x	x	x	x	x	X	
OB17A	88–02	x		x	x		x	x	X	x
OB18A	88	x	X	x	x	x	x	x	X	
OB19A	89–2	x	X	x	x	x	x		X	x
OB20	89–08	x		x	x		x	x		x
OB21A	89–08	x		x	x		x	x	X	x
OB22	84–89	x	X	x	x	x	x	x		x
OB23	89–08	x		x	x		x	x		x
OB24	89–02	x		x	x		x			X
OB26	88–89	x	X	x	x	x	x	x		
OB27	03–08	x		x						x
OB28	89–00	x		x	x		x	x		
OB29	85–02	x		x	x		x	X		x
OB30	89–08	x		x	x		x	x		x
OB41	89			x	x		x	x	X	

Bore	Collection years	^{234,238}U	^{210}Po	^{226,228}Ra	^{230,232}Th	^{227}Ac	^{210}Pb	^{40}K	^{137}Cs	ICPMS
OB43	89			x	x		x		X	
OB44	89–02	x	x	x	x	x	x	x		x
OB46	89	x	X	x	x	x	x			
OB47	89–90	x	X	x	x	x				
OB48	89–90	x	X	x	x	x				
OB49	89	x	X	x	x	x				
OB50	89	x	X	x	x	x				
OB51	89	x	X	x	x	x				
OB79/6A	92	x								
OB79B	96	x								
RN23551	89–07	x		x	x	x				x
RN9329	03–08	x		x						x
RP1N1	03	x		x						
RP1N2	03	x		x						
TDSCN	89	x		x		x				
TDSC-S	89					x				
TD(East Wall)	89	x		x						
TD(Westall)	89					x				
C12	08									x
MC24	03									x
MC27	03–04									x
MC27DEEP	06									x
RN22211	08									x
RN23562	08									x

Steps for completion

SSD has reviewed and organised its historical groundwater data and placed the data into spreadsheets and databases. All spatial information for the bores sampled by SSD has been entered into the SSD Ranger groundwater ArcGIS database. The aim for 2009–10 is for DOR and EWLS to complete the processing of their respective available data sets so data can be combined and entered into an overall groundwater GIS database, with each organisation validating and entering their data into this database.

It is then intended to review the data from a hydrogeological perspective and determine their appropriateness for representation of the groundwater characteristics in the Ranger area. In particular, the aim is to produce an agreed set of groundwater quality baseline data that will be used to develop closure criteria for Ranger groundwater. This is a high priority objective given the hydrogeo/chemical modelling outputs that will be required to support the closure strategy for Pit 1 in the short term, and Pit 3 in the longer term.

The review will also provide the basis for each organisation to reassess the scopes of their respective groundwater sampling programs and allow any significant inadequacies to be addressed. The information gained will form the basis of recommendations for possible changes to the scope and extent of far-field groundwater quality monitoring at Ranger and the involvement of the Supervising Scientist Division.

Acknowledgments

The Northern Territory Department of Resources is acknowledged for collection of the bore water samples and for providing the aliquots of water to SSD for analysis. EWLS and DOR are acknowledged for providing bore histories and associated data.

References

Martin P & Hancock GJ 2004. *Routine analysis of naturally occurring radionuclides in environmental samples by alpha-particle spectrometry*. Supervising Scientist Report 180, Supervising Scientist, Darwin NT.

Medley P, Bollhöfer A, Iles M, Ryan B & Martin P 2005. Barium sulphate method for radium-226 analysis by alpha spectrometry. Internal Report 501, June, Supervising Scientist, Darwin. Unpublished paper.

Ryan B & Bollhöfer A 2007. A summary of radionuclide activity and dissolved metal concentrations in Nabarlek borewaters from 1996 to 2005. Internal Report 530, September, Supervising Scientist, Darwin. Unpublished paper.

Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

P Medley, A Bollhöfer & K Turner

Introduction

Surface water samples in the vicinity of the Ranger and Jabiluka project areas are routinely measured for their radium-226 (^{226}Ra) activity concentrations to check for any significant increase in ^{226}Ra activity concentrations 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 radium, which may then be incorporated into the human body upon consumption. Water samples are collected weekly in Magela Creek (Ranger) from both upstream and downstream sites, and monthly from Ngarradj Creek (Jabiluka) downstream site. Samples are not collected from these locations during the dry season when there is no contiguous surface water flow.

All Ngarradj samples are analysed for total ^{226}Ra by ***eriss***'s environmental radioactivity laboratory using a method described in Medley et al (2005). From the 2006–07 wet season onwards, weekly samples obtained from Magela Creek have been combined to give monthly averages. Analyses of the complete data set and combined wet season samples from previous years had shown that combining weekly samples would provide valid monthly radium results.

Results

Magela Creek

The ^{226}Ra activity concentration data for the 08–09 wet season in Magela Creek are compared with previous wet seasons in Figure 1.

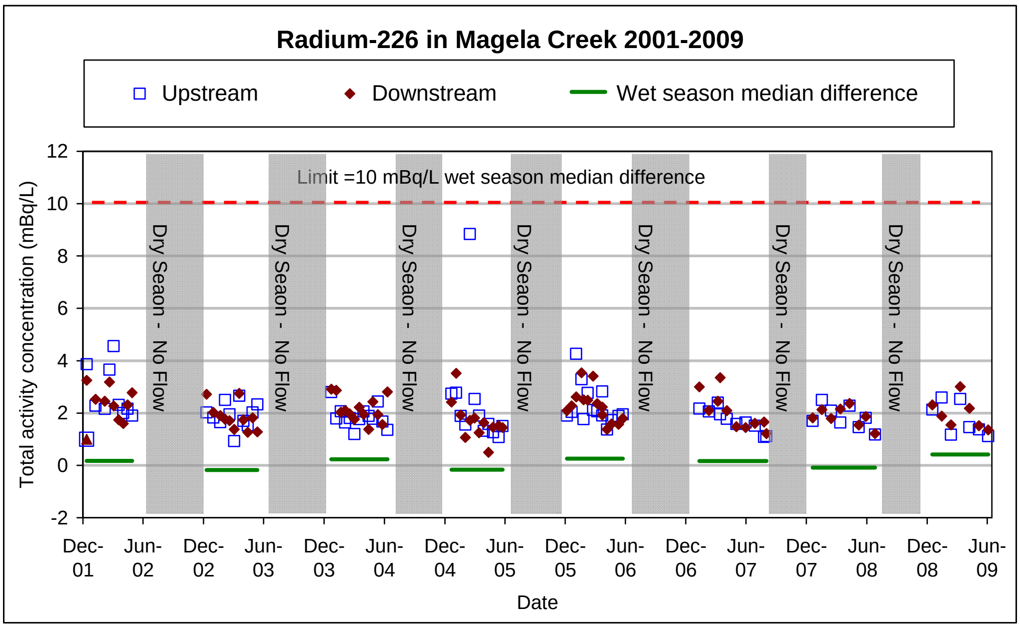


Figure 1 Radium-226 in Magela Creek for the 2001–09 wet seasons

The data show that not only are the levels of ^{226}Ra very low both upstream and downstream of Ranger mine, but also there is no difference between these locations. Wet season median values for each location and the wet season median difference between locations are reported in Tables 1 and 2.

Table 1 Median and standard deviations of the ^{226}Ra activity concentration for individual wet seasons (2002–06)

		2002–03	**2003–04**	**2004–05**	**2005–06**
Magela Creek					
Median and standard deviation	upstream	2.0 (± 0.5)	1.8 (± 0.4)	1.7 (± 2.1)	2.0 (± 0.8)
	downstream	1.8 (± 0.5)	2.0 (± 0.5)	1.6 (± 0.7)	2.3 (± 0.7)
Wet season median difference		-0.2	0.2	-0.2	0.3
Ngarradj					
Median and standard deviation	upstream	1.4 (0.6)	1.1 (± 0.4)	1.3 (± 0.3)	1.0 (± 0.4)
	downstream	1.1 (1.5)	0.9 (± 0.9)	1.0 (± 0.6)	0.5 (± 0.5)
Wet season median difference		-0.3	-0.2	-0.3	-0.5

Table 2 Median and standard deviations of the ^{226}Ra activity concentration for individual wet seasons (2006–09) and for the entire study period (2001–09)

		2006–07	**2007–08**	**2008–09**	**All years 2001–09**
Magela Creek					
Median and standard deviation	upstream	1.7 (± 0.4)	2.0 (± 0.5)	1.5 (± 0.6)	1.9 (± 1.0)
	downstream	1.9 (± 0.7)	1.9 (± 0.4)	1.9 (± 0.6)	1.9 (± 0.6)
Wet season median difference		0.2	-0.1	0.4	0.0
Ngarradj					
Median and standard deviation	upstream	1.1 (± 0.5)	N/A	N/A	1.1 (± 0.5)
	downstream	1.0 (± 0.3)	1.0 (± 0.3)	1.0 (± 0.3)	1.0 (± 1.7)
Wet season median difference		-0.1	N/A	N/A	-0.1#

median for 2001–2007

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 human radiological protection purposes (Klessa 2001). This value was based on the potential dose received from the ingestion of ^{226}Ra in the freshwater mussel *Velesunio angasi* (Martin et al 1998).

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 green lines in Figures 1 and 2) and should not be more than the limit of 10 mBq/L. The wet season median difference for the entire 2001–09 wet season data set is approximately zero. The data for the eight sampling seasons indicate that

^{226}Ra levels in Magela Creek are due to the natural occurrence of radium in the environment (upstream data) and that ^{226}Ra activity concentrations in Magela Creek water are not elevated (wet season median difference of zero) downstream of Ranger mine.

Ngarradj Creek

^{226}Ra activity concentrations in Ngarradj are very low. Although there were significant upstream-downstream differences observed in individual samples during the first two wet seasons, Figure 2 shows that ^{226}Ra activity concentrations at the Ngarradj downstream site have been similar to those at the upstream site since December 2003. This coincides with the establishment of the long-term care and maintenance phase at Jabiluka in the 2003 dry season. The wet season median difference is approximately zero for all years, except for the 2001–02 wet season. However, even in that season the wet season median difference was very low (< 2 mBq·L^{-1}), indicating human health was not at risk from the presence of ^{226}Ra in Ngarradj.

Since monitoring data from 2003 onwards have shown that there has been no significant difference between upstream and downstream values, and moreover since the absolute values are in any case very low and barely above detection limit, monitoring at the upstream site has been discontinued while Jabiluka remains in long-term care and maintenance. From the 2007–08 wet season onwards, the downstream data for each season are compared with the previous season's data for this location to check that there are no significant upward deviations from this control record. ^{226}Ra results (monthly samples) for the 2008–2009 wet season at the Ngarradj downstream site are comparable with the very low values of previous years, indicating that the downstream environment remains unimpacted. A t-test indicated that there is no significant difference between the 2008–09 data and the previous 5 wet seasons (P = 0.476).

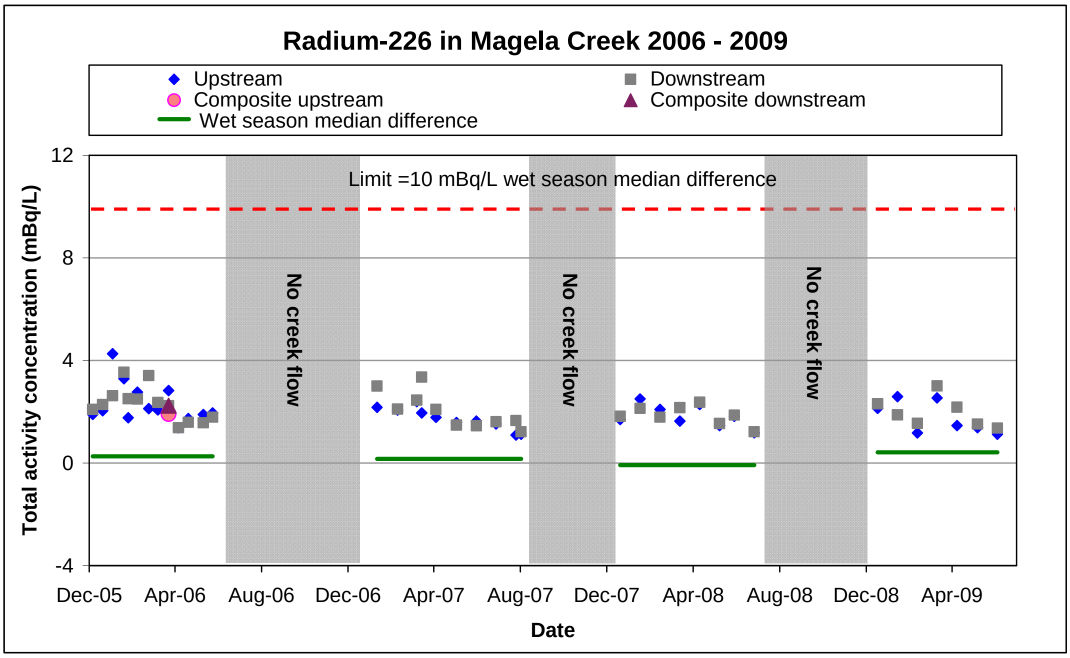


Figure 2 ^{226}Radium in Ngarradj Creek for the 2001–09 wet seasons

References

Klessa D 2001. Water Quality in Magela creek upstream and downstream of Ranger. Internal Report 380, Supervising Scientist, Darwin, Unpublished paper.

Martin P, Hancock GJ, Johnston A & Murray AS 1998. Natural-series radionuclides in traditional north Australian Aboriginal foods. *Journal of Environmental Radioactivity* 40, 37–58.

Medley P, Bollhöfer A, Iles M, Ryan B & Martin P 2005. Barium sulphate method for radium-226 analysis by alpha spectrometry, Internal Report 501, Supervising Scientist, Darwin, Unpublished paper.

Sauerland C, Martin P & Humphrey C 2005. Radium 226 in Magela Creek, northern Australia: Application of protection limits from radiation for humans and biota, *Radioprotection*, Suppl 1, 40(2005), 451–456.

Surface water transport of mine-related solutes in the Magela Creek catchment using continuous monitoring techniques

K Turner & 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 RP1 and GC2). These data are used for the assessment of potential impacts arising from activities carried out on the mine site (Supervising Scientist 2007, 2008, Turner et al 2008a,b, Turner 2009, Turner & Jones 2009). Relevant background information has been reported previously at ARRTC 22 (see Turner & Jones 2009).

A critical attribute of SSD's continuous monitoring network is the ability to remotely monitor (via 3G telemetry) events in the creek system. Telemetry provides a means for early warning of increases in sediment or solute inputs 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). Since 2005–06, Mg loads in Magela Creek have been derived each wet season using continuous EC data recorded at 10 minute intervals. 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 from upstream of the mine, a dynamic assessment of the intra- and inter-seasonal fluxes of salts in the system can be made.

Methods

Detailed continuous monitoring methods have been reported previously at ARRTC 22 (Turner & Jones 2009). Continuous monitoring over 2008–09 wet season included some changes to the reported methods summarised as follows:

1 Up until the 2008–09 wet season, two downstream monitoring stations were maintained, each located in different channels in an anastomosed section of Magela Creek slightly downstream of G8210009. During 2008–09 wet season, only the station located within the western-most channel of Magela Creek was monitored (MCDW);

2 Grab samples collected as part of the routine weekly grab sampling program were collected alongside the pontoons, allowing direct comparison between grab and continuous data (see 'Results of the stream monitoring program in Magela Creek catchment', ***eriss*** research summary 2007–08, Brazier 2009);

3 The automatic samplers were programmed to collect:

   a weekly samples at the upstream and downstream stations on the same day as the routine weekly grab sampling program;

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

Results

The flow conditions in each of 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 flow volumes for Magela Creek (as measured at GS210009, adjacent to the 009C compliance site) since inception of the continuous monitoring program 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 2005

Year	Annual cumulative rainfall (mm)	Annual cumulative discharge (GL)
2005–06	2107	485.4
2006–07	2540	845.2
2007–08	1673	416.6
2008–09	1186	235.2

Electrical conductivity – magnesium relationships

Relationships between EC and Mg at each of the four continuous monitoring locations (MCUGT, MCDW, RP1 and GC2) have been derived by correlating Mg concentrations in grab water samples with concurrent measurements of in situ EC. These relationships were reported at ARRTC 22 (see Turner & Jones 2009).

The data collected over the 2008–09 wet season have been used to refine the EC-Mg relationships at each of the sites. This was particularly important for:

1 MCDW as it lacked higher EC (>35 µS/cm) data points to provide a sufficiently high level of confidence in the fit of the data in this region of the relationship (Turner et al 2009);

2 RP1 as Mg concentrations have been on an upward trend over the past few years (ERA 2008).

Inclusion of the higher concentrations of Mg in the MCDW (captured using event-based sampling) and RP1 datasets has resulted in the relationships of best fit changing from linear to a slightly curved quadratic (Figure 1). A linear relationship is still the best fit for the Mg-EC relationships for MCUGT and GC2 (Figure 1).

Different EC-Mg relationships exist for each of the sites as a result of different Mg sources, concentration ranges and relative contributions of the constituent major ions present at each of the sites (Figure 1.). The slope of the regressions for RP1 and GC2 are similar where EC <500 µS/cm, consistent with the similarity of major ion compositions at both sites (Figure 1b and 1c respectively). The non-linear relationship for RP1 where EC >500 µS/cm is due to the formation of the zero-charged ion pair ($MgSO_4^0$) which occurs in waters with higher solute concentrations. A neutral ion pair does not contribute to the measured EC. Solution speciation modelling using the thermodynamic computer model MINTEQA2 indicates that at the highest concentrations of Mg measured in RP1, the neutral ion pair accounts for approximately 25% of the Mg present. As a result of the formation of the neutral ion pair in RP1, the relationship between EC and Mg is quadratic (Figure 1b).

The slopes for the Magela Creek upstream and downstream sites are similar for periods of flow characterised by EC values of 0 to 20 µS/cm, during which periods the solute load is dominated by water from upstream of the mine site (Figure 1a). This condition can occur when there is little or no input from the minesite, or during flood flows where total load is dominated by solutes coming from upstream. For EC >20 µS/cm, the slope for the downstream site is higher, indicating a greater influence of Mg on EC compared with other solutes present, as is expected with input of Mg-dominated mine waters. The existence of these two regimes at the downstream site is a result of variable mixing of upstream waters with mine waters. The resultant composite fit for the EC-Mg relationship is best described by a quadratic function (Figure 1a).

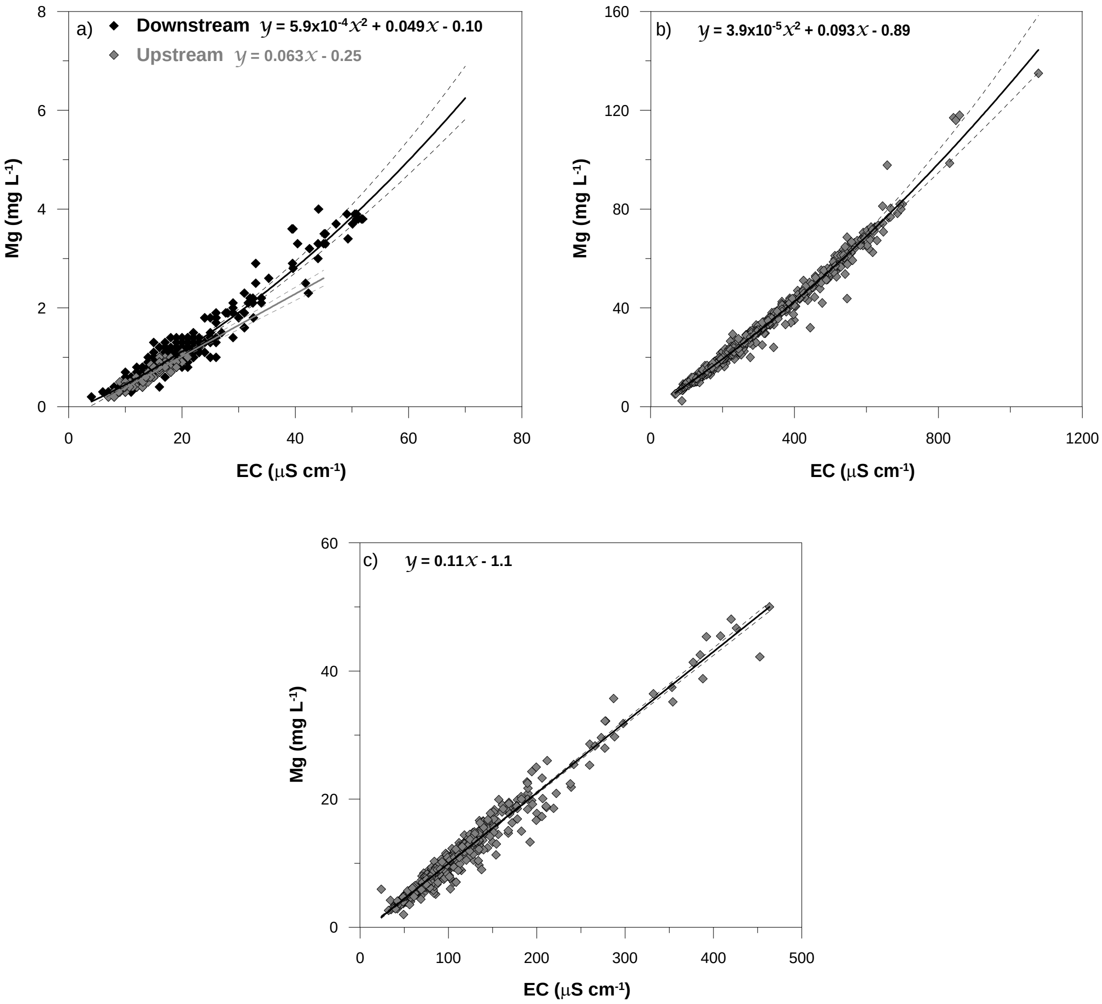


Figure 1 Relationships between EC and Mg concentration and upper and lower 95% confidence limits for the a) upstream ($R^2 = 0.86$, $P<0.0001$) and downstream ($R^2 = 0.94$, $P<0.0001$) sites on Magela Creek, b) RP1 ($R^2 = 0.99$, $P<0.0001$) in the Coonjimba Creek catchment and c) GC2 ($R^2 = 0.96$, $P<0.0001$) in the Corridor Creek catchment

Magnesium loads

The method used for predicting Mg concentrations using the continuous EC data was described in ARRTC 22 (see Turner & Jones 2009).

Minesite point sources

The Mg concentrations predicted using the continuous EC measurements in the minesite tributaries have been used in conjunction with the measured flows at these locations to calculate

Mg loads moving down these catchment lines through each wet season since 2005–06 (Table 2).

Table 2 Estimated Mg loads (t) exported from Coonjimba (RP1) and Corridor Creeks (GC2) for the 2005–06 to 2008–09 wet seasons

Year	RP1		GC2	
	Volume discharge (GL)	Mg load	Volume discharge (GL)	Mg load
2005–06	1.4	55	2.6	14
2006–07	3.4	110	4.9	17
2007–08	3.1	160	3.4	20
2008–09	0.35	61[1]	1.3	29

[1] Total is made up of the sum of the load passively discharged (46 t) over the RP1 weir and the load discharged from pumping and siphoning (15 t)

The Mg loads in Table 2 are within the range of previously reported values for RP1 (derived using interpolated weekly grab sample data), with the low value for the 2008–09 wet season reflecting the well below average wet season rainfall (ERA 2008a).

The increased annual Mg load exported from GC2 during 2008–09 is due to additional Mg inputs to the Corridor Creek system, including dry season surface flows from the Pit 1 catchment works and a period of pumped discharge of water (7.05 ML) from RP1 into the upper Corridor Creek catchment during February 2009 (ERA 2008a).

Mine site diffuse sources

To provide an estimate of the Mg load potentially available for export to Magela Creek from the Land Application Areas (LAAs) via shallow groundwater flow during a given wet season, the Mg load added to each of the LAAs during the antecedent dry season has been estimated using water systems management data supplied by ERA in its Annual Wet Season (2008a) and Annual Environment Reports (2008b). The total annual load of Mg applied to each LAA has been estimated using monthly irrigation volumes taken from Ranger Annual Environment Reports (October 2005 to July 2009) and mean monthly Mg concentrations in irrigation waters (RP2 for MLAA and JELAA and cell 9 of RP1WLF for Djalkmara LAA) (Table 3) (ERA 2008a).

Table 3 Mg loads (tonnes) applied to the Magela, Jabiru East and Djalkmara LAAs

Antecedent dry season	Magela LAA	Jabiru East[1]	Djalkmara	Total Mg load
2005	69.5	–	37.4	106.9
2006	75.1	57.2	55.1	187.4
2007	86.3	56.2	64.5	207.0
2008	0.2	2.8	5.4	8.4

[1] Jabiru East was commissioned in 2006

Magela Creek

Magnesium loads in Magela Creek have been calculated over the past four wet seasons using the continuous EC data measured at MCUGT and MCDW and total Magela discharge measured at GS8210009 (Table 4).

The Mg loads measured upstream in Magela Creek during the 2008–09 wet season were lower than for previous years which is consistent with the lower rainfall and consequent runoff experienced in the region during this period. The loads measured at RP1 and GC2 are consistent with values from previous years. The lower loads applied to the LAAs during the 2008 dry season reflect the lower rainfall (and hence runoff) of the preceding wet season.

Table 4 Mg loads (t) measured in Magela Creek (upstream and downstream of the mine) and mine waters (RP1 and GC2) and applied to LAAs

Time period	Magela Creek		Minesite		
	US	DS	RP1	GC2	LAAs
2005–06	174	405	55	14	106.9
2006–07	140	592	114	17	187.4
2007–08	145	371	163	20	207.0
2008–09	82	203	61	29	8.4

US = Upstream; DS = downstream

Load balance at Magela Creek downstream

The total annual Mg load measured in Magela Creek downstream of the mine (*DS*) in any given wet season should be described by Equation 1. Note that LAAs on mine site tributaries (RP1 LAA and Corridor Creek LAA) are assumed to report to Coonjimba Creek or Corridor Creek upstream of the monitoring points RP1 and GC2, respectively, and hence are accounted for in the loads estimated at these point sources.

$$DS = US + RP1 + GC2 + ROC \qquad (1)$$

DS = the total annual Mg load measured in Magela Creek downstream of the mine

US = the natural background Mg load for the Magela Creek catchment upstream of the mine site

RP1 = the Mg load input from the Coonjimba Creek catchment including RP1

GC2 = is the Mg load input from the Corridor Creek catchment

ROC = is the Mg load from the rest of the catchment which should be dominated by wet season washout of shallow groundwater from the LAAs on the mine site that are adjacent to Magela Creek (Magela LAA, Djalkmara LAAs and Jabiru East LAA)

Currently, there are two unknowns or unconstrained terms in the above equation. Firstly, the Mg load estimated at the downstream site is a potential overestimate since it is derived using EC data from the west channel only, and Magela flow across all three channels (described in Turner & Jones 2009, Development of Magela Catchment area solute budget using continuous monitoring systems). Secondly, the extent of inter-seasonal washout of Mg from the soil profile in the LAAs has to be inferred as there is no direct measure of this. If there was complete washout, the difference between the loads at the upstream site and the downstream site should equate to the input of mine-derived solutes. Table 5 compares the difference between the upstream and downstream Mg loads in Magela Creek with the sum of potential inputs from mine sources.

The RPD between the measured and predicted downstream Mg loads is typically >100% (with the exception of the anomalous 2007–08 season where the RP1 discharge is questionable), with the measured downstream load being greater than the sum of mine-derived inputs. This is likely to be because the loads estimated at the downstream site are overestimates by virtue of the cross-channel gradient in EC that occurs at low to medium flows at this location (Supervising Scientist 2008).

Table 5 Comparison of the difference between measured and predicted downstream MG loads (tonnes)

Time period	Measured DS load	Predicted DS load (US + RP1 + GC2 + ROC)	RPD%
2005–06	405	350	131
2006–07	592	458	142
2007–08	371	535	58
2008–09	203	180	123

DS = Magela Creek downstream

US = Magela Creek upstream

RPD% = relative % difference between measured and predicted DS Mg load

This flow-dependent lateral distribution of mine-derived Mg has implications for deriving the total Mg load for the creek (across all channels), *DS*, as the apportioning of the total stream discharge (and EC) between the three channels has not previously been well-defined as a function of flow. Since the Mg loads estimated at MCDW have been calculated by multiplying the total flow across Magela Creek by the characteristically higher Mg measured in the western channel, the loads derived using this procedure are likely overestimates.

This particular issue was addressed during the 2008–09 wet season by measurement of cross channel EC profiles and concurrent discharge in the western channel at MCDW. An Acoustic Doppler Current Profiler (ADCP) was acquired by SSD to facilitate routine measurement of cross channel stream discharges in Magela Creek. To determine the proportion of flow traveling down the western channel at MCDW, the discharge measured in this channel alone was compared to the total discharge measured concurrently at GS8210009 (Magela Creek discharge across all three channels). The data obtained for the 2008–09 wet season (8 gaugings carried out at MCDW over 5 days) show that a log relationship (R^2 = 0.98) exists between the flow at the two sites (Figure 2). Up to 60% of the total Magela flow travels down the western channel at MCDW under low flow conditions (≤ 20 m3/s) (Figure 3). The proportion of total Magela flow travelling in the western channel at MCDW decreases with increasing total flow. Under high flow conditions, greater proportions of the total Magela flow travel along the central and eastern channels.

During the 2008–09 wet season, ERA carried out some cross-sectional EC profiling in Magela Creek at the compliance site G8210009, located a few hundred metres upstream of MCDW. The data provided by ERA showed that when Magela flow was between 20 and 120 m^3/s, there was a definite EC gradient across the stream, with higher EC measured close to the west bank compared with the EC measured along the cross section profile towards the central channel. More intensive cross sectional EC and flow profiling will be carried out by SSD during the 2009–10 wet season at both G8210009 and MCDW.

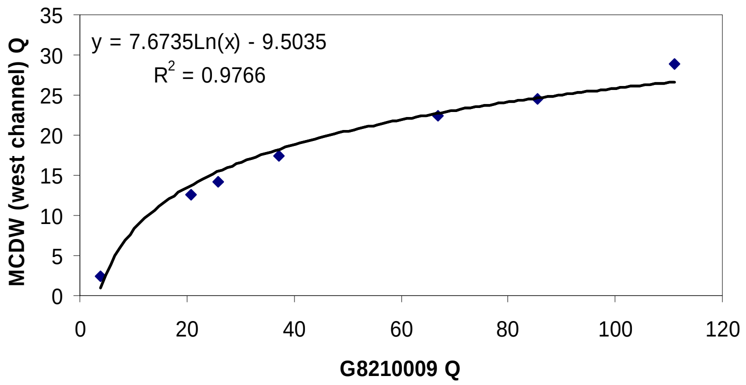


Figure 2 Discharge measured in the western channel at MCDW against total Magela Creek discharge measured at G8210009

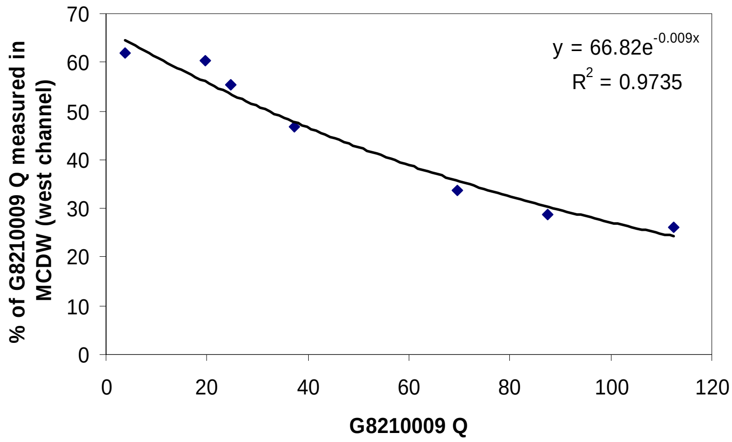


Figure 3 Percentage of discharge travelling along the western channel as a function of total Magela Creek discharge measured at G8210009

Steps for completion

Construction of a reliable load balance is dependent on resolving one or both of the unknowns in equation 1. Recent analysis of flow gaugings carried out at MCDW has shown that it may be possible to use the continuous discharge measured at G8210009 to predict flow in the western channel at MCDW, thereby increasing the reliability of the Mg loads calculated at this site. While there is a significant non-linear relationship between total Magela Creek flow and the flow in the western channel at MCDW for measurements made between January and March 2009, more data from high flow events are needed before flow measured at G8210009 can be used as a reliable predictor under the full range of flow conditions for west channel flow at MCDW. Gaugings will be conducted over a greater range of flows to determine if the current relationship applies to higher flows. This work will be done during the 2009–10 wet season.

The total Mg load transported in Magela Creek downstream of the mine, *DS*, will then be able to be calculated by adding the loads measured at MCDW and the loads estimated in the central and eastern channels (using the distribution of flow between the three channels and the concentration of Mg from upstream). Once the *DS* loads have been adjusted accordingly, then equation 1 can be rearranged to solve for *ROC* which will allow quantification of Mg inputs derived from the LAA and any other potential diffuse sources.

References

Brazier J 2009. Chemical and physical monitoring. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 46–50.

ERA 2008a. *ERA Ranger Mine Wet Season Report*. Energy Resources of Australia Ltd, Darwin, NT.

ERA 2008b. *ERA Annual Environment Report*. Energy Resources of Australia Ltd, Darwin, NT.

Supervising Scientist 2007. *Annual Report 2006–2007*. Supervising Scientist, Darwin.

Supervising Scientist 2008. *Annual Report 2007–2008*. Supervising Scientist, Darwin.

Turner K 2009. Continuous monitoring of water quality in Magela Creek. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A. Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 71–74.

Turner K & Jones D 2009. Development of Magela Catchment area solute budget using continuous monitoring systems. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A. Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 75–85.

Turner K, Moliere D, Humphrey C & Jones D 2008a. Continuous monitoring of water quality in Magela Creek. In ***eriss** research summary 2006–2007*. eds Jones DR, Humphrey C, van Dam R & Webb A. Supervising Scientist Report 196, Supervising Scientist, Darwin NT, 37–42.

Turner K, Moliere D, Humphrey C & Jones D 2008b. Characterisation of solute transport in a seasonal stream using continuous in-situ water quality monitoring, In *Water Down Under 2008,* Proceedings of the 31st Hydrology and Water Resources Symposium and 4th International Conference on Water Resources, 15–17 April 2008, Adelaide, South Australia, Engineers Australia (CD).

Review of solute selection for water quality and bioaccumulation monitoring

K Turner & D Jones

Background

The suite of major ions and trace metals measured by SSD as part of its surface water chemistry and bioaccumulation monitoring programs was initially selected based on a number of assessments carried out prior to commencement of mining, and during the mine's initial operational period (Office of the Supervising Scientist 2002, Klessa 2000, 2001a,b). Baseline data were collected as early as 1975 to characterise natural inputs to Magela Creek. Potential metals of concern for the future mining operation were identified by comparing metal concentrations in samples of ore and waste with 'background' concentrations from unmineralised areas, coupled with the results from a leaching study conducted on a pad of material from the Ranger 1 orebody (Milnes & Fazey 1988). The suite of metals identified by these early studies has largely been maintained to the present day in the routine analysis of water samples collected by SSD. However, the reason why some of these metals continue to be analysed is for reasons of sample quality control rather than environmental impact assessment. The latter aspect is discussed in more detail below.

There are many metals/metalloids that have historically been identified worldwide as having a potential to bioaccumulate. They are (listed in order of decreasing potential to bioaccumulate) cadmium, mercury, selenium, arsenic, beryllium, chromium, cobalt, copper, iron, lead, manganese, silver, uranium, vanadium and zinc. All of these metals were analysed as part of the trace element suite for the original SSD bioaccumulation program. However, it was found that all were present at extremely low levels in mussels and a number of fish species.

Rather than simply continuing to analyse, a priori, for all of these metals in biota collected in Magela Creek and in billabongs downstream of Ranger, it was determined that a risk assessment should be carried out using the results from chemical analysis of catchment drainage lines contributing metals to the system. This approach took the assessment of relevant metals beyond that based on simple comparison of mineralised and unmineralised rock since it will identify those metals that are actually being dissolved by contact with water and hence potentially capable of being transported downstream along minesite catchments.

Comparison of the composition of minesite waters with composition of the water from Magela Creek upstream and downstream of the mine enabled a risk assessment to be made of those metals that are of most potential concern. It is those metals that are present at higher-than-background levels downstream of the mine that should be considered more closely for inclusion in the routine water quality monitoring program to assess downstream impact of operations at the Ranger mine.

A detailed chemical assessment of the full trace metal profile of minesite waterbodies and major catchment runoff lines had not been carried out since the cessation of mining of Pit1 and the start of mining of Pit 3 in 1996. Since that time, the waste stockpiles have come to be dominated by material from Pit 3, and it is possible that the trace element composition of runoff and seepage water could have changed as a result of the different provenance of this second orebody. Consequently, contemporary trace element data were required since the previous data would not provide a sufficiently robust basis upon which to carry out a contemporary risk assessment.

To this end, comprehensive chemical analysis of on-site waterbodies, catchment drainage waters and Magela Creek upstream and downstream of the minesite was undertaken during the 2005–06 wet season. A range of analytes was identified to be of potential environmental importance, based upon: i) concentrations present in mine waterbodies relative to background concentrations; ii) attenuation by natural processes in catchment drainage lines; and iii) likely or inferred potential for biological impact.

Ranger mine and Magela Creek

The minesite is located within the Magela Creek catchment area which runs through the north-east corner of the Ranger Project Area. During the wet season months Magela Creek receives mine-derived waters that are passively released along Coonjimba and Corridor Creek catchment lines, as well as mine-related constituents that are leached from the land application areas (LAAs) in the vicinity of these creeks (see Map 1 for locations of these sites).

Both the Coonjimba and Corridor Creek catchment lines have been substantially modified, with the construction of wetland filters and various bunds and weirs to assist with flow control and passive water treatment. The Ranger Water Management Plan (ERA, 2005) outlines the company's water management practices and provides details of the components of the site water management system. In summary, the management system is divided into three components based on the source of the water and degree of interaction between the water and mining or milling processes.

- **Process water:** Confined to the tailings dam and Pit 1, process water has to be retained on site and can only be disposed of by evaporation or following treatment to a prescribed level.
- **Pond water:** Runoff and seepage from the mill and mine areas, including the low grade ore stockpiles, are directed to Ranger retention pond 2 (RP2). If RP2 capacity is reached, the excess water flows via a spillway structure into Pit 3. Pond water is currently disposed of or treated by a combination of methods, including wetland filtration, land application and treatment in a Microfiltration/Reverse Osmosis (MF/RO) plant.
- **Sediment control water:** Runoff from waste rock dumps and natural woodland areas that reports to RP1 (northern part of the minesite) and the Corridor Creek wetlands (southern part of mine site), which ultimately discharge into Magela Creek via the Corridor Creek and Coonjimba Creek flow lines, respectively.

Only the pond and sediment control waters can impact directly, or indirectly, on water quality in Magela Creek.

Methods

Surface water samples were collected from a number of tributaries and constructed waterbodies on the Ranger lease as well as upstream (control) and downstream (exposed) locations in Magela Creek (Table 1).

Minesite waterbodies that discharge to Magela Creek during the wet season were the focus of this study.

Table 1 Sampling sites

Site	Description	Major inputs
MCUS	Magela Creek upstream	Undisturbed areas of Magela catchment upstream of Ranger mine
009C & 009W	Magela Creek downstream, central channel and west channel, respectively	MCUS, Corridor Creek via Georgetown Billabong, RP1 via Coonjimba Billabong and land application area runoff
VLGCRC2	Very Low Grade Cross Road Culvert	Runoff from waste rock and low grade ore stockpile
CCWLF (Cell 1)	Corridor Constructed Wetland Filter	VLGCRC2, land application area runoff
GC2	Corridor Creek downstream	CCWLF, land application area runoff
RP2	Retention Pond 2	Water from Pit 3, runoff and seepage from stock piles, processing and milling area and haul roads
RP1	Retention Pond 1	RP1 Constructed Wetland Filter, land application area runoff and seepage from bunded structures in the upper catchment

During the 2005–06 wet season, samples were collected on four occasions (Table 2) over the period of initial, mid and recessional creek flow to determine the extent to which the composition of the waters changed over the course of the wet season and the effect, if any, this would have on the risk assessment.

Table 2 Sampling dates

Date	Sites		
	Status of discharge to Magela Creek	Mine lease area	Magela Creek
21 Dec 2005	Prior to release of RP2 and GC2 water to Magela Creek	RP2, RP1, GC2, CCWLF	MCUS, 009C, 009W
19 Jan 2006	Initial period of release form RP1 and GC2	RP2, RP1, GC2, VLGCRC2	MCUS, 009C, 009W
22 & 23 Mar 2006	Release flow established at RP2 and GC2	RP2, RP1, GC2, VLGCRC2	MCUS,009C, 009W
20 & 21 Jun 2006	After cessation of RP1 and GC2 water to Magela Creek	RP2, RP1, GC2	MCUS, 009C, 009W

On each sampling occasion pH, electrical conductivity (EC), temperature, dissolved oxygen (DO) and turbidity were measured in situ and in the laboratory. Water samples were filtered in the field at time of collection and acidified at SSD's Jabiru Field Station prior to analysis for an extensive suite of dissolved trace metals by Inductively Coupled Plasma Mass Spectrometry (ICPMS).

Results

The mean values of pH and EC measured in the Ranger retention ponds, GC2 and at the upstream and downstream sites in Magela Creek during the 2005–06 wet season are shown in Table 3. Each of the waterbodies studied showed some variation in pH over the wet season, most notably in RP1 and RP2, where pH decreased during the rainfall months. The locations sampled on the minesite had higher mean EC values compared with Magela Creek, reflecting the much higher concentrations of major ions in these site waters.

Table 3 Mean and standard deviation of pH, EC and turbidity from each site

Site	pH		EC	
	Mean	***SD***	**Mean**	***SD***
RP2	5.7	*1.2*	1252	*25*
RP1	7.2	*0.9*	423	*178*
GC2	6.4	*0.3*	101	*46*
009C	5.8	*0.5*	15	*4*
MCUS	5.8	*0.5*	13.5	*3.5*

Any metals measured in the mine-derived waters at concentrations less than the corresponding analytical detection limits were considered to present negligible risk to the environment. Hence they will not be discussed further. The mean concentrations of metals/metalloids present at levels higher than detection limits in mine waters and in Magela Creek over the 2005–06 wet season are presented in Table 4. For each element the sites are arranged in descending order of mean concentrations values, with the standard deviations associated with each mean shown in parentheses.

The concentrations of metals in RP2 and VLGCRC2 are higher than in RP1 and GC2. VLGCRC2 and RP2 both receive surface runoff and seepage from waste rock and low grade ore stockpiles and contain metals dissolved by water in direct contact with high surface areas of exposed rock. Although many metals are present in RP2 at elevated concentrations, they are not of direct risk to the surrounding environment as untreated RP2 water is not discharged into Magela Creek.

RP1 and GC2 receive waters that are ‘polished’ by passage through wetland filters as well as being further diluted by water from cleaner sub-catchments. RP1 and GC2 also receive runoff and seepage from land application areas, where metals initially present in the RP2 water are attenuated by absorption in the soil profile (Hollingsworth et al 2005).

The elements that may pose the greatest potential risk to the natural receiving aquatic system are those that are present in substantially higher concentrations in RP1 and GC2 relative to upstream Magela Creek, as water from both of these minesite locations ultimately discharges into Magela Creek. However, it must also be recognised that there are natural billabongs located between RP1 and GC2 and Magela Creek that provide additional polishing of discharge waters. In the case of GC2 it is Georgetown Billabong, and in the case of RP1 it is Coonjimba Billabong (see Map 2 for locations).

The log-transformed concentration data produced from the four wet season sampling occasions were analysed using ANOVA, followed by a Tukey’s test to distinguish significant differences ($P < 0.05$) between sites (Table 5). It is important to note that the low sample sizes and high standard deviations associated with data arising from seasonal sampling over the 2005–06 wet season will somewhat reduce the power of the statistical analyses.

Table 4 Summary of mean element concentrations (standard deviation) measured in mine waterbodies and in Magela Creek ordered from the highest to lowest concentration for each element

Element	Site			
	Concentration (standard deviation)			
Aluminium	GC2 109 (61.0)	MCUS 55.4 (39.7)	009C 55.3 (40.2)	RP1 20.0 (15.7)
Arsenic	RP1 0.23 (0.098)	GC2 0.183 (0.058)	009C 0.095 (0.057)	MCUS 0.079 (0.046)
Boron	RP1 18.8 (7.16)	GC2 12.7 (3.75)	009C 8.20 (0.570)	MCUS 8.00 (0.837)
Barium	RP1 36.6 (17.7)	GC2 15.3 (11.8)	009C 3.11 (0.498)	MCUS 2.86 (0.493)
Cadmium	RP1 5.86 (2.21)	GC2 2.10 (1.10)	009C 0.480 (0.192)	MCUS 0.467 (0.225)
Copper	GC2 1.32 (0.767)	RP1 0.406 (0.375)	MCUS 0.238 (0.231)	009C 0.190 (0.123)
Iron	GC2 213 (75.7)	009C 124 (32.9)	MCUS 113 (24.2)	RP1 64.0 (45.6)
Magnesium	RP1 58.8 (24.5)	GC2 9.27 (3.67)	009C 0.860 (0.313)	MCUS 0.650 (0.274)
Lead	GC2 0.237 (0.035)	RP1 0.072 (0.078)	MCUS 0.03 (0.024)	009C 0.023 (0.022)
Rubidium	RP1 9.84 (3.51)	GC2 3.76 (1.49)	009C 0.492 (0.095	MCUS 0.432 (0.088)
Sulfate	RP1 223 (89.2)	GC2 30.2 (20.0)	009C 1.2 (1.03)	MCUS 0.300 (0.089)
Uranium	GC2 11.4 (3.85)	RP1 7.55 (3.69)	009C 0.059 (0.021)	MCUS 0.0207 (0.008)

Aluminium, As, B, Ba, Ca, Cu, Fe, Mg, Pb, Rb, SO_4 and U were present in RP1 and/or GC2 at concentrations significantly higher ($P < 0.05$) than those measured upstream in Magela Creek. However, uranium and SO_4 were the only analytes that were significantly elevated ($P < 0.05$) at the downstream site in Magela Creek compared with the upstream site.

To further refine the above analysis, Student t-tests were used to compare the Magela upstream and downstream weekly water quality monitoring data measured between 2001 and 2009 (a large dataset, $n > 200$). Concentrations of Ca, Fe, Mg, Mn, SO_4 and U observed downstream of the mine were significantly ($P < 0.05$) higher than values observed at the upstream site. The remaining analytes (Al, Cu, Pb and Zn) were not significantly different between the upstream and downstream sites. This indicates that while there are many elements present at concentrations greater than background in the mine waters at source, the majority of these elements are essentially completely attenuated during passage of the water through the tributary creek lines, in Georgetown and Coonjimba Billabongs and by dilution or adsorption on particulates present in Magela Creek.

Table 5 Summary of results from one-way ANOVA and Tukey's *post hoc* tests on differences in element concentrations measured in mine waterbodies and in Magela Creek

Element	df	F	*P*	Tukey's HSD multiple comparison test – SITE – Concentration (standard deviation)			
Aluminium	3	3.903	0.030	GC2	MCUS	009C	RP1
Arsenic	3	4.316	0.022	RP1	GC2	009C	MCUS
Boron	3	12.67	<0.000	RP1	GC2	009C	MCUS
Barium	3	34.87	<0.000	RP1	GC2	009C	MCUS
Cadmium	3	38.32	<0.000	RP1	GC2	009C	MCUS
Copper	3	5.889	0.007	GC2	RP1	MCUS	009C
Iron	3	5.9	0.007	GC2	009C	MCUS	RP1
Magnesium	3	135.2	<0.000	RP1	GC2	009C	MCUS
Lead	3	8.01	0.002	GC2	RP1	MCUS	009C
Rubidium	3	111.8	<0.000	RP1	GC2	009C	MCUS
Sulfate	3	152.1	<0.000	RP1	GC2	009C	MCUS
Uranium	3	242.0	<0.000	GC2	RP1	009C	MCUS

Sites joined by a common line are not significantly different from each other.

Concentrations are in µg/L except for magnesium and sulfate which are in mg/L.

Variables that were similar amongst all sites (not significantly different) have not been included in the table. Sample sizes: RP1, n = 5; GC2, n = 3; 009C, n = 5; and MCUS, n = 6.

df = degrees of freedom, *F* = F-ratio, *P* = significance

Conclusions

The current routine suite of water quality analytes for Magela Creek comprises Mg, Ca and SO_4 as the major ions, with Al, Cu, Fe, Pb, Mn, U and Zn as the measured trace elements (Supervising Scientist 2002). The current suite clearly includes all of the potential 'risk' metals/solutes identified above, as well as some additional ones, namely Al, Pb and Zn. It should be noted that these latter three metals continue to be analysed primarily for quality control purposes, rather than for environmental impact assessment because they provide sensitive markers of sample contamination during collection or in the chemical analysis laboratory. A high value for either one or all of these three metals provides a warning that the rigorous (clean) procedures involved in collection or handling a sample for trace metal analysis have been compromised.

Steps for completion

Results from this study provide a high degree of confidence that the routine water quality and bioaccumulation sampling programs conducted by SSD are not omitting any potential metals that could be of concern from either toxicological or bioaccumulation perspectives. A full trace metal profile (as described above) of relevant mine waters and upstream and downstream sites in Magela Creek will be conducted at least once per wet season in future. This will provide a quality control check to ensure that all mine-related metals that might (now or in the future) pose a risk to the receiving waterways are included in the routine monitoring suite.

References

ERA 2005. Water Management Plan 2005–2006. Company Report.

Hollingsworth I, Overall R & Puhalovich A 2005. Status of the Ranger irrigation areas – Final Report. Report to ERA.

Klessa DA 2000. *The chemistry of Magela Creek: A baseline for assessing change downstream of Ranger*. Supervising Scientist Report 151, Supervising Scientist, Darwin.

Klessa DA 2001a. A review of groundwater chemistry monitoring data at Ranger. Internal Report 363, Supervising Scientist, Darwin. Unpublished paper.

Klessa DA 2001b. Water quality in Magela Creek upstream and downstream of Ranger: A summary of performance for 2000–2001 and derived triggers and limits for 2001–2002. Internal Report 380, Supervising Scientist, Darwin. Unpublished paper.

Milnes AR & Fazey PG 1988. *Acid leaching of uranium from ore stockpiles and waste dumps in the Ranger Project area, East Jabiru*. Technical paper no 2 to Ranger Uranium Mines Pty Ltd.

Office of the Supervising Scientist 2002. Review of water standards (historic document). Internal Report 397, December, Supervising Scientist, Darwin. Unpublished paper.

Supervising Scientist 2002. Supervising Scientist Monitoring Program: Instigating an environmental monitoring program to protect aquatic ecosystems and humans from possible mining impacts in the Alligator Rivers Region. www.environment.gov.au/ssd/monitoring/pubs/env-mon-prog-background.pdf (accessed 24 April 2009)

Results from the routine stream monitoring program in Magela Creek catchment, 2008–09

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 2008–09 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 94–104, 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) have been developed under the SSD research program.

The results from the stream chemical and biological monitoring program for 2008–09 are summarised below. ^{226}Radium activity concentrations in Magela Creek for 2008–09 are reported separately (see 'Surface water radiological monitoring in the vicinity of Ranger and Jabiluka, pp 54–57, in this volume).

Chemical and physical monitoring

J Brazier

Routine weekly sampling program in Magela Creek

An overview of the water quality objectives for Magela Creek and the measures of success applied to meeting those objectives is provided in Iles (2004).

The 2008–09 wet season is the first time that water quality grab sampling and continuous monitoring have been co-located. This action was approved by ARRTC and the MTC following detailed statistical analysis (Brazier et al 2009) of the time series data collected at both the upstream (reference) and downstream locations. This analysis concluded that there would be no loss in power to detect potential impact by co-location

The first water chemistry samples for the 2008–09 wet season surface water monitoring program were collected from Magela Creek on 26 November 2008, immediately after commencement of surface flow. Weekly sampling continued throughout the season with the last samples collected on 10 June 2009. On 16 June 2009, MTC stakeholders agreed that continuous surface flow had ceased in Magela Creek and monitoring of the creek was no longer required.

On 11 February 2009, a value for electrical conductivity (EC) of 45 µS/cm was measured in the grab sample collected from the downstream site (Figure 1). This exceeded the statistically derived guideline value of 43 µS/cm, and corresponded with elevated magnesium (3.3 mg/L) and sulfate (12.2 mg/L). The continuous monitoring data (Figure 2) showed that the value of 45 µS/cm corresponded with the peak of an EC event that lasted 30 hours and for which EC remained above 43 µS/cm for 5 hours.

SSD considers that the pulse of magnesium and sulfate originated from RP1 (via Coonjimba Billabong). It is likely that an increase in flow (and water level) in Magela Creek that occurred on 8–9 February had initially restricted flow from Coonjimba Billabong. As the Magela Creek water level dropped between 9 and 11 February, water held back in Coonjimba Billabong drained out causing the increase in EC at the downstream site (Figure 2).

Ecotoxicological research conducted by SSD suggests that there was no detrimental environmental impact from this short-lived event (see 'Effects of magnesium pulse exposure on aquatic organisms, pp 27–31, in this volume).

On 18 February, uranium was approximately 6% of the limit (0.37 µg/L) at SSD's downstream site compared with 0.028 µg/L at the upstream site (Figure 3). This concentration is similar to uranium concentrations measured by the creekside field toxicity monitoring program on two occasions in 2002–2003, and once in the 2006–2007 wet season. On each of these occasions, field toxicity monitoring (including the in situ test conducted 16–20 February 2009) showed no detectable biological effects (as expected, noting that the ecotoxicologically-derived guideline value for U is 6 µg/L).

The routine grab sample collected on 18 March 2009 coincided with another higher EC event at the downstream site (Figure 1). The values of EC, magnesium and sulfate measured in this sample were 44 µS/cm, 3 mg/L and 10 mg/L, respectively. Continuous monitoring data (Figure 2) showed that this event peaked at 47 µS/cm and lasted about 20 hours, with EC exceeding 43 µS/cm for 8 hours. There had been increased discharge in Magela Creek during the previous day (from increased rainfall in the catchment) and the resultant water level decrease on 18 March would have led to increased drainage from Coonjimba Billabong back into Magela Creek, hence explaining the increase in EC.

From mid-April, typical end-of-wet-season trends were apparent as the water level decreased. Manganese concentrations at the downstream site increased as groundwater influences started to dominate, and electrical conductivity between the upstream and downstream sites became similar as minesite influences decreased.

Overall, the data from the continuous and water quality grab sample monitoring programs indicate that water quality in Magela Creek was comparable with previous seasons (for the west channel). Figure 4 shows that uranium concentrations for the 2008–09 wet season were comparable with previous seasons for the downstream west channel environment.

The results from the in situ toxicity monitoring program using freshwater snails (see later in this paper) provided reassurance that the aquatic environment of Magela Creek remained protected from activities at the Ranger mine.

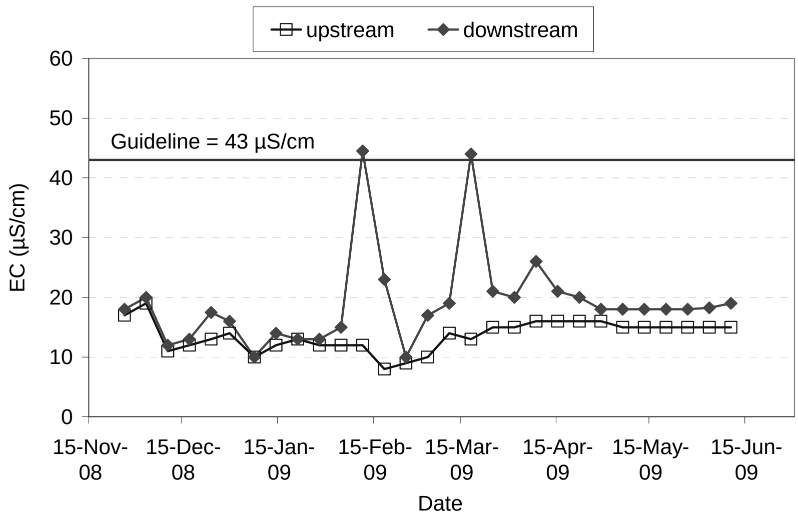


Figure 1 Electrical conductivity measurements in Magela Creek (SSD data) between November 2008 and June 2009

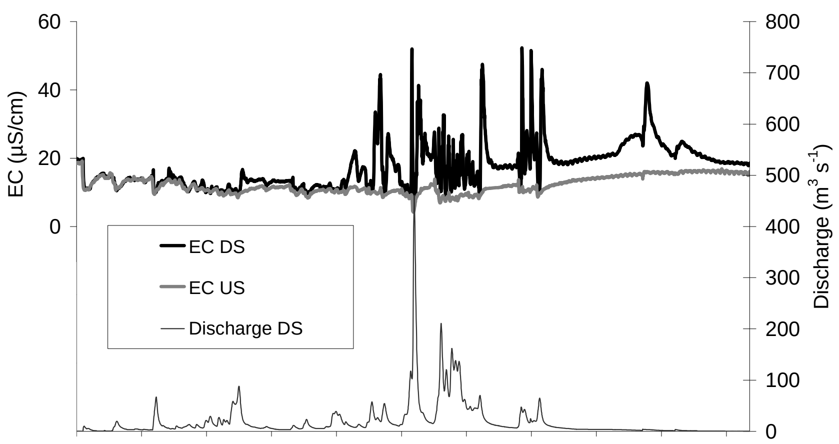


Figure 2 Electrical conductivity and discharge measurements in Magela Creek between December 2008 and April 2009 – continuous monitoring data

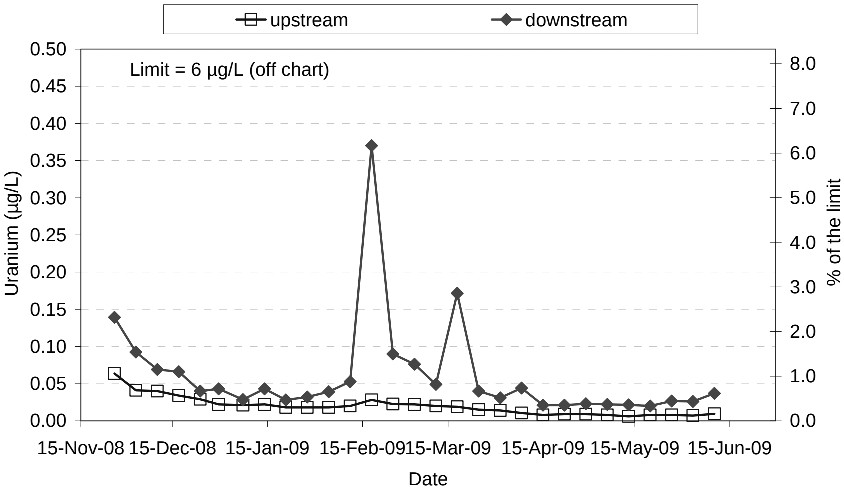


Figure 3 Uranium concentrations measured in Magela Creek by SSD between November 2008 and June 2009

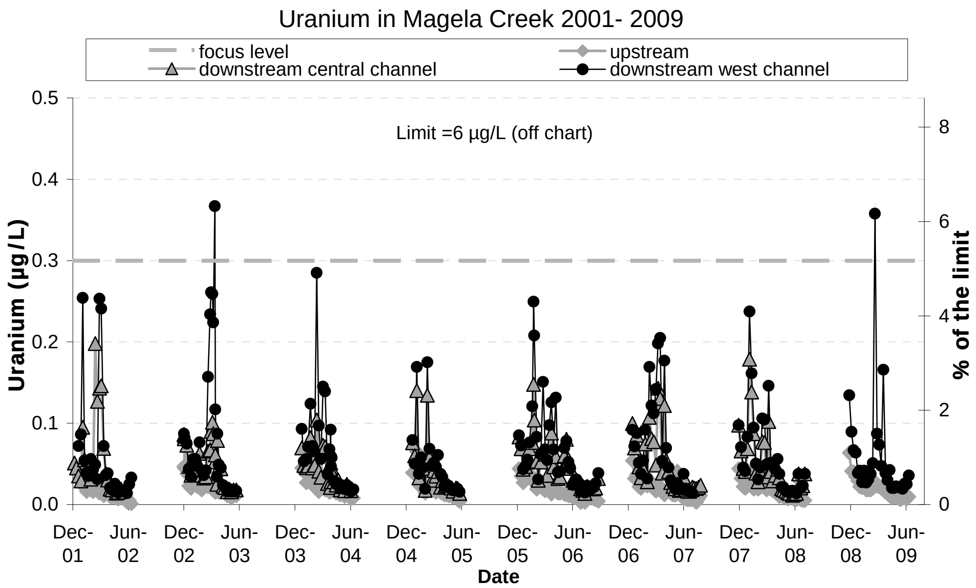


Figure 4 Uranium concentrations in Magela Creek between 2001 and 2009 (SSD data)

Chemical and physical monitoring of Gulungul Creek

Weekly grab sampling for routine analysis of water chemistry variables at the upstream site was discontinued for the 2008–09 wet season because this site does not represent a useful reference location for the Gulungul catchment. Water chemistry data measured at this site indicate that upstream (natural) catchment influences compromise its effectiveness for assessing downstream impacts from the mine. Weekly grab sample monitoring continued at the downstream site. Continuous monitoring of flow, electrical conductivity (EC) and turbidity was maintained at both the downstream and upstream sites.

The first water chemistry samples for SSD's 2008–09 wet season surface water monitoring program were collected from Gulungul Creek on 30 December 2008, immediately after commencement of surface flow. Weekly sampling continued throughout the season with the last samples collected on 20 May 2009. On 22 May 2009, MTC stakeholders agreed that continuous surface flow had ceased in Gulungul Creek and monitoring of the creek was no longer required.

There was considerable work carried out during the 2008 dry season on raising the embankment height of the tailings storage facility (TSF), using substantial quantities of waste rock sourced from the southern waste rock dump. Water run-off from this waste rock may have contributed to the observed elevations in EC (Figures 5 & 6), uranium (Figure 7) and sulfate concentrations at the Gulungul Creek downstream site compared with concentrations observed in the recent past few years. Discharge in Gulungul Creek (Figure 6) was also lower than previous years due to less rainfall in the catchment, and hence dilution of solutes would also have been less compared with previous years.

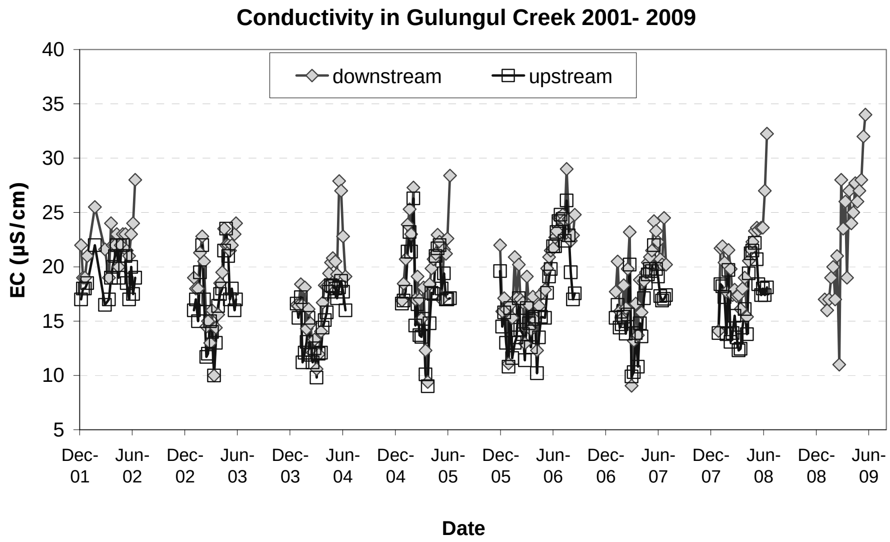


Figure 5 Electrical conductivity measurements in Gulungul Creek for the 2008–09 wet season

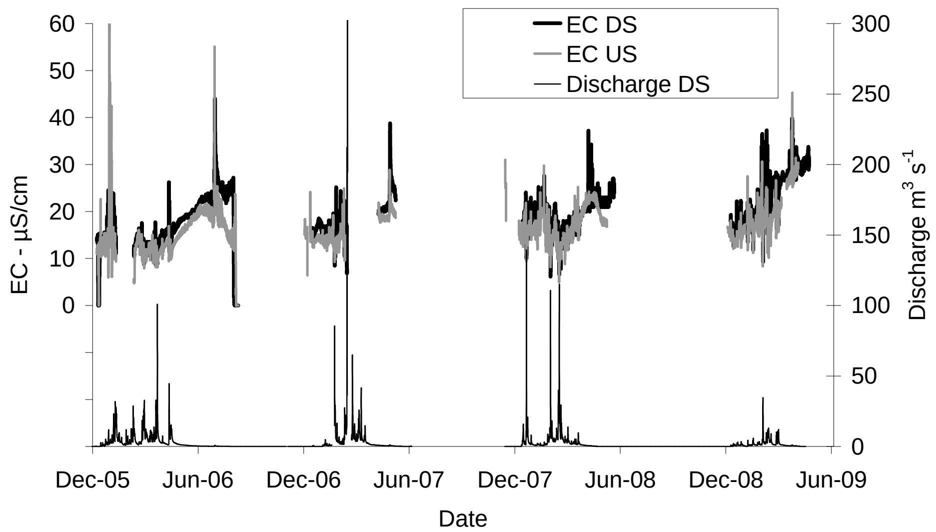


Figure 6 Electrical conductivity and discharge in Gulungul Creek 2005–2009 – continuous monitoring

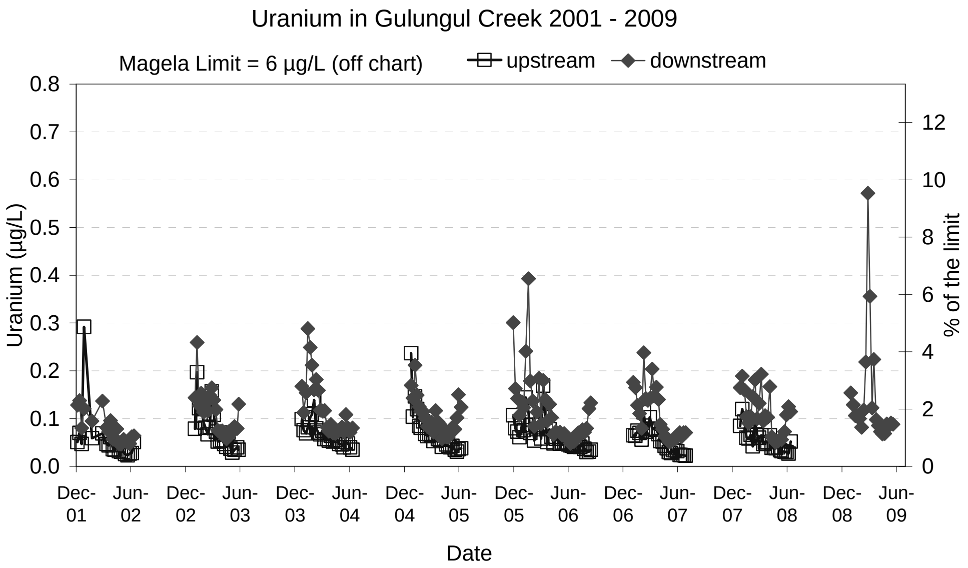


Figure 7 Uranium concentrations in Gulungul Creek between 2000 and 2009 (SSD data)

On 25 February 2009, a uranium value of 0.57 µg/L (Figure 8) measured at the downstream site (<10% of the Magela Creek limit) coincided with slightly elevated electrical conductivity (28 µS/cm) and sulfate concentration (2.7 mg/L). On 4 March 2009, uranium measured 0.36 µg/L at the downstream site (Figure 8), again coinciding with slightly elevated EC (24 µS/cm) and sulfate concentration (2.7 mg/L). Since the results of biological monitoring of macroinvertebrates in Gulungul Creek during recessional flows (see later in this paper) show no evidence of impact, and the values of the chemical variables are less than the guidelines and limits set for Magela Creek, it is considered that none of these excursions are environmentally significant.

After mid March 2009, uranium decreased to concentrations less than 0.2 µg/L (< 2% of the limit) which is similar to previous season's measurements (Figure 7).

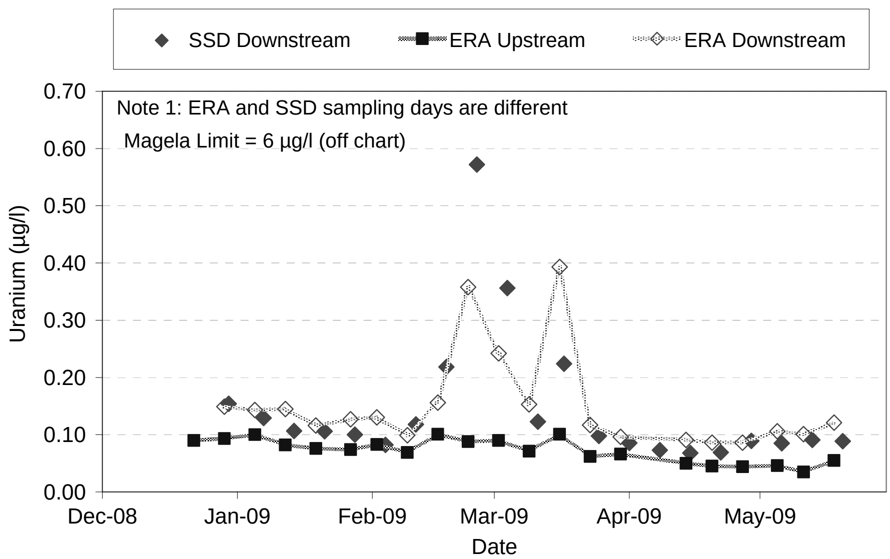


Figure 8 Uranium concentrations measured in Gulungul Creek by SSD and ERA during the 2008–09 wet season

From early April, recessional flow characteristics became apparent with electrical conductivity at the upstream and downstream sites becoming more similar and manganese concentrations increasing as groundwater inputs started to dominate.

Overall, the water quality of Gulungul Creek suggests that the aquatic environment in the creek remained protected from activities at the Ranger mine for the 2008–2009 season.

References

Brazier J, Humphrey C & Buckle D 2009.Future of the weekly water grab sampling program in Magela Creek catchment. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 66–70.

Iles M 2004. Water quality objectives for Magela creek – revised November 2004. Internal Report 489, December, Supervising Scientist, Darwin. Unpublished paper.

Toxicity monitoring in Magela Creek

C Humphrey, C Davies & D Buckle

In this form of monitoring, effects of water from the Ranger minesite on receiving waters are evaluated using responses of aquatic animals exposed to creek water. The response indicator has been reproduction (egg production) in freshwater snails, *Amerianna cumingi*, with each test running over a four-day exposure period.

For wet seasons between 1990–91 and 2007–08, toxicity monitoring was carried out using the ‘creekside’ methodology, in which a continuous flow of water from the adjacent Magela Creek was pumped through tanks containing test animals located under a shelter on the creek bank. There were a number of practical constraints with this method, including high staff demands, reliance on complex powered pumping systems (in an area of high electrical storm activity) and vulnerability to extreme flood events. These constraints led to a rigorous evaluation of the viability of an in situ testing technique whereby floating containers are deployed directly in the creek. This method offered the potential of substantially lower staffing, infrastructure and maintenance requirements. Humphrey et al (2009a) describe in detail the results of the two-year creekside versus in situ comparative assessment that demonstrated that the in situ technique is technically robust and constitutes an appropriate replacement for the creekside methodology. During the 2008–09 wet season, in situ toxicity monitoring was undertaken for the first time as the sole toxicity monitoring procedure.

Nine in situ toxicity tests were conducted on a fortnightly basis (ie every other week) over the 2008–09 wet season. The first test commenced on 4 December 2008 and the final test for the season commenced on 30 March 2009. Snail egg production at upstream and downstream sites was generally similar across all nine tests (Figure 1A) and the pattern of egg production across all tests was similar to that observed in previous wet seasons. Importantly, the mean upstream-downstream difference value across the nine wet season tests plots around the running mean (since 1991–92 wet season, Figure 1B) while individual difference values (Figure 1A) are within the maximum and minimum values recorded over this time series (full dataset not shown here).

Improvements to the statistical analysis of toxicity monitoring data using Analysis Of Variance (ANOVA) testing were described in Humphrey et al (2009b). The most important of the ANOVA factors tests for differences in the upstream-downstream difference values between two time periods – in this case and, in particular, test results for the current (2008–09) wet season versus all pre-2008–09 test data. No significant difference was found between the 2008–09 data and data from previous wet seasons ($p = 0.886$), confirming the visual assessment made from the graphical results. From these results it is concluded that no adverse effects on freshwater snails from inputs of Ranger minesite waters to Magela Creek occurred during the 2008–09 wet season.

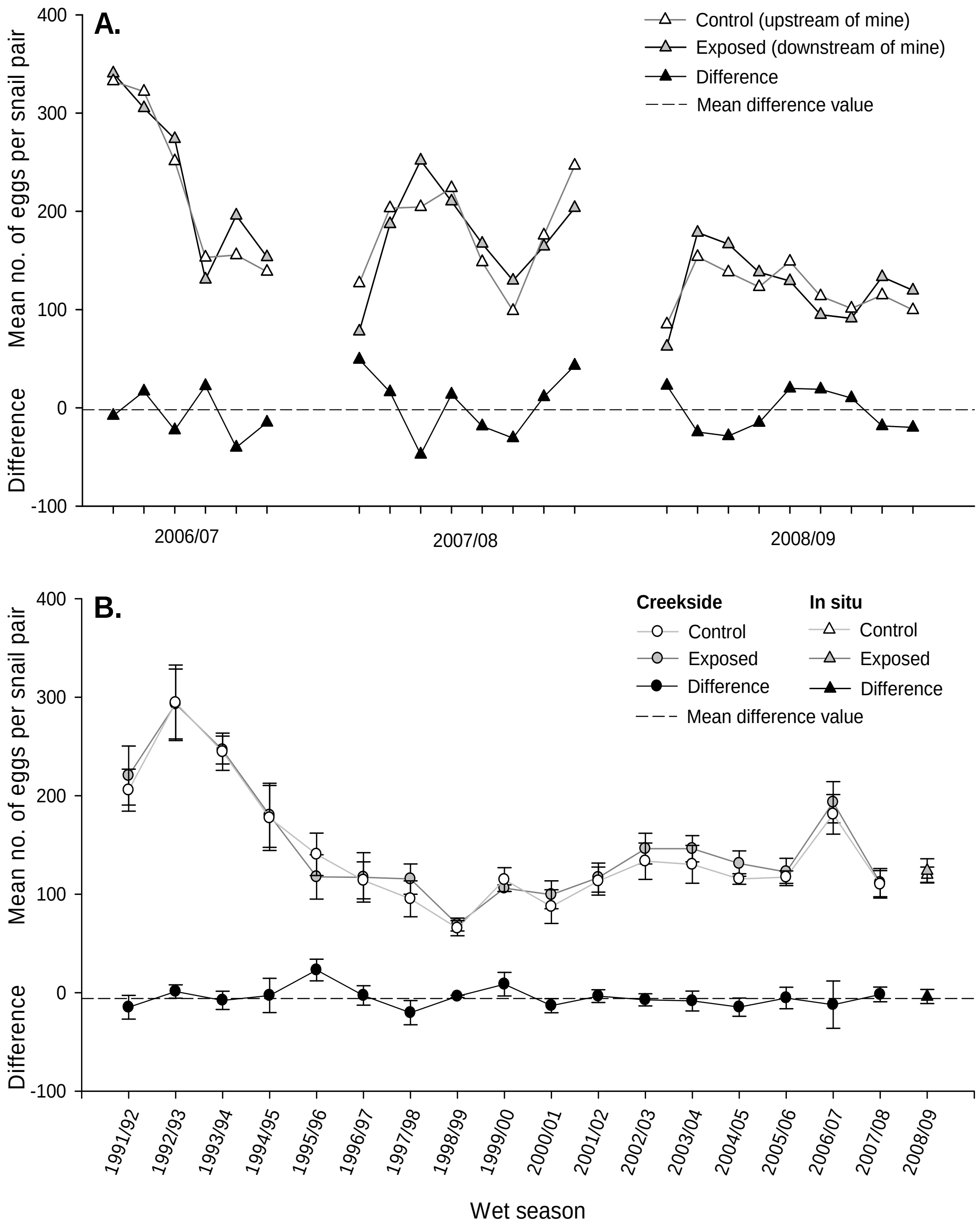


Figure 1 A In situ toxicity monitoring results for freshwater snail egg production for past three wet seasons. B. Toxicity monitoring results for the entire period between 1992 and 2009. Error bars represent standard errors about the mean.

References

Humphrey C, Buckle D & Davies C 2009a. Development of in situ toxicity monitoring methods for Magela Creek. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 86–90.

Humphrey C, Davies C & Buckle D 2009b. Toxicity monitoring in Magela Creek. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 51–52.

Bioaccumulation of uranium and radium in freshwater mussels from Mudginberri Billabong

A Bollhöfer, C Humphrey, B Ryan & D Buckle

Mudginberri Billabong is the first major permanent waterbody downstream (12 km) of 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 mine-derived inputs to Magela Creek must remain within acceptable levels. Enhanced body burdens of mine-derived solutes in biota could also potentially reach limits that may harm the organisms themselves as well as provide early warning of bioavailability of metals and radionuclides. Hence the bioaccumulation monitoring program serves an ecosystem protection role in addition to the human health aspect.

Mussel bioaccumulation data were obtained intermittently by SSD from Mudginberri Billabong from 1980 to 2001. From 2002 onwards, there has been regular (annual) sampling from Mudginberri Billabong and a control site in the nearby Nourlangie catchment (Sandy Billabong). Only data from 2000 onwards (where methods were standardised and control sites included) will be discussed in this paper. The data gathered prior to 2000 have been presented and discussed in the relevant and respective SSD annual reports.

Uranium concentrations in freshwater mussels, water and sediment samples collected from Mudginberri and Sandy Billabongs are shown in Figure 1. The mean concentrations of uranium in mussels from both Mudginberri and Sandy Billabongs are very similar from 2000 onwards, with no evidence of an increasing trend in concentration in Mudginberri mussels over time.

The lack of any increase in concentration of U in mussel tissues through time, with essentially constant levels observed between 1989 and 1995 (as reported in previous reports), and consistently low levels from 2000 to the last sample taken in October 2008, indicates absence of any significant mining influence.

Concentrations of Ra in mussels are age-dependent (Figure 2) and also appear to be related to growth rates, seasonal soft body weights, water chemistry and sediment characteristics (Brazier et al 2009, also see ‘A study of radionuclide and metal uptake in mussels from Mudginberri Billabong’, pp 99–104, in this volume. A longitudinal study along the Magela Creek catchment conducted in 2007 measuring uptake of radium and uranium in mussels showed that radium uptake was largely due to natural catchment influences rather than a mining-related feature (Brazier et al 2009).

The average annual committed effective doses calculated for a 10-year old child who eats 2 kg of mussel flesh, based upon average concentrations of ^{226}Ra and ^{210}Pb from Mudginberri Billabong mussels collected between 2000 and 2008 is approximately 0.2 mSv. The average for Sandy Billabong mussels collected between 2002 and 2008 is approximately 0.1 mSv.

The generally consistent relationship between mussel age and Ra concentration for each billabong (Figure 2) currently provides a robust baseline against which any future mine-related change in Ra concentrations can be detected.

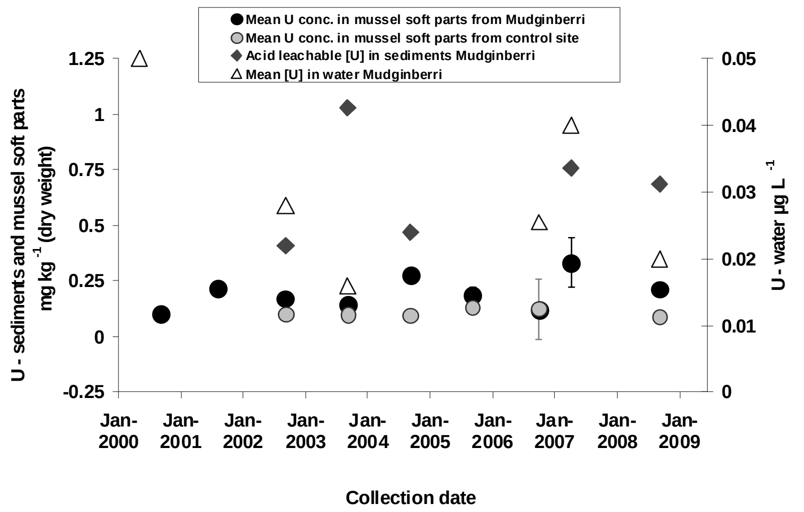


Figure 1 Mean concentrations of uranium measured in mussel soft-parts, sediment and water samples collected from Mudginberri Billabong and Sandy Billabong since 2000

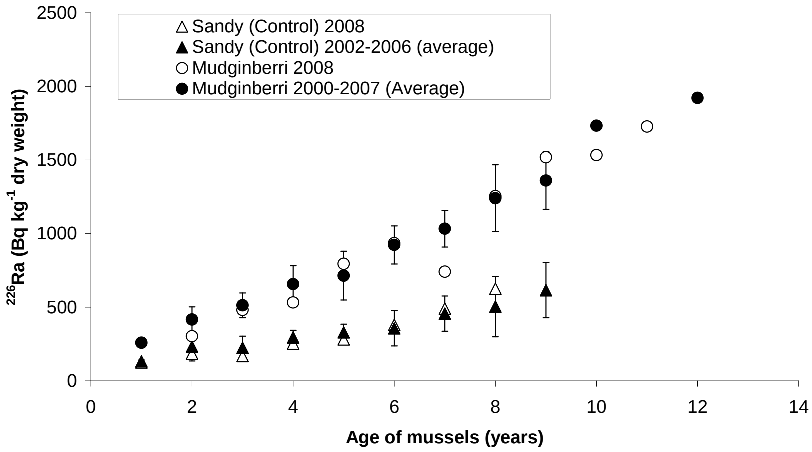


Figure 2 ^{226}Ra activity concentrations in the dried flesh of freshwater mussels collected from Mudginberri Billabong 2000–2008 and Sandy Billabong 2002–2008. Mussels were not collected from Sandy Billabong in 2007. The error bars are ± 1 standard deviation.

In October 2008, a longitudinal study of radium and uranium uptake in mussels in Mudginberri Billabong was undertaken to determine if the location of sampling in the billabong had a significant effect on the levels of Ra and U measured in mussels. The findings from this work are presented in ‘A study of radionuclide and metal uptake in mussels from Mudginberri Billabong’, pp 99–104, in this volume.

References

Brazier J, Bollhöfer A, Humphrey C & Ryan B 2009. A longitudinal study of radionuclide and metal uptake in mussels from Magela Creek and Mudginberri Billabong. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 91–97.

Monitoring using macroinvertebrate community structure

C Humphrey, L Chandler & C Camilleri

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 Supervising Scientist 2004). The design is now a balanced one comprising upstream and downstream sites at two 'exposed' streams (Gulungul and Magela Creeks) and two control streams (Burdulba and Nourlangie Creeks).

Samples were 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, dissimilarity indices are calculated. These indices are a measure of the extent to which macroinvertebrate communities of the two sites differ from one another. A value of 'zero' indicates macroinvertebrate communities identical in structure while a value of 'one' indicates totally dissimilar communities, sharing no common taxa.

Disturbed sites, including those impacted by activities other than mining, may be associated with significantly higher dissimilarity values compared with undisturbed sites. Compilation of the full macroinvertebrate dataset from 1988 to 2009 has been completed with results shown in Figure 1. This figure plots the paired-site dissimilarity values using family-level (log-transformed) data, for the two 'exposed' streams and the two 'control' streams.

Improvements to the presentation and statistical analysis of macroinvertebrate data were described in Humphrey et al (2009). Multi-factor ANOVA can be used to test whether or not macroinvertebrate community structure has altered significantly at the exposed sites for the recent wet season of interest, using dissimilarity values derived for each of the five possible randomly-paired upstream and downstream replicates. Only data gathered since 1998 have been used for this analysis. Data gathered prior to this time were based upon different and less rigorous sampling and sample processing methods, and/or the absence of any sampling in three of the four streams. (Sampling in Gulungul Creek and the control streams only commenced in 1994.)

Inferences that may be drawn from the data shown in Figure 1 are weakened because there are no 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; fixed) and Site (nested within CI; random) showed no significant difference (in dissimilarity) between the control and exposed streams from earlier years (back to 1998) compared with those from 2009 (ie the BA x CI interaction is not significant). While the Year x Site (BA CI) interaction is significant in the same analysis ($p = 0.011$), 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.

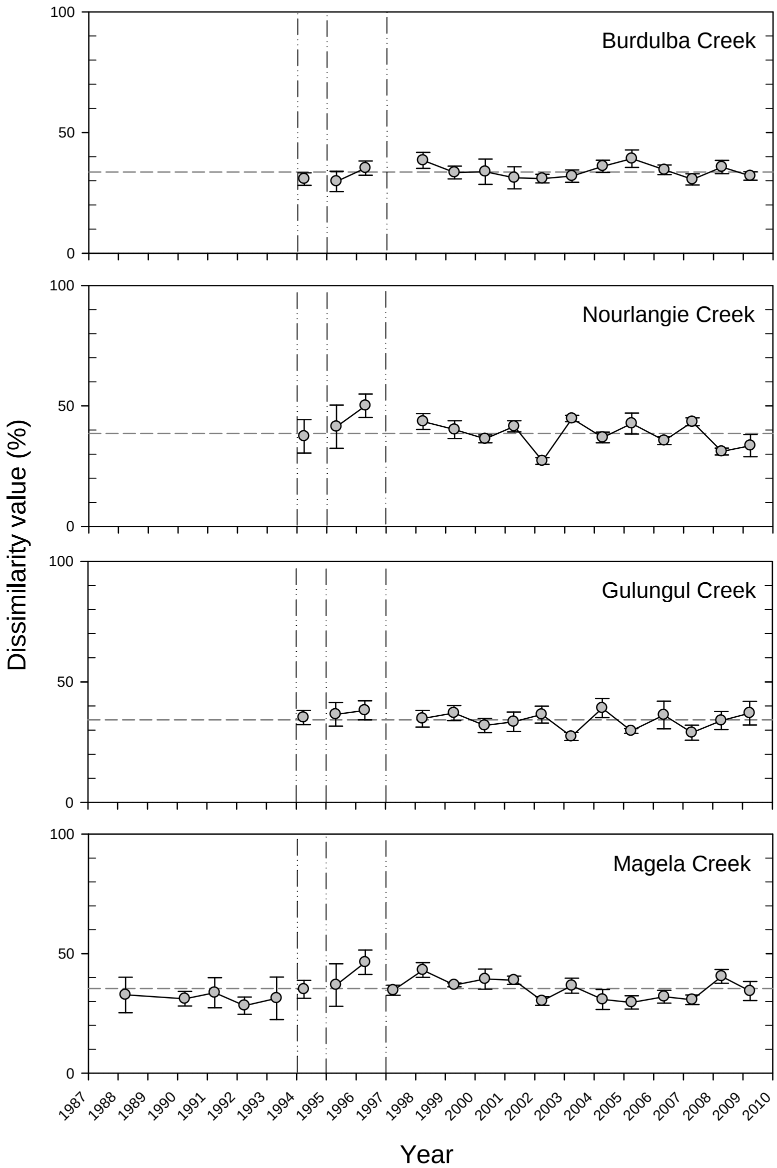


Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of Ranger mine for the period 1988 to 2009. The dashed vertical lines delineate periods for which a different sampling 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.

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. Samples close to one another in the ordination space indicate a similar community structure. Figure 2 depicts the ordination derived using the *pooled* (average) within-site macroinvertebrate data (unlike the replicate data used to construct the dissimilarity plots in Figure 1). Data points are displayed in terms of the sites sampled in Magela and Gulungul Creeks downstream of Ranger for each year of study (to 2009), relative to Magela and Gulungul Creek upstream (control) sites for 2009, and all other control sites sampled up to 2009 (Magela and Gulungul upstream sites, all sites in Burdulba and Nourlangie). Because the data-points associated with these two sites are generally interspersed among the points representing the control sites, this indicates that these 'exposed' sites have macroinvertebrate communities that are similar to those occurring at control sites. This was verified using ANOSIM (ANalysis Of SIMilarity, effectively an analogue of the univariate ANOVA) testing, to determine if exposed sites (Magela and Gulungul downstream) are significantly different from control sites in multivariate space. ANOSIM conducted on pooled (within-site) data from all years to 2009 showed no significant separation of exposed and control sites for the respective comparisons (P>0.05).

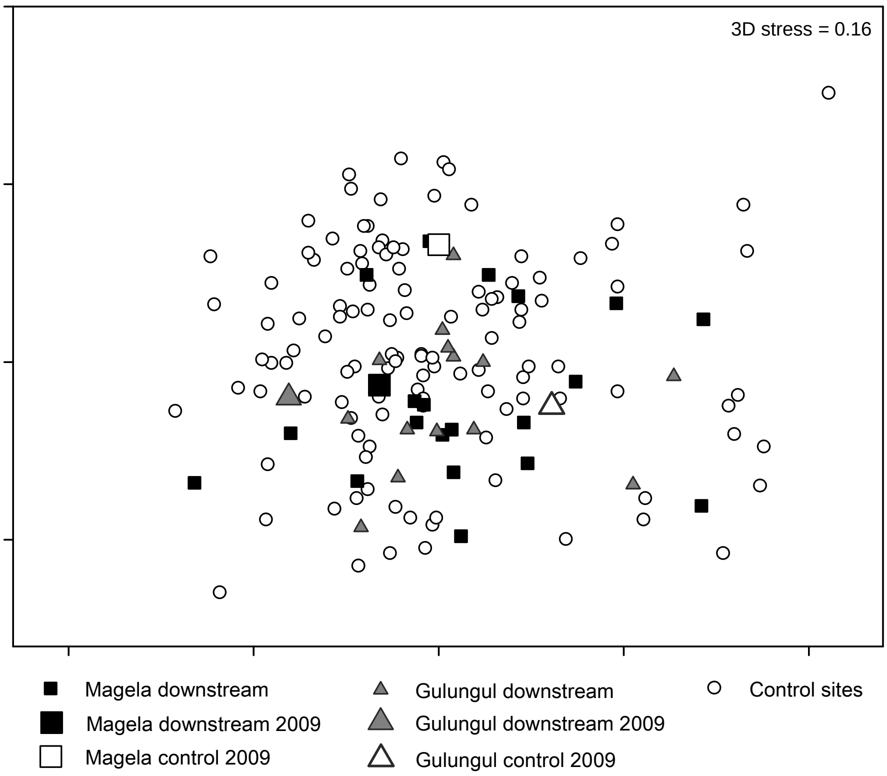


Figure 2 Ordination plot of macroinvertebrate community structure data from sites sampled in several streams in the vicinity of Ranger mine for the period 1988 to 2009. Data from Magela and Gulungul Creeks for 2009 are indicated by the enlarged symbols.

Collectively, these graphical and statistical results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining in the period 1994 to 2009 have not adversely affected macroinvertebrate communities.

References

Humphrey C, Chandler L & Hanley J 2009. Monitoring using macroinvertebrate community structure. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 57–59.

Supervising Scientist 2004. *Annual Report 2003–2004*. Supervising Scientist, Darwin.

Monitoring using fish community structure

D Buckle, C Humphrey & C Davies

Assessment of fish communities in billabongs is conducted between late April and July each sampling year. Data are gathered using non-destructive sampling methods from 'exposed' and 'control' sites in deep channel billabongs annually, and shallow lowland billabongs dominated by aquatic plants, biennially (every other year). Details of the sampling methods and sites were provided in the 2003–04 Supervising Scientist annual report (Supervising Scientist 2004, chapter 2, section 2.2.3). These programs were reviewed in October 2006 and the refinements to their design detailed in Buckle and Humphrey (2008, 2009; shallow and channel billabong fish communities respectively).

For both deep channel and shallow lowland billabongs, comparisons are made between a directly-exposed billabong (Mudginberri) in the Magela Creek catchment downstream of Ranger mine versus control billabongs from an independent catchment (Nourlangie Creek and Wirnmuyurr Creek). 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 85–87, in this volume). A significant change or trend in the dissimilarity values over time could imply mining impact.

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 2009 is shown in Figure 1.

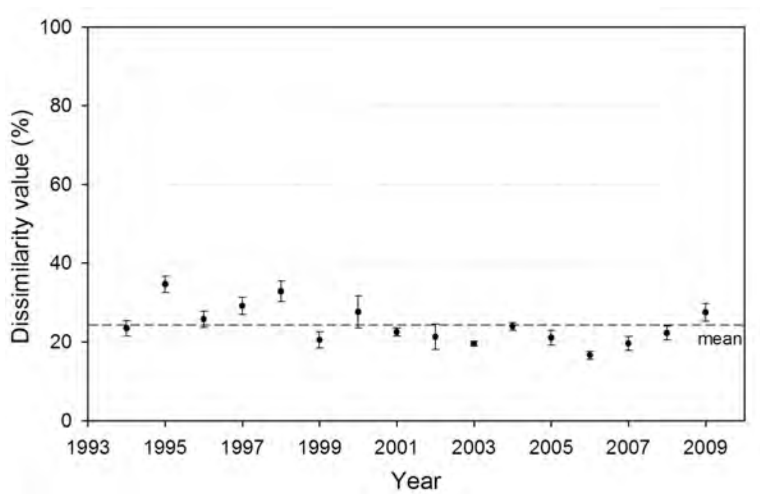


Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

In the 2003–04 Supervising Scientist annual report, a decline in paired-site dissimilarity measures over time was noted. This decline in dissimilarity remains significant over the full dataset (1994 to 2009, $P < 0.001$), despite the increase in dissimilarity that has occurred since 2006 (Figure 1). In Buckle and Humphrey (2009), a change in method procedure between the visual canoe (1989–2000) to the visual boat (2001–present) was identified as an issue that required closer scrutiny in the context of the changes seen in the dissimilarity index. This change potentially confounds the observed decline over time due to a significant increase in the time taken to complete each replicate visual count since 2001. The increase in transect times (since 2001) corresponds with a significant step down in the community dissimilarity value, which could potentially explain the overall decline over time. Increased transect times have typically resulted in increased observations of the more cryptic species. Theoretically, this could alter the paired-site community dissimilarity values between the two billabongs, particularly where observations of these less common species are more pronounced in one of the billabongs relative to the other billabong. To date, however, it has not been possible to characterise and quantify the effect, if any, that altered transect times have had on the community dissimilarity values. This is largely due to the complexities of changes in species abundances and their cumulative influence over the dissimilarity value over time.

Notwithstanding, the dissimilarity observed in 2009 (the highest recorded since the introduction of the visual boat in 2001) has occurred without change in sampling method, suggesting that transect times accompanying the change in observation method may not be so influential in determining dissimilarity values. The paired-site fish community dissimilarity value has increased since 2006 and may suggest that natural shifts in community structure over time are occurring. If this is the case, the nature of the community shift should become more evident over the next few years, leading to a possible explanation for the previously-identified decline or step down over time in community dissimilarity values.

In Humphrey et al (2006), the chequered rainbowfish (*Melanotaenia splendida inornata*) was identified as the species that has had most influence on the change in the paired-billabong dissimilarity value. This species, due to its habit, appears unaffected by differences in transect times coinciding with the change to the observation method (Buckle & Humphrey 2009) and as such, the abundances of this species may be regarded as reliable for the entire period that sampling has been conducted in Mudginberri Billabong, from 1989 to 2009. Chequered rainbowfish declined significantly in abundance after about 1996 with relatively low abundances sustained until 2008 (Figure 2) (Buckle & Humphrey 2009). The elevated abundance in rainbowfish in 2009 (Figure 2) provides insights as to the possible cause of population fluctuations, and by association, therefore, the possible cause of interannual changes to the paired-billabong dissimilarity values.

For example, one of the environmental correlates identified in the decline in rainbowfish between 1989 and 2008 is the increase in grasses, and in particular the exotic para grass (*Urochloa mutica*), on Magela floodplain (Humphrey et al (2006). These grasses are still expanding on the floodplain yet rainbowfish abundances in 2009 have returned to values akin to those observed pre-1996 (Figure 2), suggesting the habitat conditions on Magela floodplain (the recruitment source for these fishes) may not be overly important.

In both Humphrey et al (2006) and Buckle and Humphrey (2009) measures of wet season discharge in Magela Creek were identified as correlates of rainbowfish abundance. Rainbowfish abundance from 1989 to 2009 remains negatively correlated with wet season discharge (total monthly discharge in January ($p = 0.016$), February ($p = 0.016$) and the wet season total ($p = 0.029$)), supporting the suggestion that wet season intensity is a factor controlling the population numbers. Thus rainbowfish abundance is higher following wet seasons of relatively low rainfall (Figure 2).

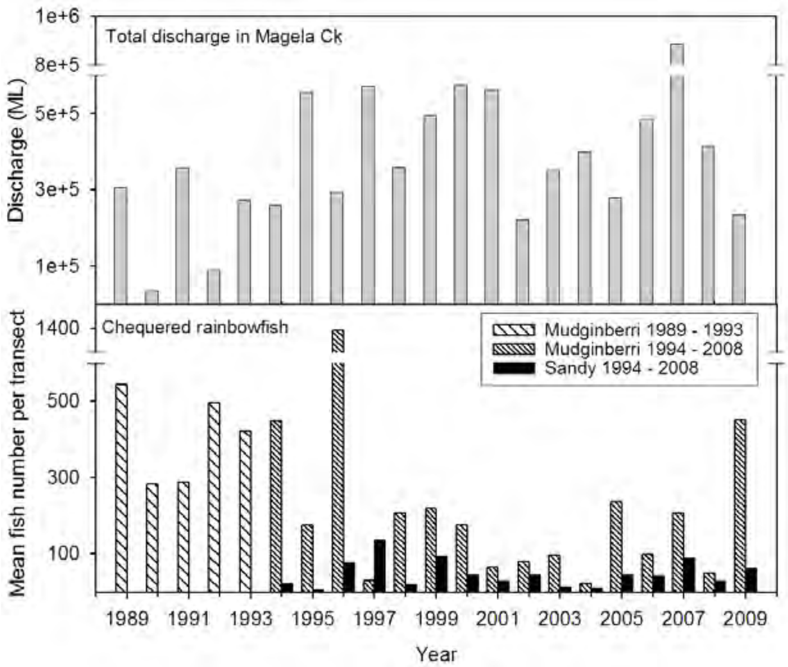


Figure 2 Relative abundance of chequered rainbowfish in Mudginberri and Sandy billabongs from 1989 to 2009 with associated total discharge in Magela Creek (gauging station G8210009)

A possible explanation for this relationship was proposed in Humphrey et al (2006). In field toxicity monitoring tests, larval rainbowfish have been observed to be relatively intolerant of naturally low solute (including nutrient) concentrations that characterise surface waters in wet seasons of high stream discharge. Another causal link may relate to the greater dispersion of fish in wet seasons of higher discharge. In wet seasons of low discharge, stimuli for migration (flood pulses) are reduced, which may lead to fish concentrating more in lowland channel billabongs (Buckle & Humphrey 2009).

Importantly, the abundance of rainbowfish does not appear to be related to any change in water quality over time as a consequence of water management practices at Ranger mine. The net input of magnesium (Mg) from Ranger has been used as a reasonably reliable surrogate measure of mine water inputs to Magela Creek (see Humphrey et al 2006 for further information). For the wet seasons over the period of record from 1988–89 to 2008–09, no significant relationship has been observed between the mine contribution of Mg and corresponding rainbowfish abundance in Mudginberri Billabong. This is not surprising as concentrations of U and Mg in Magela Creek arising from mine waste water discharges are at least two orders of magnitude lower than those known to adversely affect larval fishes including, in the case of uranium, chequered rainbowfish (Supervising Scientist 2004, chapter 3, section 3.4.1 & Humphrey et al 2006).

Shallow lowland billabongs

The last assessment of fish communities in shallow lowland billabongs (a biennial program) was conducted in May 2007 with results reported in Buckle and Humphrey (2008). This paper discusses results from sampling conducted in April and May 2009.

The monitoring program for fish communities in shallow billabongs is conducted biennially in six billabongs, comprising three 'control' versus 'exposed' billabong pairs. In a similar manner to fish communities in channel billabongs (discussed above), the similarity of fish communities in the directly exposed sites downstream of Ranger on Magela Creek (Georgetown, Coonjimba and Gulungul Billabongs) to those of the control sites (Sandy Swamp and Buba Billabongs on Nourlangie Creek and Wirnmuyurr Billabong – a Magela floodplain tributary) was determined using multivariate dissimilarity indices calculated for each sampling occasion. A plot of the dissimilarity values of the control-exposed site pairings – Coonjimba-Buba, Georgetown-Sandy Swamp and Gulungul-Wirnmuyurr Billabongs – from 1994 to the present, is shown in Figure 3.

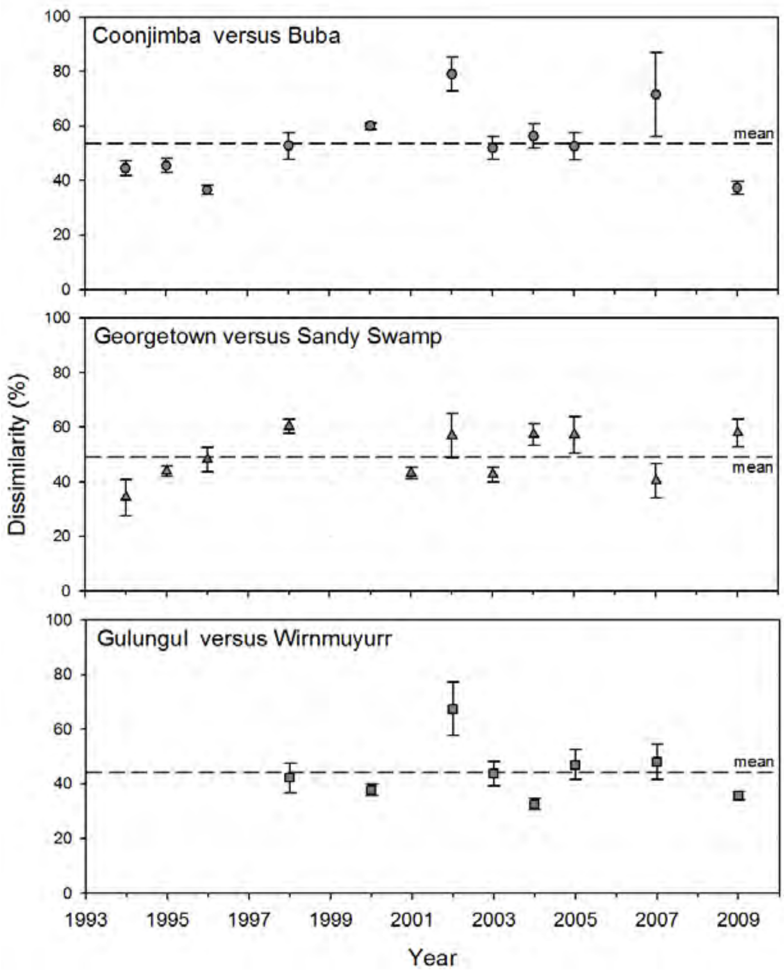


Figure 3 Paired control-exposed site dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in 'directly-exposed' Magela and 'control' Nourlangie and Magela Billabongs in the vicinity of Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of average trap enclosure data between the pairwise billabong comparisons, Coonjimba-Buba, Gulungul-Wirnmuyurr and Georgetown-Sandy Billabongs.

The paired-site dissimilarities shown in Figure 3 average between 40 and 60% indicating fish communities in each of the billabongs comprising a site pairing are quite different from one another. In Buckle and Humphrey (2008) it was identified that the particularly high dissimilarity values observed in the Coonjimba-Buba pairing for 2002 and 2007, and the Gulungul-Wirnmuyurr site pairing for 2002 (Figure 3) were attributable to high densities of particular aquatic plant types in one or both of the billabongs. Excessive plant densities are unfavourable for fish communities as fish movement, and hence residency, is physically prevented. The influence of aquatic plants on fish community structure is further supported by the slightly increased dissimilarity observed in the Georgetown-Sandy Swamp pairing in 2009. The increased dissimilarity appears to be related to an increase in the density of the emergent aquatic plant *Eleocharis* sp in Georgetown Billabong, combined with reduced plant density (dominated by emergent lilies), in Sandy Billabong. The divergence in aquatic plant habitats between the two billabongs appears to have resulted in reduced similarity (increased dissimilarity) in fish community structures between these locations (Figure 3).

In Buckle and Humphrey (2008), an increase over time was observed in the paired Coonjimba-Buba billabong dissimilarity values, irrespective of the removal of years 2002 and 2007 for which high values are associated with unusually high aquatic vegetation density in one or other of the billabongs (discussed above). The reduced dissimilarity found for 2009 has allayed concerns of increasing dissimilarity over time, as a weak relationship only is now present when the years 2002 and 2007 are included in data analysis ($p = 0.03$).

References

Buckle D & Humphrey C 2008. Monitoring of Ranger mine using fish community structure. In ***eriss*** *research summary 2006–2007*. eds Jones DR, Humphrey C, van Dam R & Webb A, Supervising Scientist Report 196, Supervising Scientist, Darwin NT, 56–59.

Buckle D & Humphrey C 2009. Monitoring of Ranger mine using fish community structure. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 60–63.

Humphrey C, Buckle D & Pidgeon R 2006. Fish communities in channel billabongs. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 48–53.

Supervising Scientist 2004. *Annual Report 2003–2004*. Supervising Scientist, Darwin.

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 2008–09 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, 2008–09', pp 74–92, this volume.

Tasks under Part (ii) are reported below where the following two summaries are provided:

1 enhancements to SSD's stream monitoring program for Ranger,

2 a study of radionuclide and metal uptake in mussels from Mudginberri Billabong.

Prior to reading these summaries, 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.

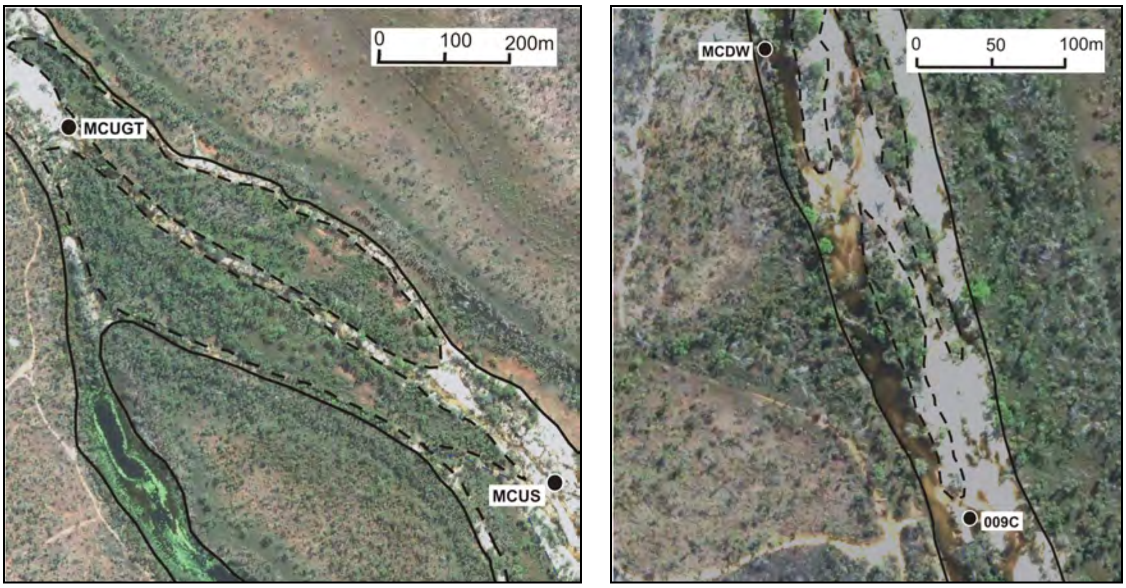
Enhancements to SSD's stream monitoring program for Ranger

J Brazier, C Humphrey, K Turner, D Jones & D Buckle

Ongoing optimisation of existing monitoring methods is one of the processes followed by SSD to ensure that best practice continues to be employed for detection of possible impacts arising from the Ranger mining operation. To this end, some significant changes were made to the Ranger stream monitoring program commencing in the 2008–09 wet season, as outlined below.

Relocation of sampling sites in Magela Creek

The key change made to the water quality monitoring program has been to relocate the Magela Creek sites at which weekly surface water chemistry grab samples have been historically collected. The upstream reference and downstream impacts detection sites, formerly MCUS and 009C (respectively), have been moved to be co-located with the continuous monitoring and in situ toxicity (biological) monitoring pontoon sites – MCUGT and MCDW, respectively (Figure 1). The reason for this change is to provide complete integration among the elements of SSD's water quality monitoring program and thereby reduce replication of effort and possible inconsistency of results between the different locations and monitoring methods. The MCDW downstream site provides a more sensitive location for detecting impacts from the minesite and thus complements rather than replicates the grab sample data produced by the compliance monitoring program carried out by Energy Resources Australia Ltd (ERA), and the check monitoring performed by the Department of Regional Development, Primary Industries, Fisheries and Resources (DRDPIFR).



A Upstream monitoring sites on Magela Creek

B Downstream monitoring sites on Magela Creek

Figure 1 Upstream and downstream monitoring sites used in the SSD's water chemistry (grab sampling and continuous) and toxicity monitoring programs. Channel boundaries are indicated by the continuous or broken (water-level-dependent) lines.

To examine the potential effect of changing the locations of the grab sampling sites on the ability of SSD's program to detect impacts from the minesite, chemical data gathered weekly from MCUGT and MCDW between the 2001 and 2008 wet seasons as part of the creekside toxicity monitoring program were compared with corresponding data collected from MCUS and 009C (the historical reference and compliance sites, respectively) as part of the routine grab sample monitoring program. Concentrations of the key analytes, magnesium, sulfate and uranium, were compared statistically between the sites using Analysis Of Variance testing (Brazier & Humphrey 2009).

The concentrations of the three analytes were shown to be statistically similar between the new upstream reference site (MCUGT) and the historical upstream reference site (MCUS) ($p>0.05$).

In contrast, the concentrations measured at the proposed new downstream site (MCDW) were found to be significantly higher ($p<0.05$), albeit by only a very small margin, than those from the compliance site (009C). This is because the compliance site is located in the central channel of Magela Creek while the new site is located in the west channel of Magela Creek. Contaminant levels downstream of Ranger have historically been higher in the west channel compared with the central channel, particularly in relation to discharge events emanating from Ranger Retention Pond 1 (RP1). Water released from RP1 enters Coonjimba Billabong, which eventually drains into the west side of Magela Creek. Results obtained for electrical conductivity (EC) from continuous and grab sample EC monitoring programs in previous years show that water from RP1 mixes incompletely across the west channel and preferentially follows the western bank, particularly during low flow periods.

While the concentrations measured at the MCDW location are statistically higher than values at the compliance site 009C further upstream, the actual magnitude of the difference is only minor, and is not regarded as being sufficient to compromise any assessment of the significance of inputs from the minesite, compared with the use of the 009C location for this purpose. Indeed, sampling in the west channel at the location of the current continuous monitoring and toxicity monitoring will, if anything, result in a more conservative assessment of the contribution of the Ranger mine to solutes in Magela Creek.

Other changes to SSD's weekly grab sampling program in Magela Creek

Commencing with the 2008–09 wet season, physicochemical parameters such as EC, turbidity and pH are being measured in the field only. This change in procedure has been made following several years of good agreement between concurrent field and laboratory measurements, demonstrating that it is possible to obtain reliable measurements in the field with well-calibrated instruments equipped with probes optimised for use in very low EC media.

To provide a further integrity check on the field measurement, the field technician is now comparing the readings taken from the field meter with those being recorded at the same time by the continuous monitoring sonde (data are remotely accessible in the laboratory). If there is good agreement (allowing for known systematic offsets in the continuous readouts), then the field measurement is recorded as valid and reported to stakeholders. If there is disagreement (ie the difference between the two measurements is outside of pre-determined tolerances), then a backup sample of water that was also collected in the field is checked in the laboratory. During the 2008–09 wet season, out-of-tolerance differences between the in situ and field probe measurements occurred on only three occasions. If the discrepancy is attributable to the field measurement, then the continuous monitoring value is reported. If the continuous monitoring measurement is deemed to be inaccurate, then the field technician will report the concern to the continuous monitoring team to allow them to correct any issues. In all three cases that occurred in the 2008–09 wet season, the lack of agreement between the continuous monitoring and field meter occurred with pH for the lower ionic strength waters from the upstream control site in Magela Creek.

The research emphasis for the water quality monitoring program during the 2008–09 wet season was placed on event-based sampling to capture episodes of elevated EC (ie higher inputs of solutes from the minesite). The data produced by this targeted program of sampling are currently being analysed to determine if there is a functional correlation between EC and U at higher EC values. If such a relationship is found then it may be possible to use this to infer U concentrations from the continuous EC trace during events of elevated EC.

Due to the remote location of the continuous monitoring autosamplers, there is often a time lag (up to 1 week) between sample collection and the physical retrieval of the sample from the field (and subsequent filtration and processing in the laboratory). Alterations to the chemical characteristics of water samples may occur when they are sitting for extended periods, which can lead to loss of dissolved metals (ie in the <0.45 µm fraction) by binding to particulate matter or sample bottle walls. To assess how the composition of Magela Creek water changes over time, a desk top study will be conducted using historical water quality monitoring data and more recent data acquired from samples collected using the autosamplers. The key objective of this study is to investigate how the dissolved (<0.45 µm) U in a sample changes over time and as a function of turbidity (suspended sediment concentration).

The results from the data analysis project described above will provide the basis for determining if event-based sampling (using the continuous monitoring system and autosamplers) may be able to provide a reliable measure of dissolved concentrations of U during periods of inputs of elevated levels of solutes from the minesite.

Changes to the weekly grab sampling program in Gulungul Creek

Weekly grab sampling for routine analysis of water chemistry variables was discontinued at the upstream location commencing with the 2008–09 wet season. This is because this site may be influenced by upstream (natural) uraniferous catchment effects that compromise its effectiveness for assessing downstream impacts from the mine. Weekly monitoring has continued at the downstream site (GCH). Continuous monitoring of EC and turbidity is being maintained at both the downstream and upstream sites.

Use of in situ testing for ongoing toxicity monitoring

As reported in the Supervising Scientist Annual Report for 2007–08 (section 3.2), a comparative assessment was made between the results from two methods for toxicity monitoring: firstly, creekside monitoring conducted for 17 wet seasons since 1992, and secondly, in situ testing that has been trialled for the past three wet seasons. Both methods of toxicity monitoring use the number of eggs produced by the freshwater snail (*Amerianna cumingi*) as the test endpoint.

Creekside monitoring, which involves pumping a continuous flow of water from the creek through tanks containing test animals located under shelters on the creek's bank, has much higher staff and infrastructure resourcing needs than in situ testing. Comparative testing of the two methods was conducted over two wet seasons. The results showed greater snail egg production in the in situ method compared with the creekside method but no differences in the upstream-downstream difference values (the critical comparative measure and response end-point) between the two methods. This finding led to the decision to replace creekside with in situ toxicity monitoring, commencing in the 2008–09 wet season.

The in situ toxicity method is providing a more environmentally-relevant testing regime compared with the creekside procedure, whilst requiring substantially reduced staff time and eliminating the need for maintenance-intensive, complex infrastructure.

Continuous monitoring intranet reporting

An automated intranet reporting system was developed prior to the start of the 2008–09 wet season to enable daily upload of continuous data collected from all the SSD continuous monitoring sites (Magela Creek, Gulungul Creek, Ngarradj Creek, Georgetown Billabong, Ranger mine trial landform) to the Department’s intranet for immediate assessment by SSD staff. Quality-assessed, validated data are uploaded to the intranet on a monthly basis. The data, which include EC, pH, turbidity, stage height, discharge and rainfall, are presented in the form of time-series plots enabling visual assessment of each parameter (Figure 2).

SSD’s continuous monitoring intranet reporting system has been upgraded during the 2009 dry season to ensure timely production, viewing and reporting of data for SSD staff for the 2009–10 wet season.

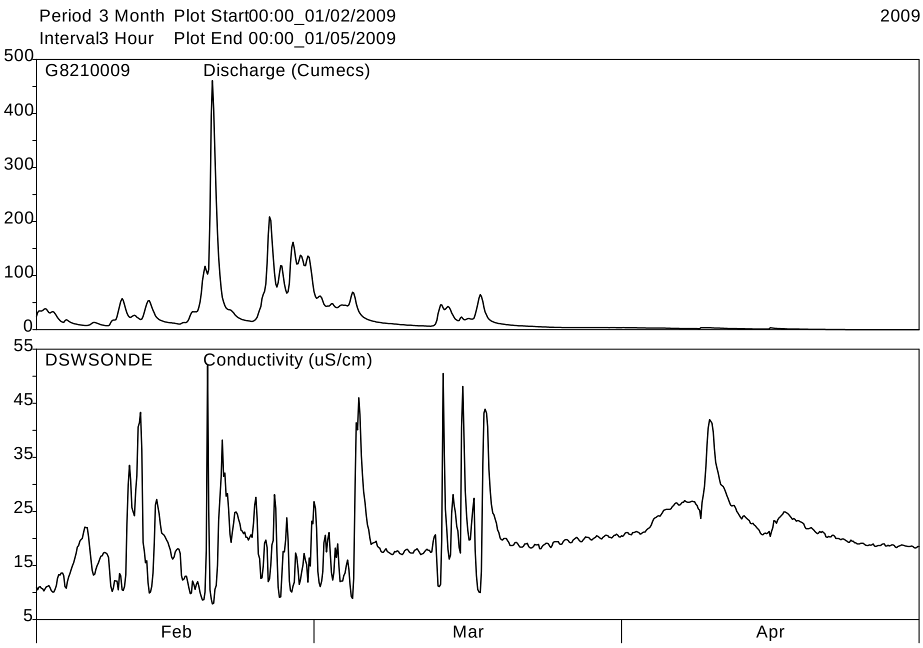


Figure 2 Time-series plot showing validated discharge and electrical conductivity data measured at the upstream site on Magela Creek between February and April 2009

Reference

Brazier J & Humphrey C 2009. Ranger stream monitoring program: relocation of surface water chemistry grab monitoring sites in Magela Creek. Internal Report 563, June, Supervising Scientist, Darwin. Unpublished Paper.

A study of radionuclide and metal uptake in mussels from Mudginberri Billabong

A Bollhöfer, C Humphrey, B Ryan & D Buckle

Background

An important component of the stream monitoring program for Ranger mine measures uptake of selected metals and radionuclides by freshwater mussels, *Velesunio angasi*, from Mudginberri Billabong. Among the suite of radionuclides measured, radium-226 (^{226}Ra) is of particular relevance as ^{226}Ra in mussels has been identified as the major contributor to the total radiological dose from ingestion of bush foods by local indigenous people (Martin et al 1998). There are several factors contributing to this: (a) freshwater mussels are an integral component of the diet of the Mudginberri Aboriginal community located downstream of the mine; (b) the high concentration factor of 19 000 for radium in freshwater mussels (Johnston 1987); and (c) the large ingestion dose coefficient for ^{226}Ra of 0.28 µSv/Bq (ICRP 1996).

During the 22nd ARRTC meeting (October 2008), results were reported from a longitudinal study of radionuclide and metal uptake in mussels from upstream of the mine down to Mudginberri Billabong in the Magela Creek catchment, a total distance of about 30 km. The study was designed to test the hypothesis that Ranger mine was not contributing to the higher radium activity concentrations found in mussels from Mudginberri Billabong compared with concentrations found in mussels from Sandy Billabong, a control site in another (adjacent) catchment. The study showed that radium and metal body burdens in freshwater mussels along the Magela catchment are driven by a number of factors such as mussel growth rate and (soft) body weight, as well as natural water chemistry gradients, particularly Ca and Mg, along the catchment that are unrelated to current mining activity at Ranger. Three observations led to this conclusion: (i) uranium concentrations in the mussels from sites in the longitudinal study were comparable with pre-mining values from 1980; (ii) ^{228}Ra/^{226}Ra activity ratios in mussel flesh decrease gradually along the catchment; and (iii) stable lead isotope ratios in mussel flesh and sediments change gradually as well, rather than a step change which would be expected for a contemporary (point source) mining-related impact (Bollhöfer et al 2010).

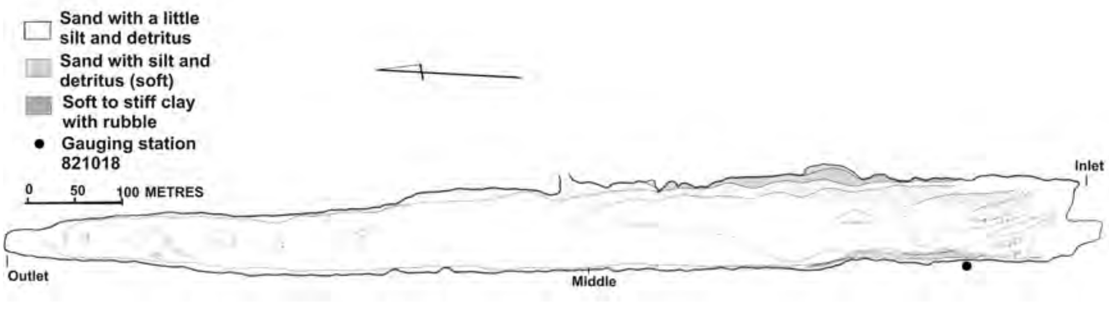


Figure 1 Mudginberri Billabong and location of 2008 sampling sites

To test whether the sampling location and associated variability in the amount of fine sediments have an influence on radium activity and metal concentrations in freshwater mussels from Mudginberri Billabong, mussels were collected for metal, stable lead isotope and radionuclide analyses from the inlet, middle and outlet of the billabong (Figure 3). Sampling occurred at the end of the 2008 dry season. The billabong edges were sampled at the three locations, as the edges are where mussels are typically concentrated and able to be harvested by local Aboriginal people.

Methods

Mussels were collected using a dredge and placed into acid-washed containers holding water collected from the billabong. Surface water samples were collected at the time of mussel collection in acid-washed plastic containers. In addition, between 300 and 400 g of the top 3–5 cm layer of sediment that the mussels were associated with was collected from each site using an Ekman grab. The sediment was placed into zip lock plastic bags and taken to the laboratory for processing.

After collection, mussels were transported to the SSD Darwin laboratories and purged for 6–7 days in billabong water before being measured for weight, length and width, and dissected to remove the flesh. Samples were freeze-dried to determine the dry weight. The age of each mussel was determined by counting the number of annual growth bands (annuli). The dried and ground flesh of each mussel was combined by age class and site, and the average dry weight per age class determined. A composite sample of each age class was cast in epoxy resin for determination of radioisotopes of radium (^{226}Ra & ^{228}Ra), lead (^{210}Pb), and thorium (^{228}Th) by gamma spectrometry. Those mussels less than 1 year of age, or in an age class with insufficient total mass (< 2 g) for analysis by gamma spectrometry, were analysed by alpha spectrometry. Aliquots of the samples were also sent for acid digestion and ICP-MS analysis of lead isotopes and heavy metals.

Results

Radionuclides in mussels

Each mussel age class was measured for the radioisotopes of lead (^{210}Pb), thorium (^{228}Th) and radium (^{226}Ra & ^{228}Ra) by gamma spectrometry and alpha spectrometry, respectively. Figure 2 shows the average ^{226}Ra activity concentration and the ^{228}Ra/^{226}Ra activity ratio per mussel age class for the recent 2008 collection (inlet, middle and outlet values shown separately) compared with data from previous end of dry season collections. ^{226}Ra activity concentrations in mussels collected in 2008 are comparable with activity concentrations determined previously.

^{226}Ra and ^{210}Pb activity concentrations are positively correlated with age, indicating bioaccumulation of these radionuclides, with the ^{226}Ra-age relationship shown in Figure 4. Differences in radionuclide activity concentrations in mussels amongst the three sites were tested, using analysis of covariance (ANCOVA) which, after taking age into account, tests for differences in regression intercepts and slopes. There was no statistically significant difference in the mussel ^{226}Ra activity concentrations (P=0.45) and there was a small but statistically insignificant difference in the Ra-age regression slopes (P=0.061) among locations, with the middle location exhibiting a slightly larger rate of increase in ^{226}Ra activity concentration with mussel age. There was no difference in the ^{210}Pb activity concentrations (P=0.67) nor the ^{210}Pb-age regression slope (P=0.16) amongst locations, respectively.

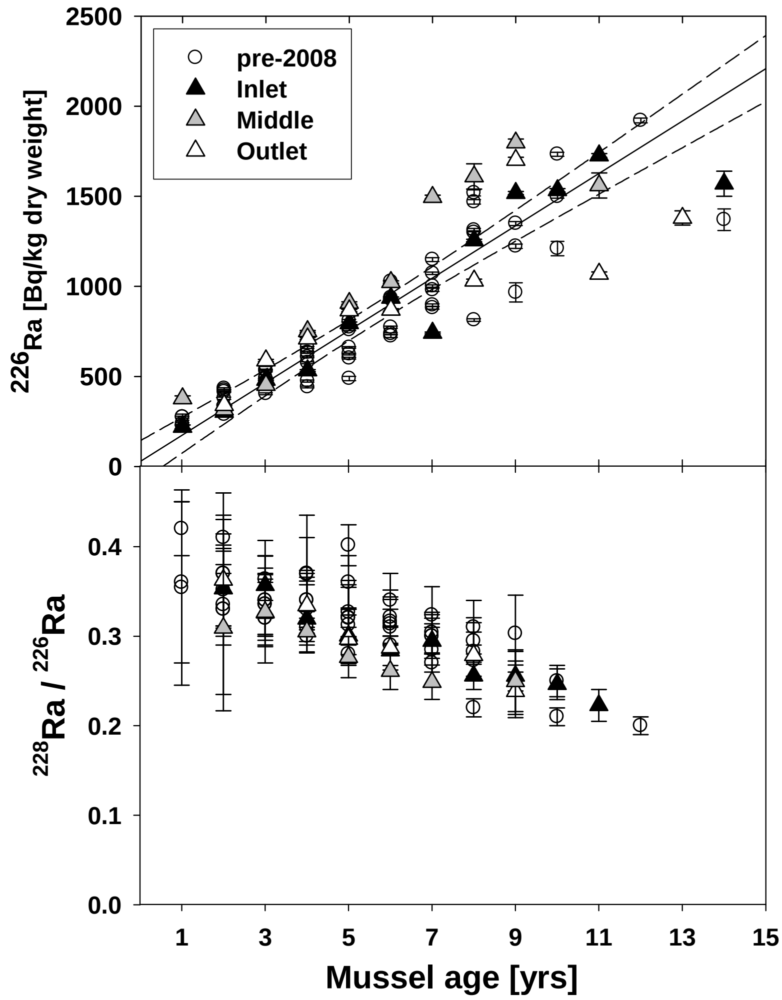


Figure 2 ^{226}Ra activity concentrations (top) and ^{228}Ra/^{226}Ra activity ratios (bottom) measured in mussels collected in 2008, and a comparison with results from previous end of the dry season collections (open circles). The solid line in the activity plot is a linear fit to all of the pre-2008 data, and the dashed lines show the associated 95% confidence limits for this dataset.

^{226}Ra is a member of the uranium decay series and ^{228}Ra of the thorium decay series. Hence the activity ratio of the two radioisotopes provides a measure of the relative contribution of uranium and thorium-rich sources, respectively, to the radium activity concentration in a sample. The lower the ^{228}Ra/^{226}Ra activity ratio is in sediments or mussels, the lower the relative contribution of radium derived from a thorium-rich source and the higher the contribution of radium derived from a uranium rich source, respectively. ANCOVA testing of the ^{228}Ra/^{226}Ra activity ratio measured in age-determined mussels (Figure 4) shows that after taking age into account, there is a significant difference in the ^{228}Ra/^{226}Ra ratio ($P=0.035$) amongst sites, with mussels collected in the middle exhibiting lower ratios than mussels from the inlet and outlet of the billabong, respectively.

Even though there are differences in the ^{228}Ra/^{226}Ra ratio measured in the mussels at the three locations, location in the billabong has no measurable effect on the ^{226}Ra and ^{210}Pb activity concentrations in freshwater mussels. Consequently, differences that have been observed over the years for ^{226}Ra and ^{210}Pb data for mussels obtained from a number of sampling locations in Mudginberri Billabong must be the result of other factors, such as the timing of mussel collection (wet versus dry season) or the duration and intensity of the preceding wet season.

Uranium and stable lead in mussels

Uranium and lead concentrations in water, whole sediment, in the < 63 µm sediment fraction (mud and clays) and in dried mussel flesh combined from each age class, were measured by inductively coupled plasma mass spectrometry.

Both uranium and lead concentrations measured in mussel flesh are positively correlated with age. ANCOVA combined with a Tukey's multiple comparison testing showed that this increase with age is highest in mussels from the middle section of the billabong ($P<0.05$), whereas mussels from the inlet and outlet show similar values. This site difference in concentrations cannot be explained by mussel condition (relative body weights), which, in agreement with earlier studies from the early 1980s, are highest at the inlet and gradually decrease towards the outlet (Figure 3) (Humphrey & Simpson 1985).

Measurements of the concentration of uranium and lead in water collected at the three sites showed no difference amongst the sites. However, the concentrations of metals in whole sediment were generally higher in the middle, as a consequence of the higher proportion of mud and clays (65%) found there compared with the inlet (29%) and outlet (13%) sampling locations in the billabong. It appears that the higher proportion of fines may be the cause of the higher metal concentrations observed in mussel tissue from the middle of the billabong.

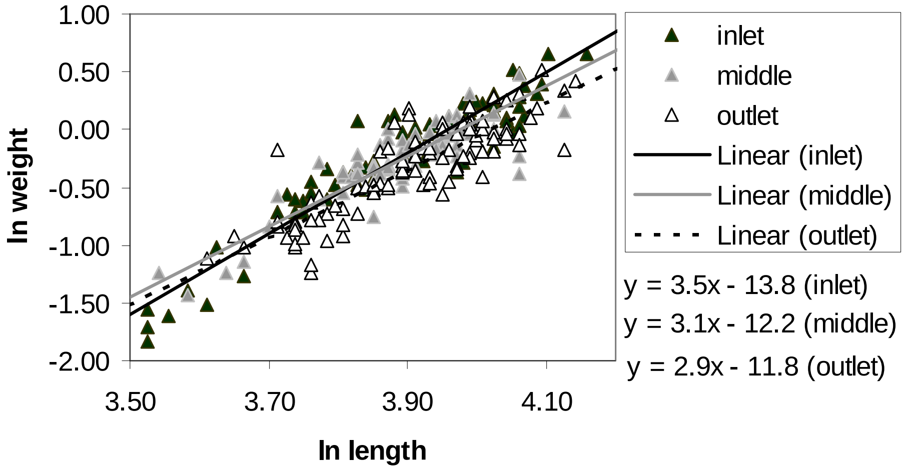


Figure 3 Mussel dry weight ($\log_e$ transformed) plotted against ($\log_e$ transformed) shell length. The slope is lowest at the outlet, indicating poorest mussel condition, and highest at the inlet.

^{206}Pb and ^{207}Pb are the stable end-members of the uranium decay series (^{238}U and ^{235}U, respectively), while ^{208}Pb is the stable end-member of thorium decay (^{232}Th). In Figure 4 the $^{206}Pb/^{207}Pb$ isotope ratios measured in mussel tissue are plotted against the $^{208}Pb/^{207}Pb$ ratio and a comparison is made with data from the 2007 longitudinal study and from a previous collection in 2005. This method enables the determination of the relative contribution of different sources to the total lead concentration in a sample.

Common lead isotope signatures are, for example, the PDAC (Present Day Average Crustal) ($^{206}Pb/^{207}Pb \approx 1.20$ and $^{208}Pb/^{207}Pb \approx 2.48$) or the Broken Hill and Mt Isa lead isotope signatures ($^{206}Pb/^{207}Pb < 1.04$ and $^{208}Pb/^{207}Pb < 2.32$), respectively (Doe 1970). Broken Hill and Mt Isa lead has been used in Australia and worldwide for many decades for the manufacturing of industrial lead products, and contamination with this type of lead can be traced via its low $^{206}Pb/^{207}Pb$ and $^{208}Pb/^{207}Pb$ isotopic fingerprint (Bollhöfer & Rosman 2000, 2001). In contrast, high $^{206}Pb/^{207}Pb$ and low $^{208}Pb/^{207}Pb$ ratios indicate a contribution from a uranium rich source, whereas high $^{208}Pb/^{207}Pb$ indicates a thorium rich source. As lead isotopes are physically and chemically alike and not discriminated by environmental processes, varying proportions of lead from different sources in a biological sample can be directly related to the total lead isotopic composition seen in the sample.

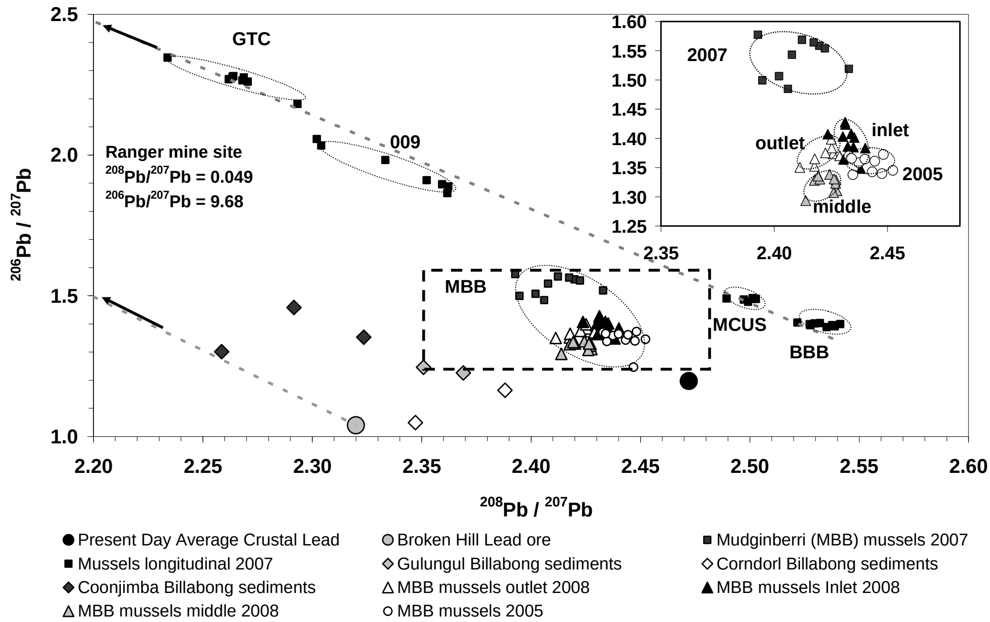


Figure 4 $^{206}Pb/^{207}Pb$ plotted against $^{208}Pb/^{207}Pb$ isotope ratios measured in mussel tissue from Mudginberri Billabong, and previous data from Magela Creek. Trendlines (dashed) assume mixing of radiogenic lead with the Ranger ore signature and lead with an Upper Magela catchment signature and Broken Hill lead, respectively. Each site's mussel lead isotope signature is circled with the site label. BBB: Bowerbird Billabong, MCUS: Magela Creek upstream; GTC: Georgetown confluence; G8210009: Magela Creek downstream; MBB: Mudginberri Billabong (Supervising Scientist 2007). The inset shows a magnification of the dashed rectangle depicting the Mudginberri Billabong data only.

Figure 4 illustrates that there are within-billabong variations in the lead isotope ratios measured in mussel flesh, although these differences are much less pronounced than those observed along Magela Creek for the 2007 longitudinal study. While there are only small differences in Pb isotope ratios in mussels between the two end of dry season collections from the inlet (October 2005 and October 2008, respectively), mussels collected in May 2007 exhibit a more uraniferous signature. This is most likely caused by a difference in sampling location. Due to accessibility issues at the end of the wet season, mussels were sampled further upstream, closer to the Magela Creek channel in May 2007, and hence the lead isotope ratios are more similar to those measured at G8210009. In contrast, the sampling site in 2008 was influenced to a much greater extent by billabong mud and clays, which typically show Pb isotope ratios closer to PDAC.

Lead isotope signatures measured in mussels collected in 2008 from the outlet lie in between those of the inlet and middle isotope ratios (Figure 4). Mussels collected from the middle appear to be influenced by an additional (industrial) source that exhibits lead isotope ratios similar to those found in Corndorl Billabong sediments, a location unaffected by runoff from the Ranger minesite. This is indicative of contamination with lead from sources such as lead shot, fishing sinkers, or other manufactured products containing Pb from commercial Australian orebodies. The middle sampling site is closest to (and directly downstream of) the boat ramp and would also be the site most exposed to runoff from the Mudginberri community as well as from the adjacent paddocks of the historic Mudginberri pastoral station. Together, these factors would place this site at highest risk of contamination from industrial lead.

Conclusion

This study found subtle variations in the relative contribution of sources of lead and uranium in the tissue of freshwater mussels collected from within Mudginberri Billabong. The concentrations of these metals in mussels appear to be mainly influenced by the proportion of fine sediment (< 63 µm) at the sampling site. The lead isotope ratios indicate an additional industrial source of lead, in the middle or western edge of the billabong.

Importantly, ^{226}Ra and ^{210}Pb activity concentrations in mussels (which determine most of the dose received via the ingestion of mussels) are not statistically different amongst sites. Thus the site of collection of mussels in Mudginberri Billabong is unlikely to affect the levels of ^{226}Ra and ^{210}Pb measured for the purposes of conducting a dose assessment. The results provide increased confidence that the data from previous mussel collections conducted from several locations in the billabong over the years can be directly compared, if factors that affect mussel condition, such as timing of mussel collection and the duration and intensity of the preceding wet season, are taken into account.

References

Bollhöfer A & Rosman KJR 2000. Isotopic source signatures for atmospheric lead: The Southern Hemisphere. *Geochim. Cosmochim. Acta* 64, 3251–3262.

Bollhöfer A & Rosman KJR 2001. Isotopic source signatures for atmospheric lead: The Northern Hemisphere. *Geochim. Cosmochim. Acta* 65, 1727–1740.

Bollhöfer A, Brazier J, Ryan B, Humphrey C & Esparon A 2010. A study of radium bioaccumulation in freshwater mussels, *Velesunio angasi*, in the Magela Creek catchment, Northern Territory, Australia. submitted to the *Journal of Environmental Radioactivity*

Brazier J & Humphrey C 2009. Ranger stream monitoring program: relocation of surface water chemistry grab monitoring sites in Magela Creek. Internal Report 563, June, Supervising Scientist, Darwin. Unpublished Paper.

Doe BR 1970. *Lead isotopes*. Springer-Verlag, Berlin, Germany.

Humphrey CL & Simpson RD 1985. The biology and ecology of *Velesunio Angasi* (Bivalvia: Hyriidae) in the Magel Creek, Northern Territory. Open File Record 38. Supervising Scientist for the Alligator Rivers Region.

ICRP 1996. Age-dependent doses to members of the public from the intake of radionuclides: part 5. Compilation of Ingestion and Inhalation dose coefficients. ICRP Publication 72, Elsevier.

Johnston A 1987. *Radiation exposure of members of the public resulting from operations of the Ranger Uranium Mine.* Technical memorandum 20, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra.

Martin P, Hancock GJ, Johnston A & Murray AS 1998. Natural-series radionuclides in traditional north Australian Aboriginal foods. *Journal of Environmental Radioactivity* 40, 37–58.

Supervising Scientist 2007. *Annual Report 2006–2007*. Supervising Scientist, Darwin.

Part 2: Ranger – Rehabilitation

Define the geomorphic characteristics of Gulungul Creek catchment

D Moliere, K Turner, K Evans & M Saynor

Background

This project has three aims: (1) to develop reliable impact assessment methods for quantifying the loads of fine suspended sediment transported in Gulungul Creek during rainfall/runoff events; (2) to characterise the channel stability of the creek; and (3) to characterise the bedload movement in the creek upstream and downstream of Ranger. This project was last reported at ARRTC 24 and in Moliere et al (2009). Studies such as these in Gulungul Creek are assuming greater importance for baseline characterisation given the current (tailings dam lift) and proposed expansion of mine infrastructure by ERA into the catchment.

Progress

Suspended sediment loads for event data collected between 2003 and 2008 are shown in Moliere and Evans (2008).

Data for the 2008–09 wet season (rainfall, discharge, turbidity and electrical conductivity (EC)) were collected at the upstream (GCUS) and downstream (GCDS) monitoring stations in Gulungul Creek (see Map 2 for locations). Discharge and rainfall data for the upstream and downstream sites are shown in Figure 1 and Figure 2, respectively, and the EC and turbidity data for both sites are shown in Figure 3.

Rainfall and runoff data were also collected at the mid Gulungul Gauging Station (GS210012), which is located between GCUS and GCDS (Map2) . Bed material was collected from stream cross-sections as they were being surveyed during the 2008 dry season. Analysis and reporting of the bed material data have been delayed due to staff changes and competing priorities.

During the 2009 dry season, work was carried out at each of the Gulungul Creek continuous monitoring stations to improve the ease of use and security of the installed instrumentation. This work included installation of cable connectors and conduits for each of the sensors so that they can easily be removed for maintenance or calibration and to reduce possible damage due to passing debris. New sensor mounts were also constructed to provide a more secure location for mounting the various sensors at each site.

A thorough assessment has also been made of the performance of the backup EC sensors currently used at the Gulungul Creek sites. All of SSD's continuous monitoring sites in Gulungul and Magela Creeks have backup sensors in place to provide redundancy for these critical data. In Magela Creek the backup for the upstream and downstream sites is provided by a second multiparameter sonde. However, in Gulungul Creek the backup data are provided by individual probes connected to a central datalogger. This situation is the result of the historical evolution of the continuous monitoring systems in this creek.

It was found that the backup sensors have relatively low measurement repeatability, and measurement error of ±10% for EC values ≤ 500 µS/cm. The sensors are also only capable of outputting data at 1 μS/cm resolution, which is low compared with other EC sensors utilised by SSD which are capable of outputting data at 0.01 μS/cm resolution. These backup EC sensors have not been used as a primary source of EC data since 2007, when minisondes were installed as the primary sensors at GCUS and GCDS. The current backup EC sensors will be phased out of SSD's continuous monitoring program over the next two years, and replaced with probes with higher performance characteristics.

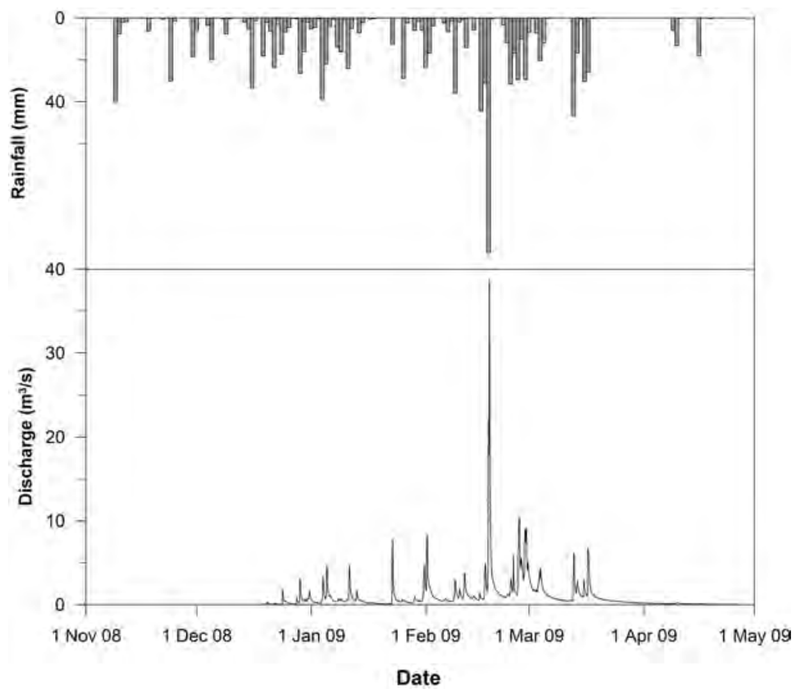


Figure 1 Discharge and rainfall data for GCUS during the 2008–09 wet season

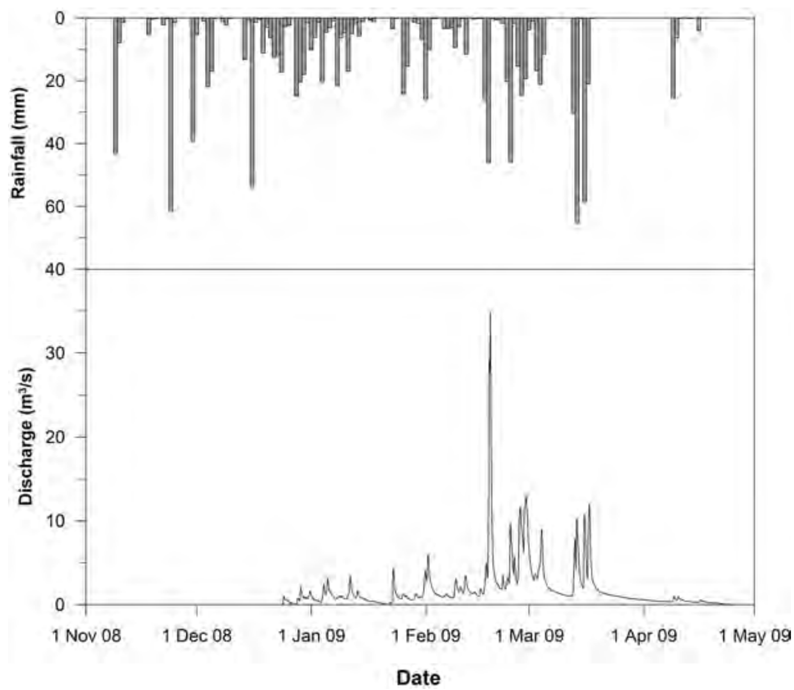


Figure 2 Discharge and rainfall data for GCDS during the 2008–09 wet season

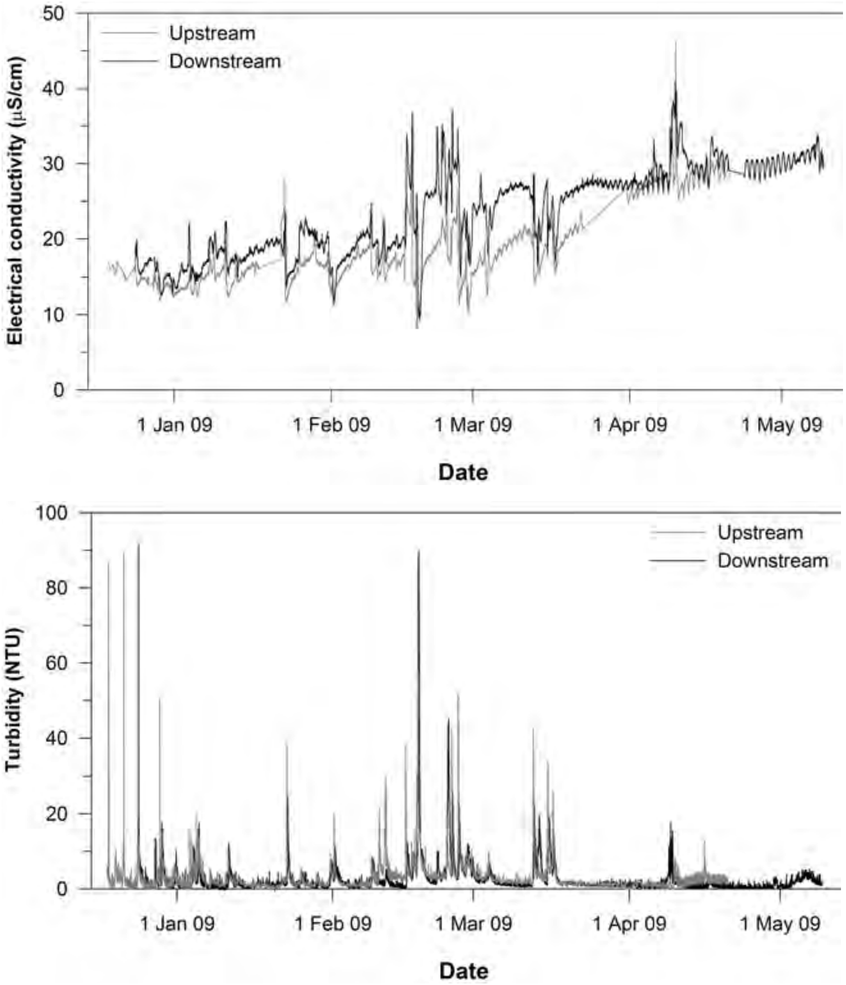


Figure 3 Electrical conductivity and turbidity data for GCUS and GCDS during the 2008-09 wet season

Future work

Rainfall, discharge, turbidity and EC data will be collected for the 2009–10 wet season. cross sections are surveyed annually during the dry season to determine channel changes within the creek. Additionally, continuous EC data will be collected at GS210012 to be able to more clearly identify any inputs contained in runoff from the waste-rock-clad walls of the Ranger tailings dam. Analyses and reporting of these data will progress as new staff are trained in the process.

Reference

Moliere DR, Evans KG & Saynor MJ 2009. Development of catchment geomorphic characteristics of Gulungul Creek – monitoring results. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 166–169.

Moliere DR & Evans KG 2010. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research*, DOI: 10.1111/j.1745-5871.2010.00641.x

Revegetation trial and demonstration landform – erosion and chemistry studies

M Saynor, K Turner, R Houghton & K Evans

Introduction

A trial landform 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. The trial landform will be used to test landform design and revegetation strategies to be used once mining and milling have ceased. The trial landform is an extension of the topography of the TSF, projecting out from the TSF wall in a north-west direction (Figure 1). It is a rectangular shape of approximately 200 m x 400 m (8 ha) in footprint area.

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

The landform was designed to test the performance (in terms of geomorphic stability and suitability for plant growth medium) of 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)

For cover layer 2 from above, two thicknesses (2 m and 5 m) of the mixed laterite and waste rock cover type are being evaluated. It is anticipated that whilst the different thicknesses are not likely to exhibit any material difference in erosion properties, they may differ in their long term ability to sustain mature, deeply rooted vegetation. However, it is the erosion potential and emanation of solutes that are the focus of the work described here.

The landform is divided into six plots (Figure 2), each of which is being used to test different vegetation planting methods and substrate types, as follows:

1 Tube stock planted in waste rock mixed with laterite material to a depth of 2 m;

2 Tube stock planted in waste rock mixed with laterite material to a depth of 5 m;

3 Direct seeding in waste rock mixed with laterite to a depth of 2 m;

4 Direct seeding in waste rock mixed with laterite material to a depth of 5 m;

5 Direct seeding in waste rock material;

6 Tube stock planted in waste rock material.

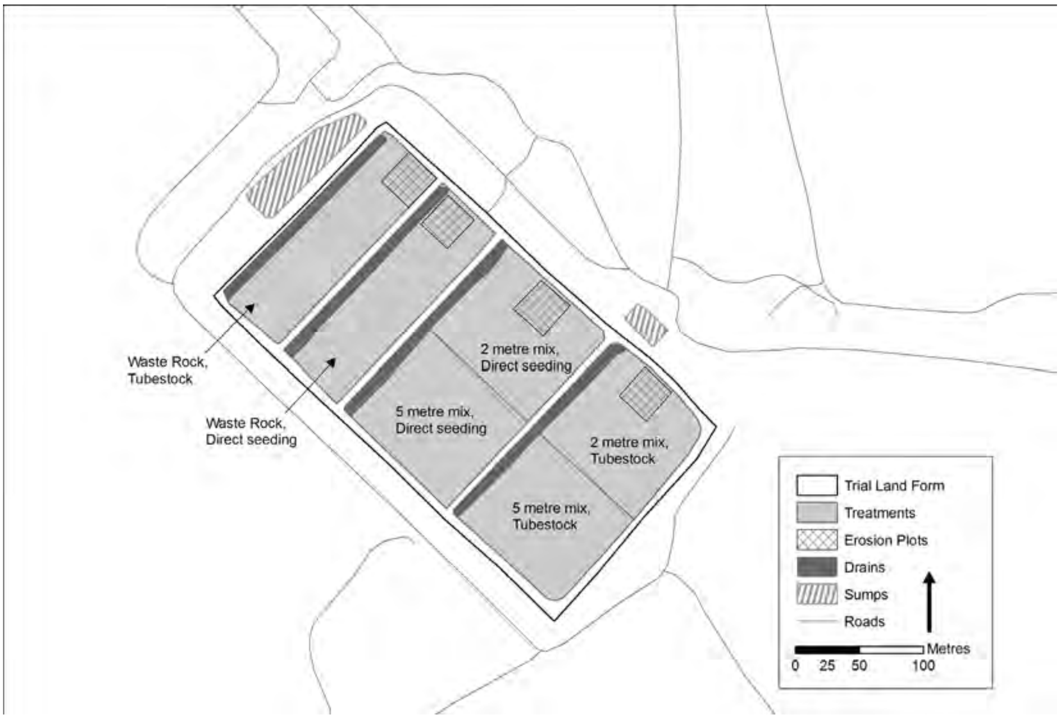


Figure 2 Layout of the plots on the trial landform

Runoff, sediment concentration and water quality (turbidity and electrical conductivity) will be measured in ensuing wet seasons from four 900 m^2 erosion plots that were constructed on the landform between February and July 2009. The plot locations are shown in Figure 2. These locations allow comparison of erosion products and water quality in runoff from a mixed waste rock and laterite substrate vegetated by direct seeding and tube stock and a waste rock substrate vegetated by direct seeding and tube stock.

Plot construction

Each of the 30 m x 30 m erosion plots are physically isolated from runoff from the rest of the trial landform surface area by damp-proof course borders held in place by concrete mortar (Figure 3). Half-section 300 mm diameter U-PVC stormwater pipes (Figure 3) have been placed at the down-slope ends of the plots to capture runoff and channel it through rectangular, broad-crested (RBC) flumes (Figure 4) where rainfall event discharge will be measured. Transported bed sediment will be trapped in a reservoir constructed upstream of the inlet to the flume (Figure 4). Plot construction is described in detail in Saynor et al (2009).

Each flume will be instrumented with a pressure transducer and shaft encoder to measure stage height, a turbidity probe, an automatic water sampler and a data logger. Continuous electrical conductivity data will be measured in the half-section stormwater pipes and in the flumes to provide a measure of the concentrations of dissolved salts in the runoff. A rain gauge will be installed near each flume to record the rainfall and rainfall intensity at each of the plots. All the he data will be downloaded once a day via mobile phone telemetry and then stored in the hydrological database, Hydstra. Decisions on how often the plots will be visited to clear bedload and to collect water samples will be made after the plots are in place and there has been an opportunity to observe erosion rates and discharge relative to size of rainfall event. The construction and instrumentation of the erosion plots was completed early in Q4 2009, in advance of the 2009–10 wet season.

Figure 3 Plastic half pipe trough and boundary

Figure 4 Reservoir and flume at the outlet of the erosion plot

Future work

For the 2009–10 wet season, the resources of the HGP program, with assistance from EWLS, will focus on ensuring the monitoring systems are functioning correctly and on successfully obtaining the first year of data from the plots and reporting these data.

Wet season suspended and bed sediment loads, EC (continuous) and water chemistry data (grab samples) will continue to be collected for several years to track the evolution of the landform surface and any mitigative effect of increasing coverage by vegetation.

Solute generation will be investigated using the continuous electrical conductivity record in conjunction with the major ion composition measured in collected water samples. It is hoped that significant relationships will be found between EC and major-ion solute composition such that continuous EC will be able to be used to infer solute export in the 2009-10 and subsequent wet seasons.

References

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.

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

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

Introduction

The majority of this project, as it relates to Ranger landform evolution simulations, was reported at the 22nd ARRTC meeting in October 2008. It was also reported in Evans et al (2009).

Progress/results

Little further work has been undertaken due to the temporary reassignment of staff to other duties. Collaborative research with Associate Professor Greg Hancock evaluating the Siberia and CAESAR models resulted in a paper presented at the 14th Australasian and Remote Sensing and Photogrammetry Conference in Darwin in October 2008 and now accepted for publication (Hancock et al in press). A paper titled '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.

Steps for completion

The following steps are required to complete this project:

1. Develop model capability to incorporate spatial variability in surface material types and vegetation distribution. Little further simulation work can be done until this step is completed.
2. Systematically test the effects of Digital Elevation Model (DEM) resolution on model outputs.
3. Use the results obtained from the Ranger Trial Landform to calibrate/validate landform evolution model performance.
4. Define the duration and intensity characteristics for extreme rainfall events and apply these to simulations to conduct a risk assessment of landform stability.
5. Compare long-term erosion rates measured on natural undisturbed sites with CAESAR and Siberia simulations for those sites.
6. Assess and evaluate the importance of vegetation on landform stability.

While collaborative research programs have been developed between SSD and Drs Coulthard and Hancock, competing projects (spatial data management project, implementation of photo database Ranger landform trial) took priority in 2008–09. In 2009–10, Dr Coulthard is planning

---

[1] School of Environmental & Life Sciences, The University of Newcastle, Callaghan, NSW 2308, Australia
[2] Department of geography, University of Hull, Hull, HU6 7RX, UK

two visits to ***eriss*** to work on updating the CAESAR model to enable it to simulate variation in surface treatments on a landform ie rock mulch, vegetation and ripping; presently, CAESAR can only simulate erosion of landforms with uniform surface.

Some initial work has commenced on classifying extreme rainfall events using methods taken from the literature. The first method being assessed is that of Casas et al (2004). They used rainfall Intensity-Duration-Frequency curves (IDF curves) to analyse rainfall duration data and to select high intensity/low frequency return interval events based on the IDF curves. Cluster analysis was then used to classify the rainfall events into four groups: 1. very short duration events representing localised rainfall with a clear seasonal influence; 2. mesoscale duration where intense rainfall rate systems may develop; 3. synoptic rainfall events with intensities exceeding 5-year return period level for 12 to 24-hour time intervals; and 4. mid to large-scale meteorological processes showing high rates for large ranges of durations (20 min to 24 hr). Applying the Casas et al (2004) approach to the local (NT) situation, firstly, the Darwin continuous rainfall record (54 years) has been analysed in Hydstra and event duration data extracted, as well as high intensity/low frequency return interval events for each duration based on Darwin IDF curves. The next step will be to classify the events using cluster analysis. The most intense events for each class will be applied to a proposed Ranger rehabilitated landform using CAESAR to simulate erosion rates for each group of rainfall events to assess likely impacts which could occur during 54-year period for which continuous rainfall has been collected in Darwin. Depending on the results arising from use of the Darwin record, the much shorter Jabiru rainfall record may also be assessed.

Other methods of determining the magnitude of extreme events will also be investigated, including slackwater deposit studies (Erskine & Saynor 2000).

References

Casas, MC, Codina, B, Redaño, A & Lorente, J 2004. A methodology to classify extreme rainfall events in the western Mediterranean area. *Theoretical Applied Climatology* 77, 139–150.

Erskine WD & Saynor MJ 2000. *Assessment of the off-site geomorphic impacts of uranium mining on Magela Creek, Northern Territory, Australia*. Supervising Scientist Report 156, Supervising Scientist, Darwin NT.

Evans KG, Hancock GR, Lowry JBC & Coulthard TJ 2009. Assess the impact of extreme rainfall events on Ranger rehabilitated landform geomorphic stability using the CAESAR landscape evolution model. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 100–105.

Hancock GR, Lowry JBC, Coulthard TJ, Evans KG & Moliere DR 2009. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models. *Earth Surface Processes and Landforms* (in press)

Hancock GR, Lowry JBC, Coulthard TJ & Evans KG 2008. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Lowry JBC, Evans KG, Coulthard TJ, Hancock GR & Moliere DR 2009. Assessing the impact of extreme rainfall events on the geomorphic stability of a conceptual rehabilitated landform in the Northern Territory of Australia. In *Proceedings of the Fourth International Conference on Mine Closure*, eds M Fourie & M Tibbett, 9–11 September 2009, Perth, Australian Centre for Geomechanics, 203–212.

Validation of the SIBERIA model, its erosion parameters and erosion rate predictions

G Hancock[1] & K Evans

Introduction

Gully incision and development are important factors in soil erosion, catchment development and subsequent increased sedimentation and water quality problems in many catchments. Understanding the spatial and temporal process of gully initiation and development is of critical importance in understanding the long-term stability of post-mining landforms, which often have steeper slopes than the surrounding undisturbed landscape, are devoid of, or have limited, vegetation cover and may export increased sediment loads to receiving streams through erosion as the landform equilibrates to it surrounds. Therefore, the ability to predict where initiation of gullies may start and how they evolve is very important for environmental management particularly mine sites. The insights gained will allow upfront engineering measures to be employed to minimise gully initiation and development.

This study examines erosion feature characteristics at Tin Camp Creek in the Northern Territory, Australia (Figure 1) and builds on a previous work that has been carried out at this site (Hancock & Evans 2006). The catchment has a geology very similar to the Energy Resources Australia (ERA) Ranger uranium mine and is thought to be an analogue for the long-term rehabilitated post-mining landscape. There appears to be a lack of studies examining gullies in undisturbed environments and long-term studies of the evolution of erosion features are scarce. Little is known about rates of headward movement and changes in width together with scour and fill.

The study catchment has largely uniform geology, soils and vegetation. Moreover, because of its small size, the climate conditions can be assumed to be uniform. The aims of this study are to investigate gullying processes in the catchment by examining trends in gully head drainage area and slope characteristics, as well as gully depth, width and length. The link with hillslope erosion is also examined.

Progress and results to date

In August 2002, gullies in the Tin Camp Creek study catchment were initially mapped after 39 representative erosion features within the entire catchment were selected for long-term monitoring. Thirty-four of these were gully heads and 5 were scour holes located in the channel downstream of a channel head. The selected erosion features covered a range of sizes from rills up to large gullies (Figure 2) (Hancock & Evans 2006). Erosion pins were also installed to measure hillslope rates of erosion and deposition adjacent to the monitored gullies.

The Tin Camp Creek gullies have been measured only over a limited 6 year period, yet of course are the product of many years of erosion. The occurrence of well-defined incisions in the catchment (and surrounding catchments) indicate that gullying is an important process in the area. Further, the fact that gullying is present in the absence of European land use practices or other anthropogenic land disturbance in the area is an important finding.

[1] School of Environmental & Life Sciences, The University of Newcastle, Callaghan, NSW 2308, Australia

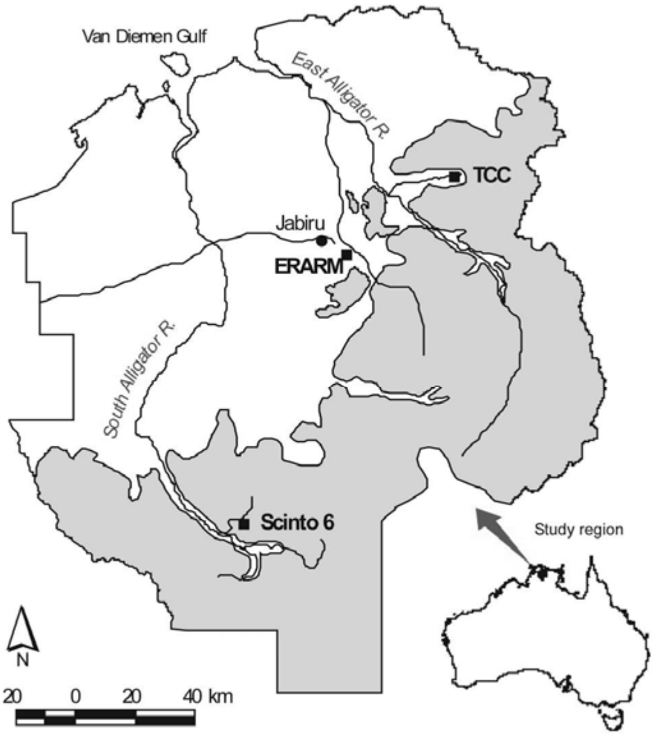


Figure 1 Location of study site (TCC – Tin Camp Creek)

Figure 2 A medium (left) and large (right) size gully within the Tin Camp Creek catchment

No statistical correlation has been found between change in erosion feature width, length or headward movement and hillslope erosion as measured by erosion pins over the study period. Similarly, no relationship was found between upslope area and erosion, nor between slope and erosion as measured by the erosion pins. Nevertheless and as demonstrated in Figure 3, erosion feature depth increases linearly with both hillslope deposition and erosion. This indicates that the deeper the hillslope erosion and depth then the deeper the nearby gully depth. Deeper hillslope erosion and deposition indicate greater overland flow but with different dynamics for each. But for both cases, a greater increase in gully depth indicates greater flow through the gully. This suggests that the gully is an efficient conduit to move water and sediment from the catchment, regardless of which processes are occurring on the nearby hillslope.

Examination of the relationship between hillslope and channel erosion and deposition on an annual basis revealed no strong trends, suggesting that the system is not sensitive to annual variability and that change occurs over longer time periods. While no major change has occurred over the 6 year period, the presence of vertical headcuts, steep side walls and large pot holes, together with the finding that these features have largely been maintained, suggests that the system had reached a quasi-steady state before the start of the study period.

Long-term studies examining erosion features are scarce (Harvey 2001). The finding that the erosion features have changed little over the 6 year monitoring period is a significant result as during this time the catchment has been subject to fire every second year (Hancock, personal observation) with all surface grass cover removed, as well as both average and in 2007, well above average rainfall. In 2006 the majority of trees were pushed over by winds from a Category 4 cyclone.

The findings reported here are similar to those of Harvey (2001). Though the climate for his study (in northwest England) is very different from Arnhem Land, over a 30-year monitoring period the gullies began a progressive trend towards stabilisation. He suggested that destabilisation may occur in response to rare extreme rainfall, with return periods greater than that on record. A change in landscape management or use may also destabilise the system.

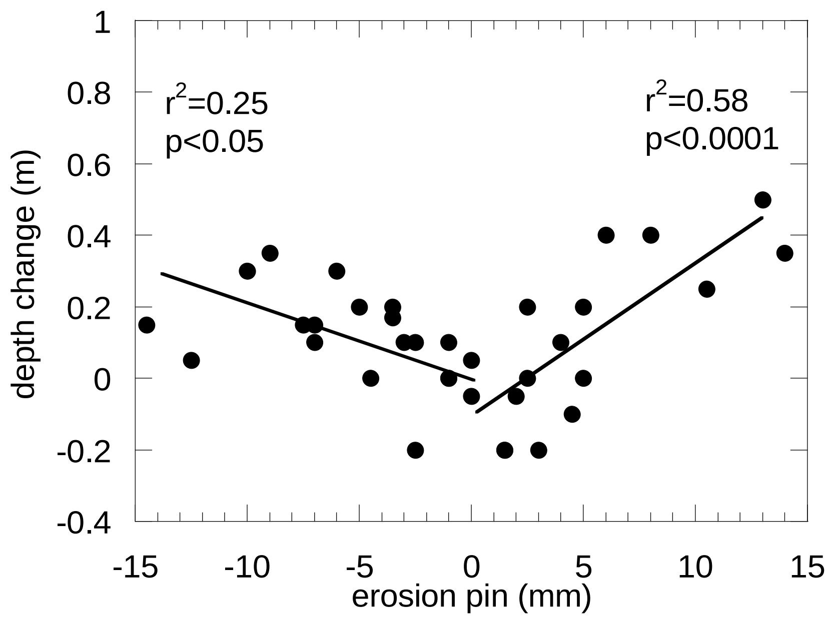


Figure 3 Erosion as measured by erosion pin and the change in erosion feature depth at Tin Camp Creek. Negative values represent deposition while positive values represent erosion.

While some of the erosion features at Tin Camp Creek became shallower (ie filled in) and reduced their width due to deposition, they were maintained and appear to be permanent features of the landscape. From a post-mining landscape perspective, this study shows what erosion features, such as gullies, may evolve to look and behave like in the very long term.

Steps for completion

This is a joint study between Associate Professor Greg Hancock, University of Newcastle, and ***eriss***. Geomorphic studies require lengthy periods of data collection and the study of gullies at Tin Camp Creek will continue to contribute to a database of natural erosion rates for the region. These data will be used for calibration and evaluation of the Siberia and CAESAR erosion and landscape evolution models (Hancock et al 2009). From these data, input parameter values can be derived for Siberia and CAESAR and applied to simulation of erosion of post-mining landforms since parts of the digital elevation model used to represent the mining landform will have areas of undisturbed land.

References

Hancock GR, Evans KG. 2006. Gully position, characteristics and geomorphic thresholds in an undisturbed catchment in Northern Australia, *Hydrological Processes* 20, 2935–2951.

Hancock GR, Lowry JBC, Coulthard TJ, Evans KG & Moliere DR 2009. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models, *Earth Surface Processes and Landforms*, in press.

Harvey AM 2001. Coupling between hillslopes and channels in upland fluvial systems: implications for landscape sensitivity, illustrated from Howgill Fells, northwest England. *Catena* 42, 225–250.

Definition of sediment sources and their effect on contemporary catchment erosion rates in the ARR: landslips

GW Staben, MJ Saynor & JBC Lowry

Background

As a result of record rainfall and floods in the 2006/07 wet season (Moliere et al 2007), a number of landslides occurred in the upper reaches of the Magela Creek and East Alligator River. These landslides occurred on well vegetated, exhumed Oenpelli Dolerite surfaces surrounded by Mamadwerre Sandstone, and had the potential to supply sediment to both Magela Creek and the East Alligator River (Figure 1). Currently the Supervising Scientist Division monitors water quality in Magela Creek to assess the impact on stream sediment loads by mine-related activities at the Ranger mine. These landslides have the potential to increase the baseline load of fine suspended sediment to Magela Creek during future wet seasons and are not mine related. It is therefore important to quantify the impact and extent of natural events affecting stream sediment loads, to enable them to be differentiated from mine impacts.

These landslides occurred in areas only accessible by helicopter, thus remotely-sensed data such as ALOS AVNIR-2 have the potential to offer an affordable means to investigate the impact of these landslides across the landscape.

This study's objectives were:

1 To evaluate the use of remotely sensed data in detecting landslides

2 To determine whether the available data can be used to accurately quantify the area of each landslide

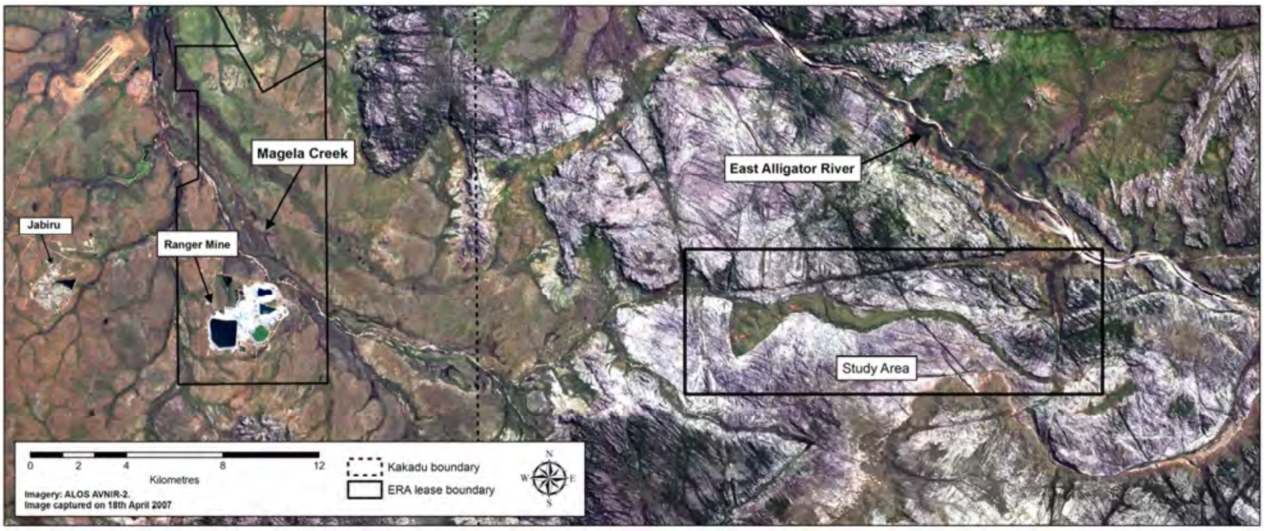


Figure 1 General map showing the location of the landslides study area

Methods

Materials

Two ALOS AVNIR-2 (Advanced Visible and Near Infrared Radiometer type 2) multispectral satellite images obtained from Geoscience Australia were used for this study. The ALOS AVNIR-2 is a push broom sensor with four spectral bands capturing data in three visible bands (blue, green and red) and one near infrared band. It has a swath width of 70 km and a spatial resolution of 10 m at nadir. The spatial resolution of an ALOS AVNIR-2 image is determined by the point angle of the sensor at time of image capture (ALOS User Handbook 2007). The only suitable image available to assess the area prior to the landslides was captured in June 2006 at a spatial resolution of 20 m. The second (post landslide) image was acquired in April 2007 at a spatial resolution of 10 m. Both images consisted of four spectral bands. A one second DEM (digital elevation model) was also used in this project. It was produced by the Defence Imagery and Geospatial Organisation and has a spatial resolution of ~30 m and a vertical accuracy estimated to be ± 17 m.

Image analysis

Image pre-processing

A number of image pre-processing steps were performed including re-registration of the 2006 image to the 2007 image and extraction of slope values (in degrees) from the DEM using the topographic modelling tool in the remote sensing software ENVI. The normalised difference vegetation index (NDVI) was then derived from the two ALOS AVNIR-2 image dates. Vegetation indices were used, as the contrast between the boundaries of the landslides was high. The four spectral bands and vegetation index for each image date and the slope data (derived from the DEM) were then combined in an 11 band layer stack at a spatial resolution of 10 m for further analysis in eCognition.

Classification

A multi-resolution image analysis was undertaken using the geographic object based image analysis (GEOBIA) software, eCognition. GEOBIA using eCognition involves two main steps, firstly, segmentation of the image into objects and then classification based on the statistics derived from these objects (Al-Kudhairy et al 2005). The multi-resolution segmentation of an image enables the user to investigate objects at a variety of spatial scales (Benz et al 2004) allowing targets to be identified and classified at a range of different spectral and spatial scales. Information from the different spatial scales can then be combined into a hierarchical network where each object is related to its context, neighbourhood, and sub- or super-objects (Bock et al 2005).

To exploit the different information available at different spatial scales, a number of segmentations were performed on the data (Figure 2). The initial segmentation (Level 1) was set at a scale to define individual landslides whilst the second (Level 2) was undertaken to enable discrimination between the sparsely vegetated sandstone plateau/escarpment regions of the image and the more fertile soil substrates which support vegetation growth. The level 2 segmentation was important as it enabled areas such as the sandstone regions in the image, which have similar spectral properties as landslides to be identified and correctly classified.

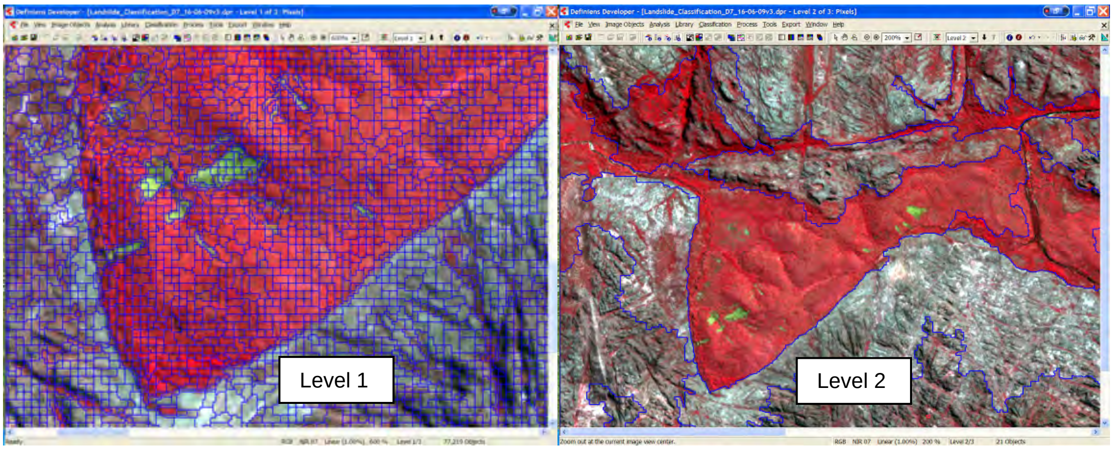


Figure 2 Examples of the multi-resolution segmentation performed on the data

A number of steps were involved in the classification process. The first was to classify the image into two broad categories (sandstone plateau/escarpment and vegetated areas) using the level 2 segmentation. This information was then used in the subsequent classification of the finer scale segmentation (level 1). A classification using membership functions was then developed for the finer scale segmentation with classes identifying areas of shadow/water, vegetation, bare ground, landslides and their associated debris flow. Investigation of the results of the level 1 classification indicated that some areas around the landslides were being incorrectly classified as debris flow. In order to enhance the classification, a chessboard segmentation (Level 0) was undertaken on the areas classified as debris flow in level 1 with further refinement of the classification then undertaken.

Accuracy assessment

Accuracy assessment was undertaken to assess both the thematic and geometric accuracy of the automated classification. Two methods were used, first a manual classification of landslides was undertaken. This included identifying and estimating the area of each landslide derived from visual interpretation of the multispectral ALOS AVNIR-2 satellite data. Vector polygons were produced using heads-up digitising methods for each landslide in a GIS (Geographical Information System) software, ArcMap. This process was undertaken with the image set at a consistent scale (1: 5000) and with the aid of oblique digital photographs taken during aerial surveys. Estimates derived from the satellite data were undertaken to quantify the number of landslides and to obtain areal estimates for all landslides in the study area, with the presence of landslides validated by aerial surveys conducted by helicopter. The second method assessed the geometric accuracy of the classification. Ground truth data were collected for 11 of the landslides located within the Magela catchment. This was done by capturing points around the perimeter of each of the landslides at approximately 3 to 10 m spacing's using a GPS (± 5m horizontal spatial accuracy). These data were then input into a GIS and converted to vector polygons for further analysis.

Progress/results

Visual assessment of the ALOS AVNIR-2 data and aerial surveys (by helicopter) identified a total of 56 landslides across the 88 km^2 study area. Comparison of these data with the results of the classification show that we were able to successfully detect 60.7% (n = 34) of the 56 landslides. Twenty-two landslides were not detected by the classification algorithm while 8 objects were incorrectly classified (commission error) as landslides. The results of the classification estimated the total area of landslides to be 11.3 ha, representing a significantly higher value than the manual classification estimate of 6.4 ha (Table 1).

Of the 6.4 ha in area estimated to be impacted by landslides the classification was able to detect 5.96 ha, while the 22 undetected landslides covered an area of 0.39 ha. This suggests that while the detection rate for individual landslides was only 60.7%, the classification was still able to detect the majority (94%) of the area impacted by landslides. Assessment of the mean size difference (based on the manual classification area calculations) between the 34 detected (0.18 ha) and 22 undetected (0.02 ha) landslides, would suggest that the ability to detect and correctly classify landslides using ALOS AVNIR-2 imagery and the GEOBIA approach is related to the size of the landslide.

The geometric accuracy of the mapped landslides was assessed by directly comparing 11 of the landslides measured by imagery, with ground truthed data for the same landslides. All 11 landslides were detected by the classification algorithm, though again, there was a significant over-estimation of the area classified as landslides using image data (Table 2).

Table 1 Summary statistics for the automated and manually-classified landslides

	No. landslides identified.		**Total area (ha)**		**Mean area of individual landslides (ha)**	
	Automated classification	Manual classification	Automated classification	Manual classification	Automated classification	Manual classification
Detected	34	56	11.27	5.96	0.33	0.18
Omission error	(22)			0.39		0.02
Commission error	8		1.50		0.19	
Total	42	56	12.77	6.36	0.33	0.11

Table 2 Direct comparison between the areas of the 11 landslides measured during the collection of ground truth data and the classified landslides

Landslide	**Ground truth area (m²)**	**Classification (m²)**	**Difference in area (m²)**
1	118	1000	882 (88%)
2	1241	2600	1359 (52%)
3	233	1600	1367 (85%)
4	262	700	438 (63%)
5	301	600	299 (50%)
6	1547	3500	1953 (56%)
7	1293	4800	3507 (73%)
8	5296	7900	2604 (33%)
9	4321	8700	4379 (50%)
10	4641	12900	8259 (64%)
11	174	1000	826 (83%)
Total	19427	45300	25873 (57%)

Comparison of the actual (ground-truthed) area of the 11 landslides to the classified areal data showed that 94% of the area covered by each of the 11 landslides on the ground was captured in the classification (Table 3). However, the commission error was very high with an average of 60% for the 11 landslides. This over-estimation of landslide area can be attributed to the miss-classification of debris flow as landslide. The area of each landslide not detected (omission error) was low with exception of landslide five where 66% went undetected. This was due to the geometric shape of the landslide which was predominantly long and thin (less than a pixel in width) resulting in the mixed pixel effect, which is due to the spectral response of the tree canopy and landslide combined. However, the scour depth of this landslide resulted in significant debris flow down-slope which aided in its detection .

Conclusions

The results of this study show that GEOBIA classification methods detected 60.7% of the landslides across the study area. It was found that even though the detection rate was modest, those landslides that were detected were estimated to represent 94% of the total area impacted by landslides in the study area. The spatial and spectral resolution of ALOS AVNIR- 2 data alone were not sufficient to discriminate between landslides and their associated debris flows. The spectral response of bare soil areas, landslides and their associated debris flow was very similar. With the addition of slope values derived from the DEM data, the discrimination between landslides and other bare soil areas in the image was possible, as landslides did not occur on flat areas. Analysis of the ground-truthed data shows that while 94% of the landslide area was captured in the classification, significant areas of debris flow were also miss-classified as landslide area. This miss-classification of debris flow as landslides was found to be due to the existence of debris flow on areas with high slope values. The results of this project show that object-based analysis of ALOS AVNIR-2 remotely sensed data can be used to detect the significant landslides in the study area. However, the use of these results to measure the landslide area is not feasible as discrimination between a landslide and its associated debris flow is not possible.

Table 3 Results of the analysis between the automated classification and the 11 landslides measured during the collection of ground truthing (GT) data

Landslide	Ground truth area (m^2)	Classification (m^2)	Area classified within the extent of the GT data	Commission error (m^2)	Omission error (m^2)
1	118	1000	93	907	25
2	1241	2600	1170	1430	71
3	233	1600	232	1368	1
4	262	700	232	468	30
5	301	600	102	498	199
6	1547	3500	1547	1953	0
7	1293	4800	1230	3570	63
8	5296	7900	4583	3317	713
9	4321	8700	4295	4405	26
10	4641	12900	4599	8301	42
11	174	1000	123	877	51
Total	19427	45300	18205 (94%)	27095 (60%)	1222 (6.29%)

References

ALOS User Handbook 2007. Earth observation Research Centre, Japan Aerospace Exploration Agency, http://www.eorc.jaxa.jp/ALOS/doc/alos_userhb_en.pdf

Al-Kudhairy DHA, Caravaggi I & Giada S 2005. Structural damage assessments from IKONOS data using change detection, object-oriented segmentation, and classification techniques. *Photogrammetric Engineering & Remote Sensing* 71, 825–837.

Benz UC, Hofmann P, Willhauck G, Lingenfelder I & Heynen M 2004. Multiresolution, object-oriented fuzzy analysis of remote sensing data for GIS-ready information. *ISPRS Journal of Photogrammetry & Remote Sensing* 58, 239–258.

Bock M, Xofis P, Mitchley J, Rossner G & Wissen M 2005. Object-oriented methods for habitat mapping at multiple scales – Case studies from Northern Germany and WyeDowns, UK. *Journal for Nature Conservation* 13, 75–89.

Moliere DR, Evans KG & Saynor MJ 2007. Hydrology and suspended sediment transport in the Gulungul Creek catchment, Northern Territory: 2006–2007 wet season monitoring. Internal Report 531, June, Supervising Scientist, Darwin. Unpublished paper.

Assess the impact of tailing subsidence on rehabilitated landform erosional stability

R Houghton

Introduction

Where the design of a rehabilitated mine landform includes disposal of unconsolidated tailings in worked-out open pit(s) and the capping of the tailings with benign material, it is critical that the backfill and capping design accommodates tailings settlement and that surface drainage quickly removes water to reduce infiltration, surface erosion and possible exposure through gully incision. If tailings consolidate at a rate and to an amount that causes the cap to subside and fracture, drainage line direction can change, causing surface flow to accumulate and discharge through a single point, causing severe incision. Fractures may also expose tailings and allow both direct inflow and percolation of water through the containment area and transport of contaminants into the groundwater system.

It is important that the impact of tailings subsidence is understood so that the effect can be allowed for in the final design of the capping structure for Pits 1 and 3 at the Ranger mine site. Investigation of uranium mine pits in the region (Nabarlek and Rum Jungle) in which tailings have been placed and subsequently backfilled and capped provides the opportunity to better understand the longer term effects of tailings subsidence in capped pits. For this research Dyson's Pit at Rum Jungle was selected as the study site to provide information that could be used to assist the design process for capping Pits 1 and 3 at the Ranger mine site.

The Rum Jungle site, comprising Whites Pit, Intermediate Pit and Dyson's Open Cut and Whites underground workings, produced uranium, copper, lead and zinc and was operational between 1952 and 1971. Rehabilitation of Dyson's Open Cut (the primary uranium pit) was completed in late 1984. This involved backfilling the pit with the processed tailings, contaminated low grade copper ore and contaminated subsurface soils from locations around the mine site. This material was capped with a constructed interblanket drainage layer shaped with a continuous low angle grade longitudinally and a transverse concave grade which converges to a central rock lined diversion channel with a final layer of plant growth medium (Pidsley 2002) (Figure 1). Information on how the rehabilitated Dyson's Pit has performed to its design characteristics provides important information that could be used in the proposed strategies for the backfill capping of tailings at Ranger mine site.

Aims

The project aim was to measure and model settlement of the capped Dyson's pit to identify specific issues that may need to be considered and addressed for capping of the in-pit tailings at the Ranger Mine. As the landform matures, slope angles and elevations may change as a result of consolidation. Thus it is important to understand how consolidation may impact on settlement of the backfilled pit landforms and, in particular, how this differential settlement may effect erosion of material from the capped pits and the behaviour of the capping itself in the context of being a sustainable water shedding construct. Specifically, it needs to be determined how the effects of differential consolidation can be incorporated in landform evolution modelling and therefore improve (in the reverse engineering sense) the final design of the rehabilitated landform.

Figure 1 Longitudinal drain in the cap of Dyson's Open Cut. Significant signs of subsidence can be seen in the vicinity of the single tree located towards the mid-background in the top right of the photograph. The drain has subsided here and acts as a basin to accumulate runoff. Signs of acid rock drainage are visible in the rock mulch in the bottom left of the photograph and in the drain toward top centre where rock mulch colour changes from grey to rust/brown.

Figure 2 Accumulation of acidic and metalliferous runoff in subsided section of surface drain. Riprap seen in foreground of Fig 1 is upslope in this photo.

Methods

The present day surface of the Dyson's landform was surveyed in 2008 to obtain a digital elevation model (DEM). The 1986 'as-constructed' surface survey was only available as a hard-copy contour plan. This plan was manually digitised and ArcGIS software used to create a DEM. A comparison of the 2008 DEM to the 1986 DEM enabled the change in surface level between the DEMs to be quantified. It was assumed that the measured differences were the result of consolidation/settlement of the buried tailings between 1986 and 2008.

The Tezaghi (1945) model (see Smith 1990, 361) using simplified linear and elastic consolidation theory for fine grained soils was used to mathematically quantify one-dimensional settlement of the cap. The main assumptions in the theory are:

1. The soil is essentially homogenous and saturated,
2. Both the soil pore water and soil particles are incompressible,
3. The coefficient of consolidation (C_v) is constant through time,
4. Darcy's law of saturated flow applies,
5. The compression-induced consolidation is in one dimension (vertically down) only,
6. Expulsion of water from the soil voids is in one direction only (vertical),
7. The change in the bulk volume of buried material is only due to a change in the void ratio, which in turn is due to a corresponding change in the effective stress. The relationship between void ratio (e) and effective stress (σ') is linear.
8. There is no instantaneous volume change on application of the overburden pressure increase (total stress increase).

Results

The topographic survey results comparing 1986 with 2008 are shown below in Figures 3, 4 and 5. Settlement due to one dimensional consolidation was estimated using the numerical method of finite difference. The model parameter values were iteratively fitted by varying the coefficient of volume decrease (m_v) until the current, observed settlement value of 2.9 m was returned for the period of 22 years. The simulated variation of settlement with time is shown in Figure 6.

The observed maximum settlement of the landform surface approximately 1.5 y after construction was 900 mm (Allen & Verhoeven 1986). This value compared well with the modelled settlement of ≈800 mm after 1.5 y. The 100 mm settlement difference suggests a reduced pore water dissipation rate in the model and is an acceptable discrepancy due to the simplicity of the 1-dimensional model. The model indicates that the amount of settlement decreases with time (Figure 6) and that equilibrium will slowly be achieved as excess pore waters are dissipated.

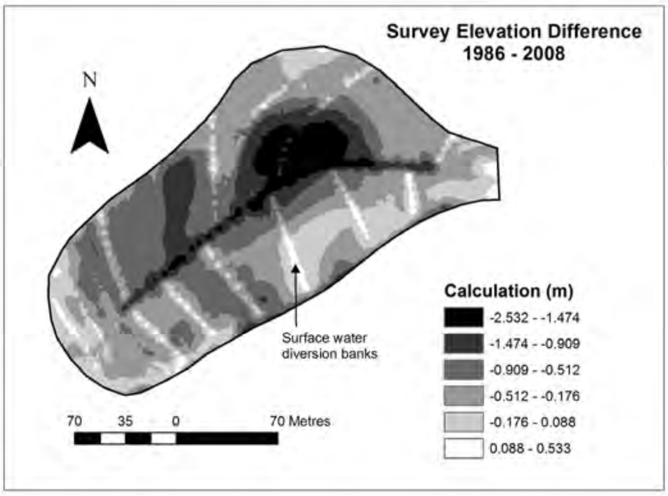


Figure 3 Dyson's surface overlay of 1986 and 2008 data showing difference in height

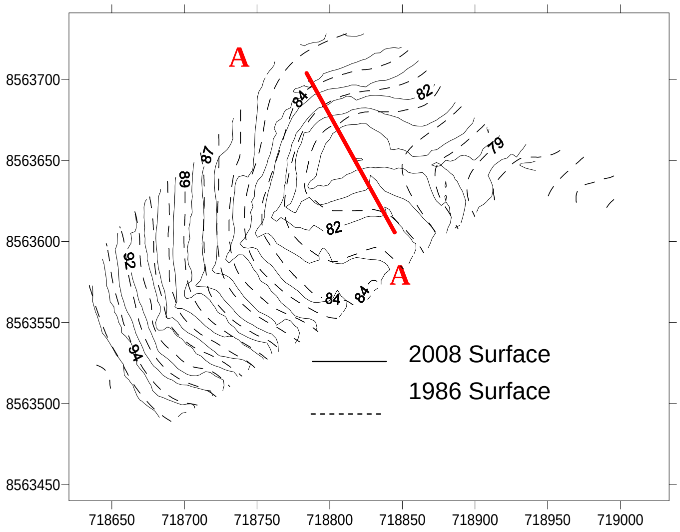


Figure 4 1986 & 2008 contour map overlay showing Section A-A. Dimensions are in metres.

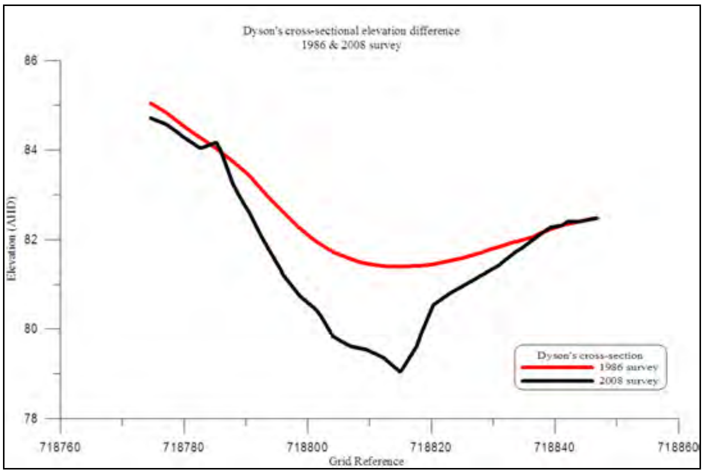


Figure 5 Section A-A showing the subsidence of the cap that has occurred between 1986 (top line) and 2008 (bottom line)

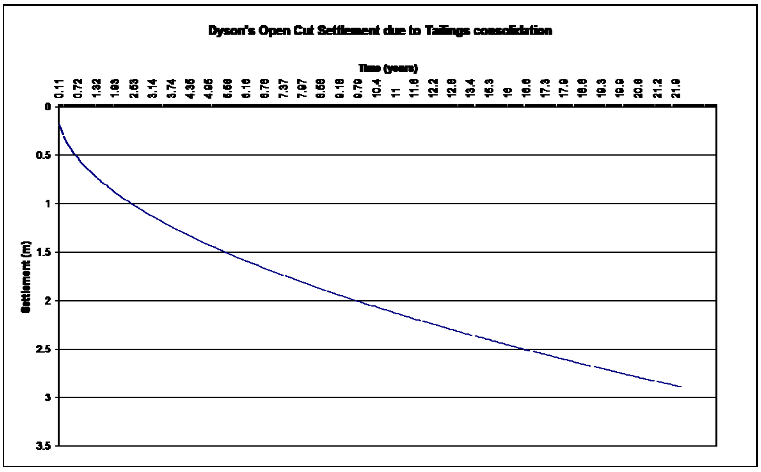


Figure 6 Plot of simulated one-dimensional consolidation/settlement against time

Conclusions

The capped pit area of Dyson's Pit was observed to have settled in the years following its construction, indicating that the original design did not adequately account for consolidation. Evidence of this consolidation has been disruption of the surface line drainage causing ponding of acidic water that promotes infiltration rather than runoff. Steeper slopes within the structure flowing into the depression could also increase erosion. These consequences of not adequately accounting for consolidation at Dyson's Pit emphasise the importance of appropriately accounting for this process in the design of the capping and backfill for the Ranger pits.

The lack of specific site geotechnical data meant that only Terzaghi's (1945) one dimensional consolidation model could be used for this work (Smith 1990, 361). The major limiting factors influencing a realistic comparison of model-predicted settlement to the observed settlement at Dyson's is the limited knowledge of the tailings geotechnical characteristics. The geotechnical characteristics could be determined by a drilling program that recovers relatively undisturbed tailing samples for laboratory testing. A drilling program has the potential to yield substantial critical information such as:

1. The vertical structure of the backfilled pit could be mapped, quantifying layer depths both in the capping layer and the underlying tailings.
2. The volume change behaviour of the tailings would be better understood through laboratory testing of the samples recovered.
3. The process by which further settlement occurs can be determined by installing piezometers in the bore holes from which the samples for geotechnical testing have been extracted. The pore water pressures may then may be monitored through time. If the recorded pressures remain relatively constant, further settlement may be attributed to secondary compression. The pore pressure information would provide the data needed to develop a more realistic settlement model.
4. Water pressure data in the pit bore could be compared to data from groundwater bores in the natural terrain downgradient from the pit to determine the flow path boundary conditions for water in the pit.

In the event that these additional data are obtained then much more comprehensive two dimensional and three dimensional models be applied.

Several successful outcomes of the study were achieved, including:

1 The settlement of the Dyson's rehabilitated landform between the years 1986 and 2008 has been quantified with the use of precision survey instruments and current gridding and mapping software to the best possible accuracy given the historical data available.

2 The major characteristics that will influence the potential instability of a landform constructed above a layer of impounded tailings are now better understood.

Whilst the modelling outputs from this study are not specifically applicable to Pits 1 and 3 at the Ranger mine, the findings clearly emphasise the need to account for the effects of consolidation during the design phase.

References

Allen CG & Verhoeven TJ (eds) 1986. The Rum Jungle rehabilitation project final project report, Department of Mines and Energy, Northern Territory.

Pidsley SM (ed) 2002. Rum Jungle Rehabilitation Project Monitoring, Report 1993–1998. Northern Territory Government Department of Infrastructure, Planning and Environment.

Smith GN 1990. *Elements of soil mechanics.* 6th ed, BSP Professional Books, London.

Pre-mining radiological conditions at Ranger mine

A Bollhöfer & A Esparon

Introduction

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) as recommended by the International Commission on Radiation Protection (ICRP 2007). This dose is on top of the natural pre-mining background dose. 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. Hence, pre-mining conditions need to be assessed accurately so that post-mining changes can be quantified in the context of the success of rehabilitation in complying with radiological exposure standards.

Historical airborne gamma surveys (AGS) coupled with ground truthing surveys, have the potential to provide a powerful tool for an area wide assessment of the pre-mining terrestrial gamma dose rates. Recent AGS coupled with ground truthing surveys have been used for regional assessments of radiological conditions at rehabilitated and historic mine sites elsewhere in the Alligator Rivers Region (Martin et al 2006, Bollhöfer et al 2008). The aim of this project is to ground-truth historical AGS data at an appropriate undisturbed radiologically anomalous site in order to extrapolate to pre-mining radiological conditions at Ranger.

An AGS of the Alligator Rivers Region was flown in 1976 and has been used to identify undeveloped radiologically anomalous areas in the vicinity of the Ranger lease as potential candidates for groundtruthing. A comparison of signal intensity with known uranium occurrences in the MODAT database suggested that Anomaly 2 to the south of the Ranger lease may be a suitable analogue site for Ranger pre-mining radiological conditions, as it exhibited a strong airborne gamma signal in the 1976 data (Esparon et al 2009). Based on the assessment of the historical AGS data it was decided to obtain groundtruthed data in the greater region of Anomaly 2. An extensive fieldwork program to the south of the Ranger lease was started in 2007, and this has continued through subsequent dry seasons.

Methods

The 1976 AGS data were acquired from Rio Tinto by the NT Government, and are available on the public domain (the *Alligator River Geophysical Survey*). 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. The line spacing of this survey was 300 m. However, the flying height is unknown.

Extensive ground gamma radiation surveys of Anomalies 2A and 2B were conducted in 2007. In addition, 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 for refining further extensive groundtruthing fieldwork conducted in July – October 2008 to precisely establish the location of Anomalies 2A and 2B.

As of August 2009 more than 1800 individual gamma dose rate readings (and some readings with an in-situ NaI detector) have been obtained in the area.

Radon exhalation was measured in July 2009 at 25 sites using conventional charcoal canisters, with 3 charcoal canisters deployed at each site for a period of three days. In addition, external gamma dose rates were measured and soil scrape samples were taken at these radon sites. Track etch detectors were also deployed for three months to measure dry season airborne radon concentration. Analysis of airborne radon concentration data and soil activity concentrations are underway.

Results

Gamma radiation

During the 2008 ground gamma surveys a spatial shift was confirmed of approximately 200 metres between the groundtruthed and the 1997 AGS data. This spatial shift needs to be accounted for before the two datasets can be quantitatively compared. Figure 1 shows the shifted and interpolated 1997 AGS data (total counts) and a comparison with the raw gamma dose rate results (µGy·hr^{-1}) from SSD's groundtruthing in 2007–2009, respectively.

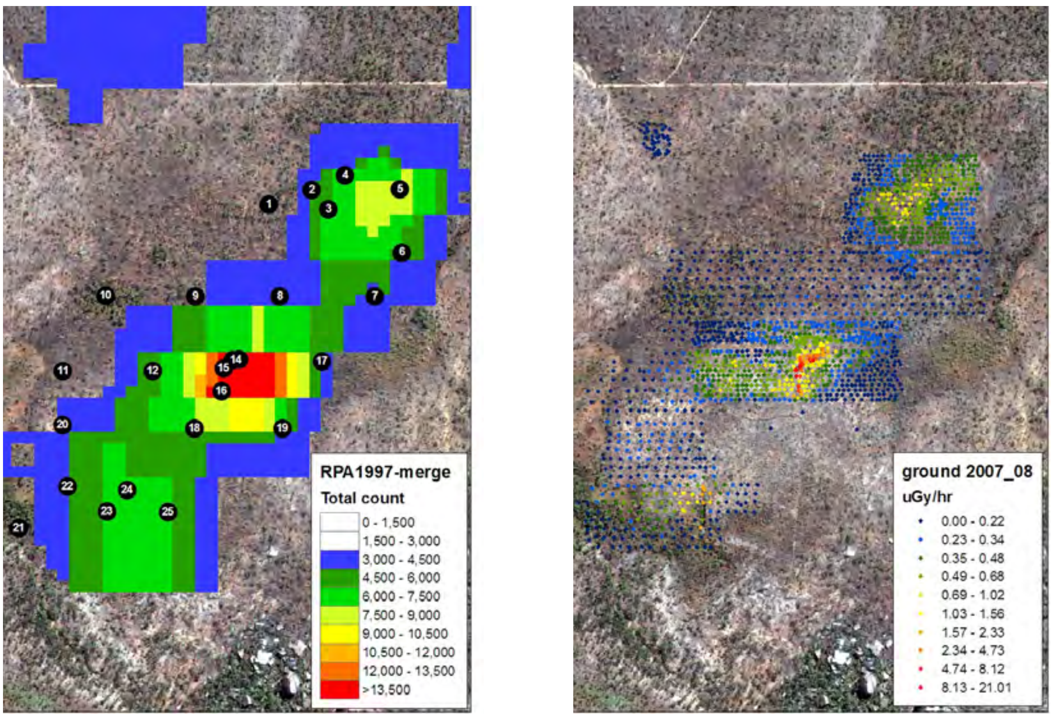


Figure 1 Interpolated1997 AGS data and locations of the radon exhalation survey points (left) and the results of the on ground gamma dose rate measurements (right) performed from 2007 to 2009

To groundtruth an AGS, the data acquired in the field (uranium, thorium, potassium concentrations, and/or gamma dose rates) are plotted against the count rates in the respective channels from the AGS. Ground-based data acquired along transects, for example, are then typically smoothed using an *n*-point average resulting in a resolution similar to the resolution of the AGS (see for example Martin et al 2006, Bollhöfer et al 2008). 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 to allow a comparison to be made between the groundtruthed and the AGS data. For example, choosing too small a pixel size to smooth the groundtruthed data will lead to a comparatively steeper slope when plotting groundtruthed versus AGS data, as localised areas on the ground that exhibit high gamma dose rates will have more influence on the regression. In contrast, a pixel size that is too large will lead to a gentler slope, approaching zero as pixel size is increased. Algorithms are being developed at present using *MatLab*, to smooth the 2-dimensional data such that the correlation of the two data sets shows the best regression coefficient (R^2). Once a relationship between the 1997 AGS and the 2007–09 ground truthed data has been established a correlation will be determined between the 1976 and 1997 AGS data sets. Initial screening of the two (1997 and 1976) AGS data sets (see Figure 2) shows a good correlation ($R^2 = 0.7$) between the two. Note that both datasets needed to be spatially corrected before a comparison was made.

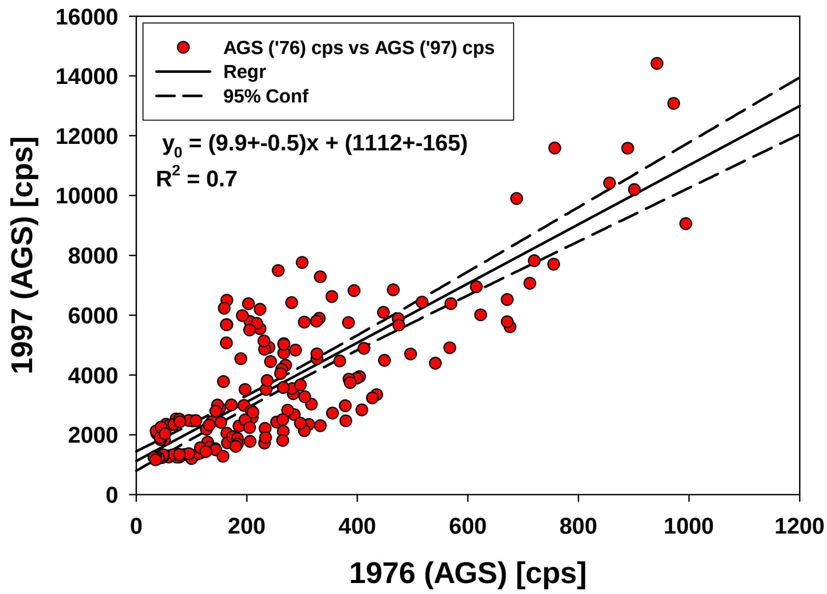


Figure 2 Total counts from the 1997 AGS plotted against total counts from the 1976 AGS

Radon flux densities

Radon flux densities have been measured across the Anomaly 2 area (black dots in Figure 1) and show large variations due to the large variations in radium activity concentrations in the soils. Figure 3 shows the geometric means of the radon flux densities plotted versus the terrestrial gamma dose rates measured at the sampling sites on a log-log scale (the inset shows the data from 0–2.5 $\mu Gy \cdot hr^{-1}$ on a linear scale).

The sampling sites have been divided according to soil type, identified by visual inspection in the field, and the results are plotted for fine gravel, loamy sand and coarse gravel/rocks, respectively. There is no correlation between radon flux densities and gamma dose rates measured for coarse material and rocks. This effect has been observed previously and is due to the variable pore space and thus highly variable radon exhalation originating from coarser material (Lawrence et al 2009). However, there is a strong positive correlation between radon flux densities and gamma dose rates for both, fine gravel and loamy sand soil types.

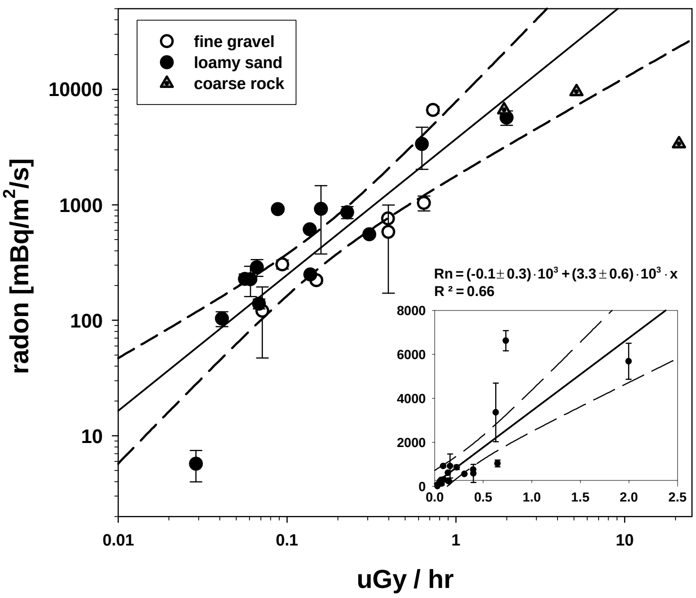


Figure 3 Radon flux densities plotted versus the terrestrial gamma dose rates measured at Anomaly 2

Statistical analysis of the variance of the data using a general linear model shows that there is no difference between the radon flux densities determined for fine gravel and loamy sands. However, there is a small but statistically insignificant difference (p= 0.06) in the slopes. Consequently, the regression using the combined (gravel and loamy sand) dataset is shown in Figure 3, which allows calculation of radon exhalation from known terrestrial gamma radiation data at Anomaly 2. A similar regression (radon flux density versus ^{226}Ra soil concentration) will be plotted once gamma spectrometry results for the soils collected are available. This regression will then be compared with results obtained previously in the region (Lawrence et al 2009) and may be used to determine the radon flux from Anomaly 2 to extrapolate to orebodies 1 and 3.

Steps for completion

Analyses of soil radionuclide activity concentrations need to be finalised. This will allow a correlation to be established between ^{226}Ra soil activity concentrations and radon flux densities at Anomaly 2, similar to the $R_{E\text{-}R}$ determined by Lawrence et al (2009). This correlation can then be used to determine radon fluxes from Anomaly 2. Radon fluxes will also be related to radon activity concentrations measured in the air via track etch detectors deployed across the Anomaly. Track etch detectors were placed in the field in July 2009 and collected in October 2009. Results are pending.

Algorithms need to be developed to upscale results from the groundtruthing data so that comparison can be made with both the 1997 and 1976 AGSs. Once data analysis is complete, the radiological conditions on ground around Anomalies 2A and 2B will be correlated to the pre-mining airborne signal in an attempt to extrapolate to the area wide radiological conditions at Ranger before mining commenced.

Acknowledgments

Jared Selwood, Bruce Ryan, Alan Hughes, Rocky Cahill and Gary Fox are thanked for assistance in the field. The Northern Territory Geological Survey and ERA are thanked for the provision of the 1976 and 1997 AGS data, respectively. Thanks to the Mirarr people for allowing access to the sites.

References

Bollhöfer A, Pfitzner K, Ryan B, Martin P, Fawcett M & Jones DR 2008. Airborne gamma survey of the historic Sleisbeck mine area in the Northern Territory, Australia, and its use for site rehabilitation planning. *Journal of Environmental Radioactivity* 99, 1770–1774.

Esparon A, Pfitzner K, Bollhöfer A & Ryan B 2009. Pre-mining radiological conditions at Ranger mine. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 111–115.

ICRP 2007. *The 2007 Recommendations of the International Commission on Radiological Protection*. International Commission on Radiological Protection Publication 103, Elsevier Ltd.

Lawrence CE, Akber RA, Bollhöfer A & Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Martin P, Tims S, McGill A, Ryan B & Pfitzner K 2006. Use of airborne γ-ray spectrometry for environmental assessment of the rehabilitated Nabarlek uranium mine, northern Australia. *Environmental Monitoring and Assessment* 115, 531–553.

Radon exhalation from a rehabilitated landform

A Bollhöfer, B Ryan, A Esparon & J Pfitzner

Introduction

A trial landform has been constructed on the minesite to study the effects of different growth media on the establishment of vegetation, both via direct seeding and planting of tubestock. It is located immediately adjacent to the north west corner of the Ranger tailings dam and covers an area of approximately 8 ha. Radon exhalation measurements will be carried out through time on the different surface treatments to provide data for the development of a post rehabilitation radiological dose assessment model. For more detail on construction of the landform and erosion plots see Saynor et al (2009) and the erosion studies project report 'Revegetation trial and demonstration landform: erosion and chemistry studies (pp 109–112, in this volume).

In this project, radon (^{222}Rn) exhalation rates for various covers and vegetation types, taking into account weathering, erosion and compaction effects, and the effect of developing vegetation on the landform, will be determined. Specifically the ^{222}Rn exhalation from the four SSD erosion plots (30 m × 30 m) will be measured over several years.

Soil 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 main contributor to radiological dose (Supervising Scientist 2007), in particular in the vicinity of the rehabilitated landform, radon exhalation and its temporal variability need to be estimated. The radon exhalation rate may change as the final landform evolves after rehabilitation of the site, and the trial landform will provide a unique opportunity to determine this variability under experimental conditions over a period of several years. The project will enable ***eriss*** and ERA to predict a long-term radon exhalation flux from the rehabilitated landform and contribute to the development of closure criteria.

The project started prior to the building of the landform, to determine the radon exhalation baseline of the original, undeveloped trial landform footprint, since ^{222}Rn can diffuse from depths of several meters to the surface (with lower layers making a decreasing contribution). The diffusion of ^{222}Rn can generally be described by Fick's law and ^{222}Rn diffusion length for dry soils has been reported to be in the 1–2 m range (Graaf et al 1992) and 2–5 m for sandy type materials (Holdsworth & Akber 2004).

Methods

Conventional charcoal canisters (or 'radon cups) are used to measure radon exhalation. 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 given in Bollhöfer et al (2005) and Lawrence (2004).

To get a true average radon exhalation from the four erosion plots, radon cups were placed randomly and put onto rocks if required, sealing the rim using a putty. This is in contrast to many other studies where radon cups are put at 'convenient' spots where they can easily be embedded into the finer grained soil. Fine grained material exhibits higher radon exhalation flux densities than solid rock. Hence, results of radon exhalation measurements can potentially be skewed towards higher values if the sampling design is not random (Bollhöfer et al 2005).

Radon cups were deployed in August 2008 on an undisturbed area of the trial landform footprint and exposed for three days to determine the radon exhalation from the substrate underlying the constructed landform. As heavy machinery was already deployed and used for trial landform construction during the survey, the entire area was not accessible. It was planned to repeat the radon exhalation survey at the same area after the top 20 cm of soil were stripped, prior to construction of the trial landform. However the same area was not accessible after stripping. Consequently, a post-stripping survey was conducted approximately 200 m to the west, in a different area of the trial landform, from 13–16 October 2008. It is important to note that heavy rainfall occurred on 14 October 2009, while the radon cups were deployed.

Construction of the landform was finished 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. This watering has been done to aid the establishment of vegetation on the trial landform through the 2009 dry season. As soil moisture content has a substantial effect on radon exhalation (and because some of the areas were constantly wet) radon exhalation was measured from the four erosion plots only, which were not irrigated. In addition, there is the possibility that the irrigation water contained significant concentrations of radium, which would be adsorbed onto the surface of particulates in the top 5–10 cm of the soil (Akber & Marten 1992) and hence increase radon exhalation rates from the surface (Lawrence 2004). Consequently, radon exhalation measurements were not made on the irrigated landform areas.

The location and a description of the four erosion plots where measurements were performed is shown in Figure 1 and Table 1. Twenty radon cups were deployed randomly across each erosion plot and exposed for four days in May 2009. They were then collected and sent to the Darwin laboratories, where they were analysed using a NaI gamma detector.

Figure 1 shows the location and the distribution of measurement results for the May 2009 radon exhalation survey on the four erosion plots. Also marked are the locations of the 2008 pre-construction survey points (open red diamonds: pre-construction August 2008; open black diamonds: post stripping October 2008). The area where the post-stripping survey was conducted is now covered by waste rock and a 5 metre layer of a 30% laterite–waste rock mix, which has been irrigated over the past few months.

Progress to date

Radon flux densities from the pre-construction substrate follow a log-normal distribution with a range from 24 to 144 $mBq \cdot m^{-2} \cdot s^{-1}$. The geometric mean and the median of the pre-construction radon flux density are both 73 $mBq \cdot m^{-2} \cdot s^{-1}$. This is similar to late dry season radon flux densities of 64±25 $mBq \cdot m^{-2} \cdot s^{-1}$, which were previously determined in non-mineralised areas in the region (Todd et al 1998, Lawrence et al 2009).

Radon flux densities measured immediately after stripping the top 20 cm of soil ranged from 57 to 919 $mBq \cdot m^{-2} \cdot s^{-1}$. The geometric mean and median are 350 and 385 $mBq \cdot m^{-2} \cdot s^{-1}$, respectively, much higher than radon flux densities prior to the area being disturbed. This difference may have been caused by differences in soil radium concentration, but may also be due to the heavy early wet season rain occurring during the sampling interval. Similar spikes in radon exhalation after short but heavy tropical rainfall events in the region have been reported elsewhere (Lawrence et al 2009).

Another reason for the higher radon flux densities observed after stripping may be that steady state conditions had not been reached when the measurements were conducted. Large radon concentration gradients at the soil/air interface immediately after stripping may have lead to an increase in radon flux densities (see Porstendörfer 1994 for background). It was not possible to repeat these measurements later to test the assumption of non-equilibrium owing to the covering of the area with waste rock. As a result of this uncertainty the radon flux densities measured after stripping are not deemed representative for the substrate at the sampling sites investigated. A pre-construction radon flux density of 73 mBq·m^{-2}·s^{-1} is thus assumed for the trial landform.

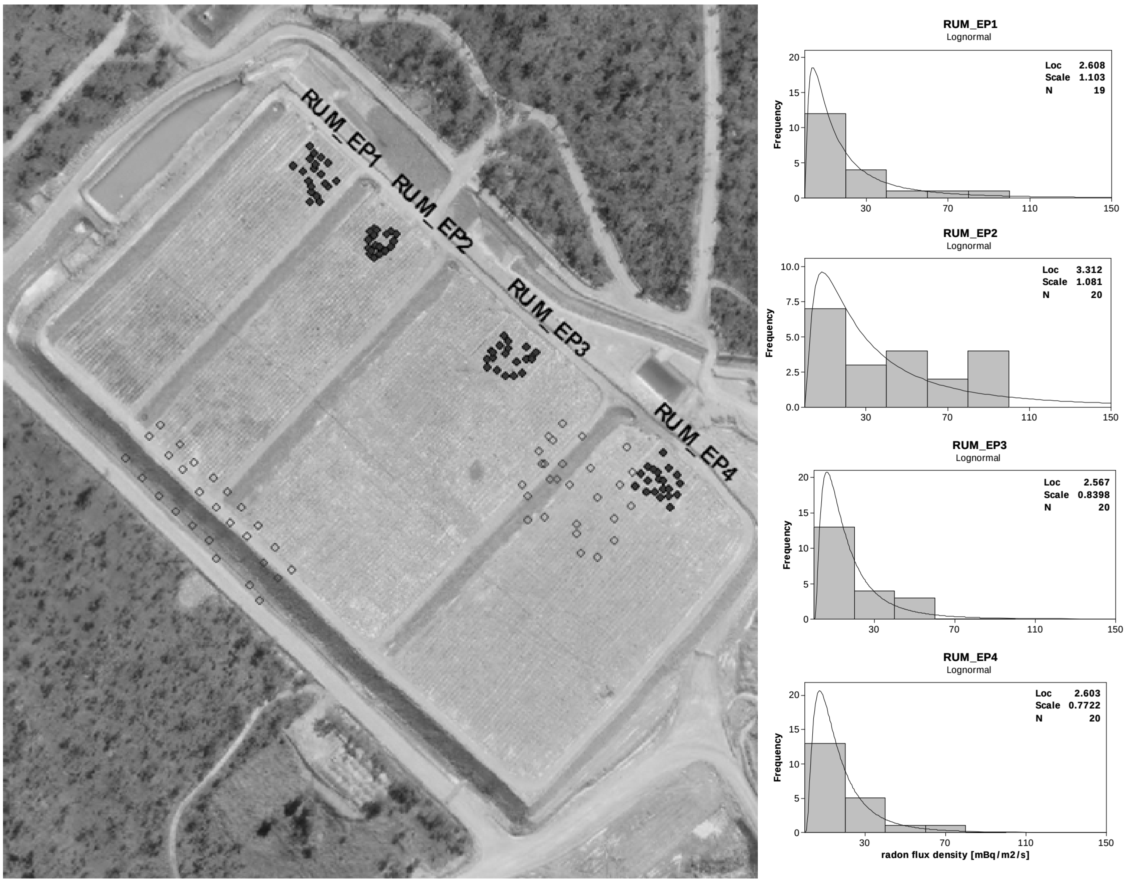


Figure 1 Locations of the radon exhalation measurements conducted in May 2009 (black diamonds). Open diamonds represent the location of the radon cups deployed in August and October 2008, respectively. In addition, the right hand side shows the distribution plots of radon flux densities measured at the four erosion plots (EP1–4).

Generally, radon flux densities measured from the four un-irrigated erosion plots are much lower than those measured on the ground prior to landform construction (Table 1). This can be explained by the rocky nature of the trial landform with typical rock sizes reaching up to 300 mm in diameter and larger. It has previously been reported that radon exhalation from fine grained soils is typically much larger than for solid rock (Lawrence et al 2009).

Radon exhalation from the 30% laterite–waste rock mix appears to be more uniform and exhibits a smaller standard error, reflecting the fact that the material in this area was mixed with a back hoe after dumping. In contrast, the waste rock only was dumped without any further mixing and consequently standard errors are larger. Radon exhalation from erosion plot 2 is approximately twice as high as the other three erosion plots, most likely due to heterogeneity in soil radionuclide activity concentrations across the trial landform. Soils from the four erosion plots were sampled for laboratory characterisation, and radionuclide activity concentration measurements were underway at the time of completion of this report.

Table 1 Description of the four erosion plots and average radon flux densities

	Treatment	^{222}Rn flux density [$mBq \cdot m^{-2} \cdot s^{-1}$] Arithmetic (*geometric*) average ± std error
RUM_EP1	Waste rock material planted with tube stock	22 (*14*) ± 11
RUM_EP2	Waste rock material planted by direct seeding	41 (*26*) ± 16
RUM_EP3	30% lateritic material mixed with waste rock (2 m), direct seeding	19 (*13*) ± 7
RUM_EP4	30% lateritic material mixed with waste rock (2 m), tube stock.	18 (*13*) ± 7

Future work

Radon exhalation surveys across the four erosion plots will be conducted every 4 months to investigate seasonal and 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 for a range of size fractions.

References

Akber R & Marten R 1992. Fate of radionuclides applied to soil in Ranger Uranium Mine land application area. In *Proceedings of the workshop on land application of effluent water from uranium mines in the Alligator Rivers Region*. Supervising Scientist for the Alligator Rivers Region, Australian Government Publishing Service, Canberra, 139–165.

Bollhöfer A, Storm J, Martin P & Tims S 2005. Geographic variability in radon exhalation at a rehabilitated uranium mine in the Northern Territory, Australia. *Environmental Monitoring and Assessment* 114, 313–330.

Graaf ER, Heijs S, Meijer RJd, Put L W & Mulder HFHM 1992. A facility to study transport of radon in soil under controlled conditions. *Radiation Protection Dosimetry* 45, 223–226.

Holdsworth S & Akber R 2004. Diffusion length and emanation coefficient of Rn-220 for a zircon and monazite sample. *Radiation Protection in Australia* 21, 7–13.

Lawrence CE 2004. Measurement of ^{222}Rn exhalation rates and ^{210}Pb deposition rates in a tropical environment. PhD thesis. School of Physical and Chemical Sciences, Queensland University of Technology, Brisbane.

Lawrence CE, Akber RA, Bollhöfer A & Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Porstendörfer J 1994. Properties and behaviour of radon and thoron and their decay products in the air. *J. Aerosol Sci.* 2, 219–263.

Saynor MG, 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, 12–19.

Supervising Scientist 2007. *Annual Report 2006–2007*. Supervising Scientist, Darwin.

Todd R, Akber RA & Martin P 1998. ^{222}Rn and ^{220}Rn activity flux from the ground in the vicinity of Ranger Uranium Mine. Internal report 279, Supervising Scientist, Canberra. Unpublished paper.

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

C Humphrey, D Jones D & K Turner

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 ARR so as to provide a basis for surface water quality closure criteria for Georgetown and Coonjimba Billabongs.

After the Ranger mine ceases operations, disturbed areas need to be rehabilitated to a condition consistent with the values of Kakadu National Park (KNP), and to be suitable for reincorporation 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.

A concern with mine-site closure, is the potential for delivery of solutes from the rehabilitated mine landform. These solutes, if present at too high a loading, could affect water quality and hence impact on the ecological values of adjacent waterbodies. Hence a key objective of the closure planning process will be to produce a design for the current disturbed area such that the delivery of solutes and suspended sediment from the disturbed footprint in the mine site catchments (eg Corridor and Georgetown Creeks) will not compromise the post closure environmental objectives for the waterbody.

In this project, monitoring data derived from lentic macroinvertebrate communities are being used as one approach towards developing closure criteria for relevant water quality indicators in waterbodies immediately adjacent to the mine site. Georgetown Billabong (GTB), the largest waterbody located in close proximity to the current operational mine area, serves as an example of how closure criteria would be derived using macroinvertebrate data. 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). Specifically, if the post-closure condition in Georgetown Billabong is to be consistent with similar undisturbed (reference) billabong environments of Kakadu, then the range of water quality data from the billabong over time that supports such an ecological condition in Georgetown Billabong (as measured by suitable surrogate biological indicators) may be used for this purpose.

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. The environmental objectives post closure would be expected to embed the similar maintenance of this biological diversity (ie values consistent with those of similar, undisturbed habitats of KNP). However, in 2006, macroinvertebrates were collected from both macrophyte and benthic (sediment) habitats with the samples from each habitat being processed separately. The aggregated data showed essentially no difference to that of the reference billabongs and hence were in agreement with the results from the previous surveys. However, when the macrophyte and benthic data sets were analysed separately it was found that, in contrast to communities from the macrophyte habitat, the sediment-dwelling communities were less diverse in GTB than in reference waterbodies (Humphrey et al 2008).

In 2007–08, an investigation commenced to determine whether this lower diversity was mine or habitat-related, through studies of sediment chemistry and sediment physical structure. Sediment samples collected from littoral areas of the billabongs in 2007 showed levels of U (~42 mg/kg) higher than values measured in samples collected from references waterbodies at the same time, and higher than measured in GTB over the past 25 years by ERA's routine monitoring program and by others (Humphrey et al 2009). In particular there appeared to have been a substantial rise in U between 2001 and the present.

Because of (1) a substantial gap of 5 years in the collection of U 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 substantial rise in U in the billabong as a whole. To address these questions, the following investigations were identified: (i) a number of GTB sediment samples collected in 2006 by EWLS should be reanalysed using similar methods to the 2007–08 study to establish whether any increase in U concentration was evident, and to quantify dependence of U extraction on method used, and (ii), sampling should be conducted in 2009 across a transect of the billabong to investigate U levels as a function of location across the billabong.

GTB sediments consist mainly of fine cracking clays, and are generally devoid of surface vegetation during the dry when the sediment exposed around the gently sloping margins undergoes desiccation-induced cracking. Of the billabongs sampled, these conditions are 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. Indeed, and as reported in 'Effects of fine suspended sediment on billabong limnology', (pp 143–149, in this volume), Walker et al (1982) also noted the very fine particle size of suspended sediments in GTB relative to other billabongs studied.

To further investigate the nature of GTB sediments, the dependence of U concentration on particle size, as well as mineralogical characterisation, was also identified as a study requirement to be carried out for selected samples collected in 2007. The results of these investigations, together with a dedicated field assessment of U sediment toxicity proposed for a three-year period commencing in May 2009 (see 'The toxicity of uranium (U) to sediment biota of Magela Creek backflow billabong environments', pp 37–45, in this volume) would determine whether or not the low benthic diversity in GTB is related to current mining activities. If unrelated to mining activities, it would confirm the findings from the previous surveys, that macroinvertebrate communities in the billabong have remained unchanged since mining commenced, despite there having been an increase in solute loads between 1996 and 2006.

Pending the outcome of sediment-related studies described above and for now, the water quality record for 2006 will be included in the database being used to specify an acceptable ceiling water quality sufficient to protect the health of the waterbody post closure. This will enable the derivation of provisional water quality closure criteria for the water column. Of course derived numerical targets based on a limited range of measured water quality parameters should be accompanied by the caveat that periodic surveys of macroinvertebrate community will still need to be carried out to confirm that no other types of impacts from the rehabilitated landform were affecting the biological integrity of the waterbody. (Any new data arising from periodic sampling will also contribute to a rolling assessment and enable refinement of the water quality closure criteria.) The results from the U sediment toxicity study referred to above will also inform the setting of water quality criteria for GTB, ensuring criteria were not relaxed to thresholds that could adversely impact upon sediment quality, by virtue of the adsorption to sediment of U initially present in the water column.

Progress to date

There have been delays in aspects of the proposed work program for 2008–09 because of higher priorities in the environmental chemistry program in this period (including continuous monitoring) and because of new resources required to establish the related U-spiking sediment experiment in Gulungul Billabong in the second half of the reporting period. The following reports on progress against the study objectives for 2008–09:

1. Assess physical sediment characteristics of billabongs from samples collected in 2006:

   This work has been delayed. However, a penetrometer was acquired in the reporting period that will be used in subsequent dry seasons to measure the extent of compaction of littoral sediment (expected to be greatest in GTB and Coonjimba Billabongs compared with reference billabongs).

2. Carry out sediment digest method comparisons to determine if the historical data can be 'normalised' by sediment digest efficiency factors, thereby removing or reducing the 'noise' introduced by the use of different methods through time:

   Study component delayed.

3. Process and chemically analyse archived sediment samples collected from GTB in 2006:

   Samples acquired from EWLS but not yet processed.

4. Collect sediment samples from GTB in May 2009 to quantify the extent of cross billabong variation of metal concentrations:

   Samples collected, laboratory processed and submitted for chemical analysis.

5. Derive and report provisional water quality closure criteria for the water column:

   Reporting delayed but provisional criteria up to and including 2006 reported by Jones et al (2008).

References

ANZECC & ARMCANZ 2000. *Australian and New Zealand guidelines for fresh and marine water quality. National Water Quality Management Strategy Paper No 4*. Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

Humphrey C, Turner K & Jones D 2009. Developing water quality closure criteria for Ranger billabongs using macroinvertebrate community data. In ***eriss** research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 130–135.

Jones D, Humphrey C, Iles M & van Dam R 2008. Deriving surface water quality closure criteria – an Australian uranium mine case study, In *Proceedings of Minewater and the Environment*, 10th International Mine Water Association Congress, eds N Rapantova & Z Hrkal, June 2–5, Karlovy Vary, Czech Republic, 209–212.

Walker TD & Tyler PA 1982. Chemical characteristics and nutrient status of billabongs of the Alligator Rivers Region, NT. Open file record 27, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Effects of fine suspended sediment on billabong limnology

D Buckle, C Humphrey & D Jones

Background and aims

This paper provides a status report for a study designed to progress the development of water quality criteria relevant to both mine operations and mine closure: the effects of fine suspended sediment on billabong limnology.

Specifically, 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 arising from both historic as well as new investigations.

The data arising from the study will be used as:

i a component of a multiple lines of evidence approach to assess direct effects of suspended sediment on tropical freshwater biota (complementing current laboratory assessment approaches on this subject ('The effects of suspended sediment on tropical freshwater biota', pp 32–36, in this volume).

ii assist with developing the suspended sediment/turbidity component of surface water quality closure criteria for billabongs adjacent to the Ranger mine site.

Suspended sediment has been identified as an aquatic ecosystem stressor that will most likely assume greater significance in the future, as a consequence of rehabilitation works on the Ranger site (see The effects of suspended sediment on tropical freshwater biota', pp 32–36, in this volume). Billabongs immediately downstream of Ranger, in particular, are at greatest risk from erosion of fine particulate matter from newly-rehabilitated landforms. Historical evidence for high suspended solids loadings and significant sedimentation that can arise in local billabongs, as a consequence of the failure of new earthwork structures, has been reported by the SSD and its consultants (Humphrey 1985, Nanson et al 1990) in relation to an intense rain-storm event that occurred over the Ranger site in February 1980 (affecting Coonjimba Billabong).

This project aims to draw upon field-effects and observational limnological data from (mainly) local billabongs from past and new studies to assist the development of operational water management triggers and closure criteria for suspended sediment. The experimental focus of the investigations will be the tracking of natural increases in turbidity that are observed in shallow backflow billabongs adjacent to Ranger over the dry season and the changes in billabong limnology that are associated with this.

Turbidity may disrupt a number of ecosystem functions, one of the best-documented of these being primary (plant) production through inhibition of light. By continuous measurement of dissolved oxygen and turbidity, together with regular (grab-sample) measurement of suspended sediment (0.45–63 μm), chlorophyll-a (phytoplankton abundance) and total organic carbon (produced by decay of plant matter and contributing to biological oxygen demand), a threshold in turbidity may be inferred at which important billabong photosynthetic functions are disrupted. This threshold may provide complementary data (to those derived from laboratory studies) upon which to derive operational and closure criteria for suspended sediment in local billabongs.

For this study, data from previous limnological studies conducted in the region are being reviewed to examine possible relationships between turbidity, chlorophyll-a and dissolved oxygen. A field investigation is also being conducted in Georgetown Billabong using in-situ continuous (5 minute intervals) measurement of dissolved oxygen and turbidity, near-surface and at depth, for the period May to October/November 2009 (encompassing the period of significant dry season increase in turbidity in the billabong). Depth profiles of water quality parameters, and surface and depth samples for chlorophyll-a and total organic carbon determinations, have been collected fortnightly over this period.

This project links to laboratory-based concentration-response experiments that are being developed for suspended sediment (Ecotoxicology group) as well as to relevant activities of the Hydrological & Geomorphic Processes (HGP) group so as to ensure maximum relevance of outputs/outcomes.

Methods

Sampling commenced on 27 May 2009 from one sampling site in the deeper waters located near the outflow of Georgetown billabong (S 12^{0}40.705', E 132^{0}55.885') (Map 2). At this site, three sub-sites were established for continuous and spot sampling by use of a floating pontoon secured in place to prevent movement and disturbance of sub-sites. Sub-site 1 was directly attached to the pontoon to measure surface water conditions (10-20 cm depth), while sub-sites 2 and 3 were located approximately 10 m apart near the bottom of the water column. They were fixed $\approx$ 20 cm above the sediment bed using custom designed (tripod) frames to both secure in place and protect (from crocodiles) the sampling and monitoring equipment. Each sub-site was equipped with a Hydrolab datasonde 5X multiprobe to measure temperature, dissolved oxygen, turbidity, EC and pH at five minute intervals. Data were downloaded frequently via mobile phone connection to track changes in the measured parameters, and to detect any equipment malfunction.

More detailed sampling was conducted each fortnight between 8:30 am – 9:30 am to standardise with the same time of day that sampling was conducted in baseline studies conducted in 1980 and 1981 (eg Humphrey & Simpson 1985) and in subsequent routine water quality monitoring carried out by ERA. Water samples, from all sub-sites, were collected each fortnight for measurements of suspended sediment, total (TOC) and dissolved (DOC) organic carbon and chlorophyll-a. Filtered water samples were obtained monthly from all sub-sites for analysis of metals (Al, Cu, Fe, Mn, Pb, U, Zn), ions (Ca, Mg, SO_4) and nutrients (NO_3, PO_4, NH_3).

An additional mid-depth (1.0-1.25 m deep) sample of chlorophyll-a was collected below sub-site 1. This mid-depth sample aimed to provide information on phytoplankton biomass at different depths.

The fortnightly visits also included collection of water quality profile data near sub-sites 2 and 3. Measurements were made at 10 cm below the surface, 25 cm, and then every 25 cm until the bottom of the billabong was reached using pre-measured cable to lower the measuring probes (depth was 2.65 m on 27/5/2009). Parameters measured included, temperature, dissolved oxygen, turbidity, EC and pH using a hydrolab quanta multiprobe and photosynthetically-active radiation (PAR) using a LI-193 underwater Spherical Quantum Sensor. The LI-193 measures photon flux from all directions or Photosynthetic Photon Flux Fluence Rate (PPFFR) and quantifies the amount of light available for photosynthesis.

Results to date

Data are still being collected for this project, thus results presented below are preliminary and subject to change with additional information.

Changes in turbidity over time

Historical turbidity data collected in 1980 and 1981 by Humphrey and Simpson (1985) and between 1982 and 2008 by Energy Resources of Australia (ERA), as well as data collected during the current project, are graphed for each month in Figure 1. The historical data indicate that turbidity during 1980 and 1981 increased earlier in the dry season and with subsequent years increased only towards the very end of the dry season. The increased turbidity early in the dry season in 1980 and 1981 is presumably due to the presence of water buffalo. Water buffalo were mostly removed by 1982 resulting in reduced turbidity early in the dry season. Increased turbidity late in the dry season since the removal of buffalo, is most likely due to the reduction in water levels to quite shallow depths (eg depth reduced from 2.65 m on 27/5/09 to 1.68 m on 29/10/09 in the deepest section of the billabong) coupled with wind or biogenic re-suspension of fine sediments. Whilst the magnitude of wet season rainfall and duration could also cause variation in turbidity events from year to year, it is unlikely to account for the early elevation of turbidity observed in 1980 and 1981. Both these years (1979–80 and 1980–81) recorded above average rainfall, with subsequent years representing both below and above average rainfall (see Figure 2.1 p 10 in Supervising Scientist 2009).

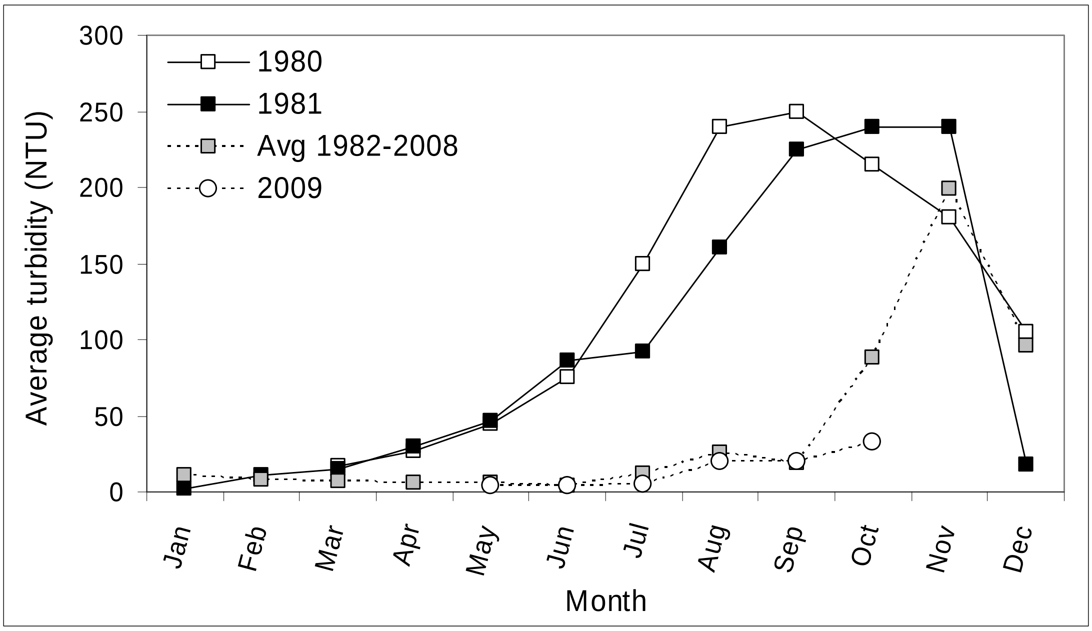


Figure 1 Average monthly Turbidity (NTU) for Georgetown Billabong from 1980 to 2009. ***eriss*** data used for 1980, 1981 and 2009. Energy Resources of Australia data used for 1982–2008.

The data collected in 2009 are consistent with the seasonal pattern of post-buffalo turbidity in Georgetown Billabong. The mean value recorded in October 2009 is below the average for 1982 to 2008 (Figure 1); however, and while data for individual years are not shown, the October mean value is consistent with values for a number of individual years since the removal of water buffalo.

The shorter time period of enhanced turbidity in Georgetown Billabong since the early 1980s (restricted now to the late dry) may, in part, be responsible for the increase in density of aquatic vegetation observed since the removal of water buffalo (Finlayson et al 1994). This suggestion is supported by Dunlop et al (2005) who reported the absence of submerged aquatic vegetation in the Condamine Balonne River (Murray-Darling catchment) when turbidity was greater than 20-30 NTU.

The changes in water clarity and billabong aquatic plant communities discussed above make a number of the findings from the earlier detailed limnological studies (1978–1980) such as those of Walker and Tyler (1982), Walker et al (1982), Walker & Tyler (1983) and Walker et al (1984), not so applicable to early dry season months under current conditions.

Suspended sediment/turbidity relationship

Data obtained during the 2009 dry season have been used to derive a preliminary relationship between turbidity and suspended sediment (SS) in Georgetown Billabong. There is a strong relationship between SS and turbidity ($R^2 = 0.937$), as shown in Figure 2. However, the line of best fit is not quite linear, most likely due to the >0.45 µm filter not capturing ultra fine particles, thus underestimating the total weight of SS. In support of this, Walker et al (1982) found a 0.1 µm filter was required to provide maximum recovery of SS in Georgetown Billabong due to the increased presence of very fine particles relative to other billabongs studied. Future and stored <0.45 µm samples will be further filtered through 0.1 µm filter papers to better understand the potential influence of ultra fine particles in the current SS sample estimates.

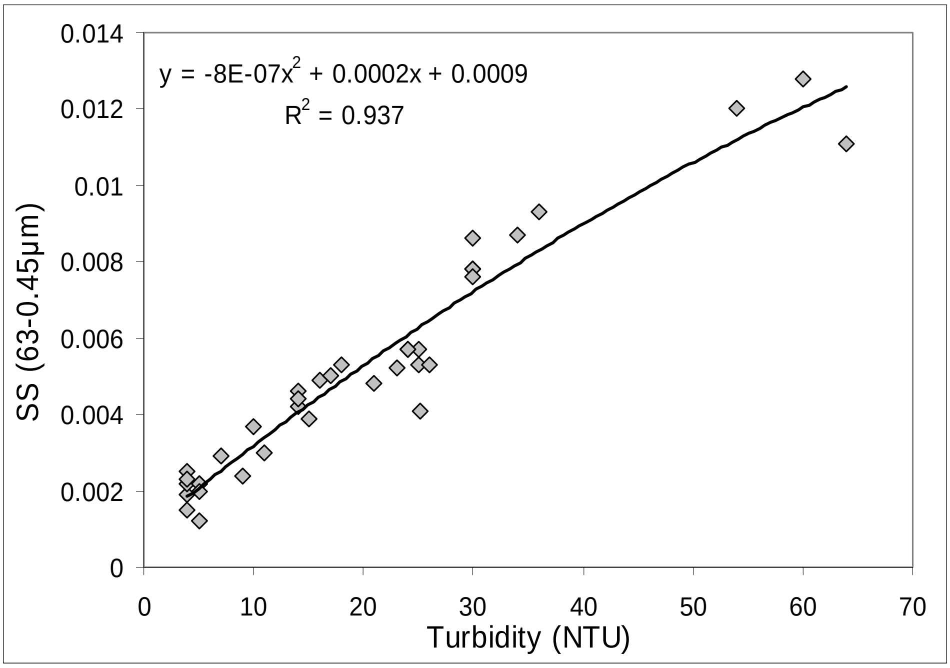


Figure 2 Suspended Sediment (0.45–63 µm) relationship to turbidity (NTU) in Georgetown Billabong during the dry season of 2009

Alternatively or in addition, the slightly non-linear relationship between SS and turbidity could be associated with an increase in the contribution of organic matter (algae etc) at higher turbidities, since organic matter is lighter than inorganic matter.

Turbidity influence over phytoplanktonic productivity

Chlorophyll-a (mg/m^3) is commonly used as a direct measure of phytoplanktonic biomass. For this project it was measured close to the water surface (10–20 cm depth), middle water column and 20 cm above the sediment bed. at fixed sampling locations. Only surface water data are presented here, data from other depths will be analysed at the completion of the project.

Surface chlorophyll-a steadily increased after the sampling program commenced with a peak observed on the 20/8/2009 at 35 mg/m^3 (24 NTU), followed by slightly reduced concentrations, then a second larger peak on the 15/10/2009 (72 & 65 mg/m^3, duplicate samples at 30 NTU). The latter sharp rise in chlorophyll-a followed a short-duration spike in turbidity (54 NTU) recorded on the 1/10/2009. The short duration of the turbidity spike prevented any assessment of the effects on chlorophyll-a over a prolonged period of reduced light availability (Figure 3), but rather may have provided a stimulus for algal growth by way of additional (suspended) nutrients.

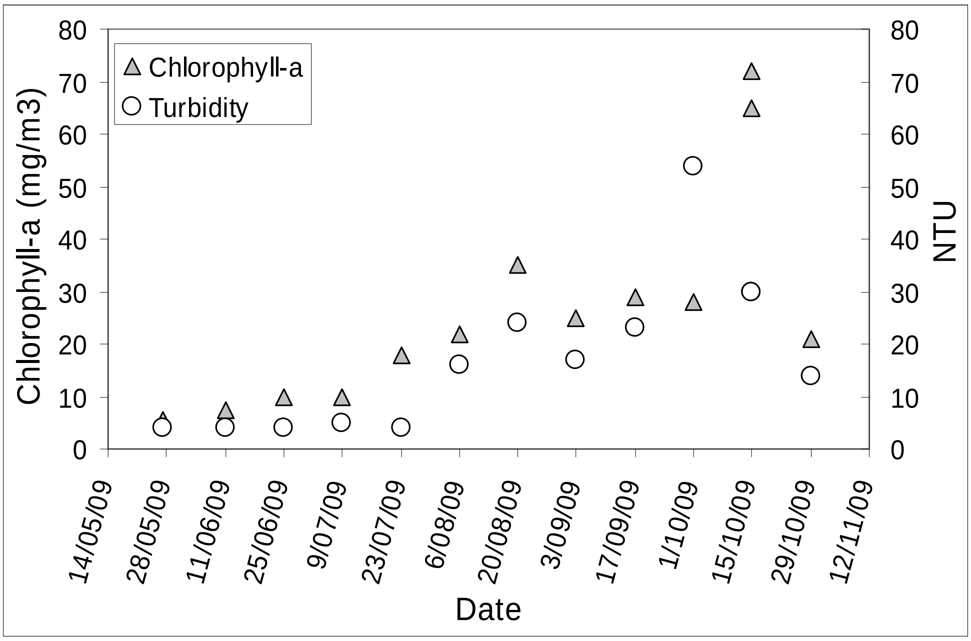


Figure 3 Turbidity (NTU) and chlorophyll-a (mg/m^3) in the surface waters (< 20 cm) of Georgetown Billabong in 2009

The (skewed) unimodal relationship observed between near-surface chlorophyll-a and turbidity from data collected in 1981 by Humphrey and Simpson (1985) suggest inhibited phytoplankton production at turbidity values around 50 NTU (Figure 4). While a similar relationship is less evident in 2009 due to the low turbidity conditions observed in this year (Figure 1) (only one turbidity value recorded above 30 NTU, Figure 3), the data collected in 2009 nevertheless appear to support the historical (1981) relationship and threshold effect. Given the lack of sampling events in 2009 for turbidity values above 30 NTU and the changed turbidity conditions observed since 1981 (Figure 1), particularly the seasonal timing, further chlorophyll-a and turbidity data are required for periods when turbidity is above 30 NTU to better define the relationship.

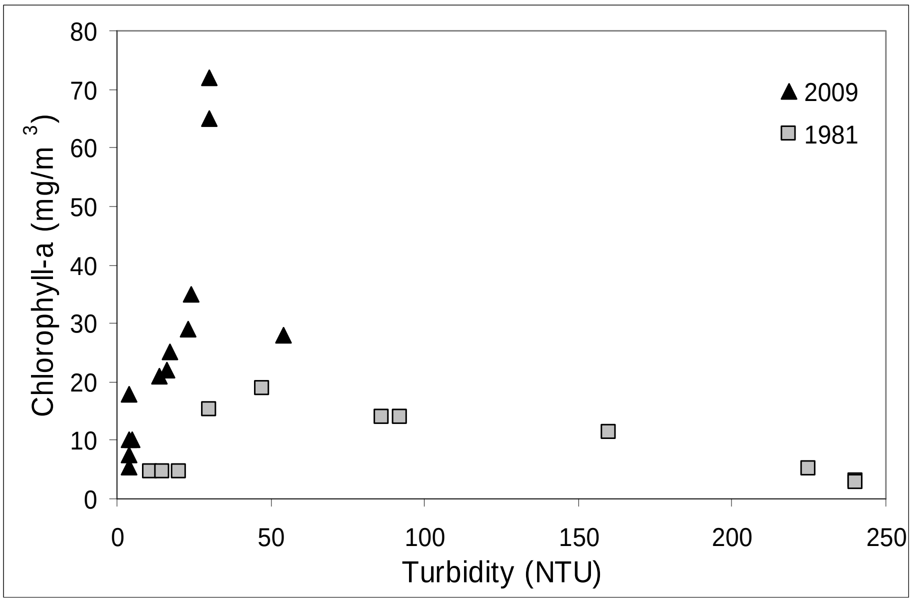


Figure 4 Relationship between turbidity (NTU) and chlorophyll-a (mg/m³) in the surface waters (10–20 cm) of Georgetown Billabong in 1981 and 2009

Further work

Data presented above are a subset of the full dataset collected. Analysis of the full dataset is yet to be completed and as such, the inferences that have been presented above are preliminary.

Preliminary recommendations

Further samples from Georgetown Billabong are required for turbidity conditions above 30 NTU to better define the relationship between turbidity and chlorophyll-a concentrations.

If an improved understanding of the slightly non-linear relationship between suspended sediment and turbidity is required, then the following analysis on any future samples is recommended:

- The residue from future 0.45–63 μm samples should be analysed for turbidity then further filtered using 0.1 μm filters to determine if the loss of ultra fine particles changes with increasing turbidity.
- Assessment of the proportion of organic and inorganic material contributing to the suspended sediment should be further considered. This information may also provide insights into the effect of algal self shading.

References

Dunlop J, McGregor G & Horrigan N 2005. Potential impacts of salinity and turbidity in riverine ecosystems: characterisation of impacts and a discussion of regional target setting for regional ecosystems in Queensland. National Action Plan for Salinity and Water Quality, Water Quality State Level Investment Project 6. [web] http://www.wqonline.info/Documents/Report_WQ06_Review_reduced.pdf.

Finlayson CM, Thompson K, von Oertzen I & Cowie ID 1994. *Vegetation communities of five Magela Creek billabongs, Alligator Rivers Region, Northern Territory*. Technical memorandum 46, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra.

Humphrey CL 1985. Recent history of Coonjimba Billabong. In *Alligator Rivers Region Research Institute Annual Research Summary 1984–85*. Supervising Scientist for the Alligator Rivers Region, Australian Government Publishing Service Canberra 1985, 48–49.

Humphrey CL & Simpson RD 1985. The biology and ecology of *Velesunio angasi* (Bivalvia: Hydiidae) in the Magela Creek, Northern Territory (4 parts). Open file record 38, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Nanson GC, East TJ, Roberts RG, Clark RL & Murray AS 1990. Quaternary evolution and landform stability of Magela Creek catchment, near the Ranger Uranium Mine, northern Australia. Open file record 63, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Supervising Scientist 2009. *Annual Report 2008–2009*. Supervising Scientist, Darwin.

Walker TD, Kirk JTO & Tyler PA 1982. The underwater light climate of billabongs of the Alligator Rivers Region, NT. Open file record 20, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Walker TD & Tyler PA 1982. Chemical characteristics and nutrient status of billabongs of the Alligator Rivers Region, NT. Open file record 27, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Walker TD & Tyler PA 1983. Primary productivity of phytoplankton in billabongs of the Alligator Rivers Region. Open file record 8, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Walker TD, Waterhouse J & Tyler PA 1984. Thermal stratification and the distribution of dissolved oxygen in billabongs of the Alligator Rivers Region, NT. Open file record 28, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Use of vegetation analogues to guide planning for rehabilitation of the Ranger mine site

C Humphrey & G Fox

Background and aims

This study aims to classify vegetation communities from a variety of potential analogue sites in the ARR, model these communities in relation to environmental factors, then apply the model as a tool for assessing the success of revegetation of the post-mine landform at Ranger.

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 targets for performance measures 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 (parent material, 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 then be specified for the rehabilitated landform at Ranger.

Data previously obtained by ***eriss*** and Earth Water Life Sciences (EWLS) 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.

For the 2006–07 and 2007–08 periods, the study focus was on additional characterisation of target plant communities (especially *Melaleuca* communities), gathering of soil physico-chemistry data for analogue sites and modelling (by EWLS) of key plant-landform relationships. The data sets produced by ***eriss*** and EWLS were combined and analysed using multivariate techniques. A preliminary plant classification was derived, with key distinguishing species identified for the main classification groups. The occurrence of these key species were modelled according to major landform characteristics.

The modelling conducted by EWLS did not include the full suite of soil physical and chemical characteristics that had been collected for (a) the analogue sites, nor (b) the soil/substrate media occurring on previous revegetation test sites on Ranger waste rock. It did not, most importantly, include the characteristics of the type of substrate (mixture of waste rock and finer-grained weathered horizon material) being proposed in the rehabilitation plan for the site. The principal aim of future work on the vegetation analogue project is to include in the development of a predictive model, all available environmental (soil and landform) and vegetation data for both analogue and Ranger revegetated sites.

In the first part of 2008–09, it became apparent that it might not be possible to source the complete set of landform variables (climate and water balance, local topography, as well as fire disturbance) that had been used in the previous modelling of the analogue sites by former EWLS staff. Instead and as an interim analysis procedure until the complete environmental dataset could be acquired, analyses conducted during this period focused on identifying possible soil property-plant relationships for the 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).

Progress

Since ARRTC 23 (March 2009), the following issues have been raised and/or addressed:

1. Additional plant survey data became available as a consequence of studies conducted to assess the impact of Cyclone Monica on the vegetation of impacted areas in the ARR (Staben et al 2009). The opportunity, therefore, was taken to add the data for an additional 55 sites (including sites from Gulungul Creek catchment, Nabarlek and Ranger minesite revegetation trials ('Heritage' area)) to the existing combined ***eriss*** and EWLS ARR plant classification to both further validate the integrity of the original classification, and seek additional classification groups that might need to be considered for the final landform design.
2. A large number of soil samples were collected by EWLS in 2009 from the newly-constructed trial landform at Ranger and from existing, constructed (mine-derived) substrates that have provided a medium for vegetation growth on various historical rehabilitation trial sites on the Ranger mine site.
3. As stated above, only weak relationships between soil properties and plant communities for the natural analogue sites have been found. However, a published popular account by a long-term naturalist working in the ARR (Woerle 1987) makes specific reference to deep and shallow soils characteristic of the mixed eucalypt woodland and dry mixed eucalypt woodland classification units, respectively, identified in earlier analogue work (Table 1). K Brennan (NT Government plant ecologist, pers comm) also considered that gross soil profile features similar to those described by Woerle (1987) should be able to distinguish the same two major classification groups.
4. Potential problems have been identified with the landscape physical characterisation variables (eg slope angle and length) produced from past vegetation and landscape modelling conducted by EWLS (Hollingsworth et al 2007, Hollingsworth & McGovern 2004). This work used a synthesised Digital Elevation Model (DEM), the source components of which are being reassessed for accuracy and applicability by ERISS. The results from the original analysis both informed selection of an analogue landform similar in size and shape to a conceptual model of the Ranger final landform, but perhaps more importantly, were used to specify the physical design criteria for the rehabilitated landform.

The following reports on progress against the four issues identified above.

1 Additional plant survey data

After standardising survey data to the same areal units used in earlier surveys, the additional plant survey data for trees and shrubs arising from Staben et al's (2009) study were added to the existing ***eriss*** and EWLS plant analogue data. Group average cluster analysis was conducted on the combined data using the PRIMER (v6) multivariate software package (Clarke & Gorley 2006). The new (re-)classification is shown in Figure 1 where site labels indicate the original three dominant vegetation classification classes (C1–C3, described in Table 1). All of the original C1–C3 sites reclassify according to their original vegetation classes. Further, additional site data are now available for the *Melaleuca* woodland classification group (C1), formerly with very few representative sites. Despite the data of Staben et al's (2009) being confined essentially to Koolpinyah lowlands (with no hill slopes representing classification C3 sites), their geographical extent is considerable. Incorporation of the additional data in the classification analysis has further confirmed the discrete and dominant vegetation units identified in previous analogue work (Table 1).

Table 1 Descriptions of the analogue communities identified in this study and, where available, the matching vegetation units according to Schodde et al (1987)

Broad vegetation community	Dominant and/or distinguishing tree or shrub species	Classification unit from this study (Fig 1A)	Vegetation units used by Schodde et al (1987)
Melaleuca woodland	*Melaleuca viridiflora* *Pandanus spiralis*	C1	Myrtle–Pandanus savannah
Mixed Eucalypt woodland	*Eucalyptus miniata* *Eucalyptus tetrodonta* *Corymbia porrecta* *Xanthostemon paradoxus* *Acacia mimula*	C2	Open forest
Dry mixed Eucalypt woodland	*Corymbia foelscheana* *Xanthostemon paradoxus* *Erythrophleum chlorostachys* *Eucalyptus tectifica* *Cochlospermum fraseri*	C3	Woodland

2 Additional mine-derived soil samples

These samples, from the trial landform and other minesite revegetation plots, were collected by EWLS and have been submitted for analysis of the same suite of soil quality variables used by Humphrey et al (2009) in their assessment of the influence of soil properties on analogue vegetation patterns. The data were not available at the time of writing of this report.

3 Potential soil-vegetation relationships

Anecdotal evidence suggests that soil properties should be able to be used to distinguish the three dominant vegetation communities in the analogue classification, despite general lack of correlation with the parameters that have been assessed to date. Further evaluation of the Georgetown analogue area will be conducted in 2009–10 to assess whether, by restricting the comparison with soil properties to an area where all three vegetation classification units are represented, that other sources of variation that may have confounded previous comparisons can be removed, Moreover, additional multivariate techniques will be used to explore potential soil-plant relationships not identified in the previous (PRIMER BIOENV-based) analyses.

4 Derivation of landscape variables used in past vegetation and landscape reconstruction modelling

Given the comparatively low relief of most of the analogue sites, concerns have been raised by ***eriss***'s SSDI group (J Lowry, pers comm) that the 50-metre DEM created of the analogue areas, and used previously to derive a range of key environmental topographic variables, may have been deficient in resolution and may not fully support the results that were derived; thus, current modelling may not be as accurate as it could be. It is proposed that a full suite of landscape variables be re-derived for inclusion in the modelling using an updated and validated DEM with appropriate resolution.

Summary of further studies required for 2009–10

A much more expanded data set than that used by EWLS will be used to investigate relationships between plant and environmental data. In particular, the expanded modelling will include current soil description data, possible additional data acquired as a consequence of issue 3 from above, as well as those from the newly-constructed trial landform, and from existing, constructed (mine-derived) substrates that have provided a medium for vegetation growth on various historical rehabilitation trial sites on the Ranger mine site. As noted above (issue 4), a full suite of landscape variables will also be re-derived for inclusion in the modelling using an updated DEM. Collectively, these studies should provide more robust identification and characterisation of target plant communities to provide the basis for specification of closure criteria and for design of post-rehabilitation monitoring programs.

References

Clarke KR & Gorley RN 2006. *Primer v6: User Manual/Tutorial, Primer E: Plymouth*. Plymouth Marine Laboratory, Plymouth, UK.

Hollingsworth ID, Humphrey C & Gardiner M 2007. Revegetation at Ranger: An analysis of vegetation types and environmental trends in analogue areas. EWL Sciences Pty Ltd. Darwin.

Hollingsworth ID & McGovern E 2004. Landscape reconstruction at Ranger Mine – approach and current status. Discussion Paper ARRTC Meeting, March 2004. EWL Sciences Pty Ltd.

Humphrey C, Fox G & Lu P 2009. Use of vegetation analogues to guide planning for rehabilitation of the Ranger minesite. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 136–146.

Schodde R, Hedley AB, Mason IJ & Martensz PN 1987. Vegetation habitats, Kakadu National Park, Alligator Rivers Region, Northern Territory, Australia. Final report to Australian National Parks and Wildlife Service, CSIRO Division of Wildlife and Rangelands Research, Canberra.

Staben G, Saynor MJ, Moliere DR, Hancock GR & Evans KG 2009. Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on Gulungul Creek catchment, Ranger mine site and Nabarlek area. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 12–19

Woerle F 1987. Ranger's territory: Adventure in Australia's far north, the story of Frank Woerle as told to Colin Thiele. Angus & Robertson, North Ryde, New South Wales.

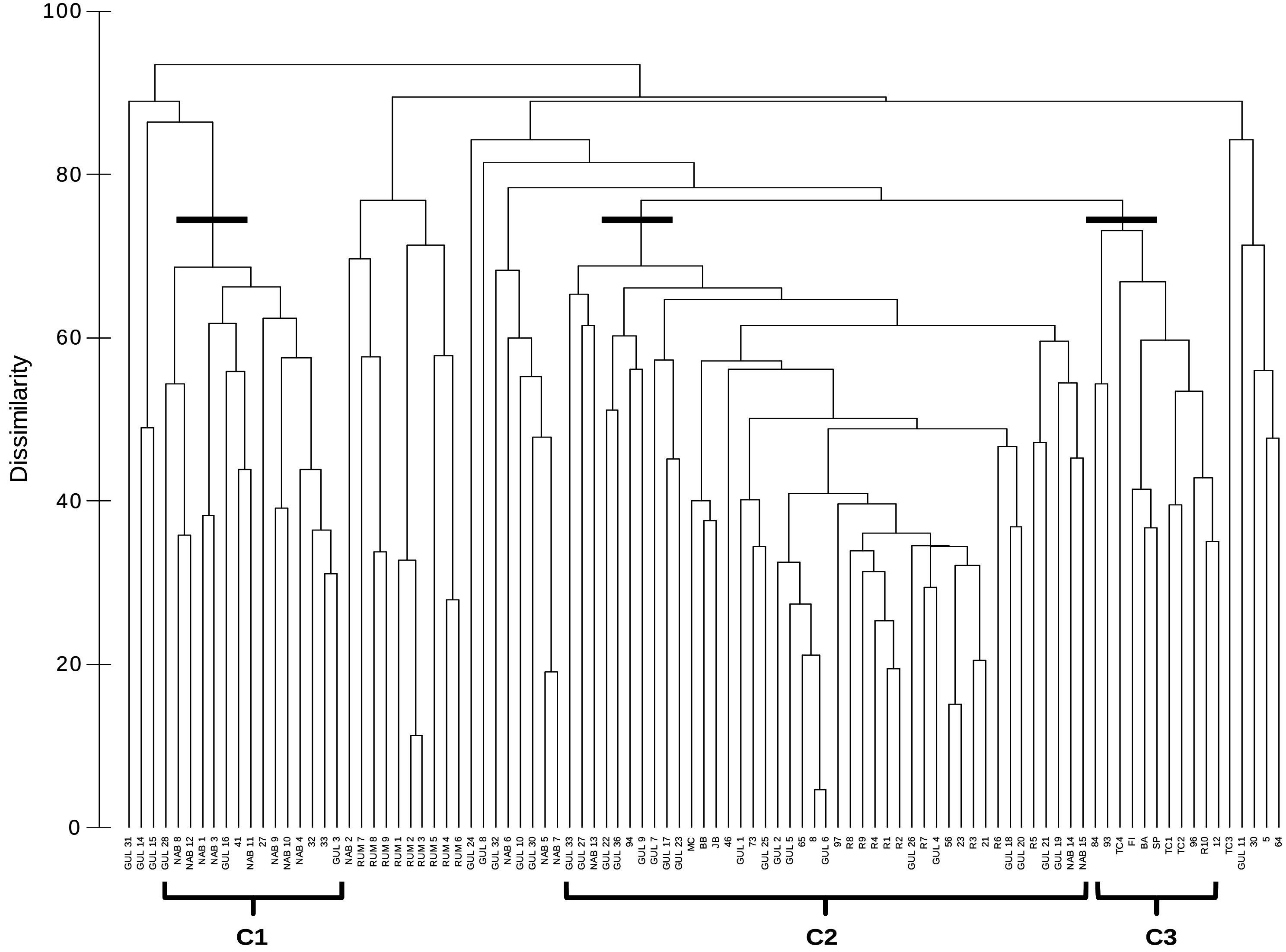


Figure 1 Cluster analysis (group average linkage) of trees and shrubs data for Alligator Rivers Region (ARR) vegetation analogue sites. (Vegetation data log transformed density/hectare units.) *Key to site codes*: Site label suffix (C1-C3) = original classification class for the site (Humphrey et al 2006, 2007; see Table 1). Numbered-only sites = EWLS Georgetown analogue area; 'R' sites = lowland Koolpinyah sites around Ranger; hill sites,'TC' (Tin Camp Creek) and 'F1' (Fisher) = schist hills; JB, BB, BA, SP = sandstone hills; MC = quartzite hill; 'GUL' sites = Gulungul catchment; 'NAB' sites = Nabarlek; RUM = Ranger minesite revegetation trials ('Heritage' area)

Charles Darwin University seed biology research

S Bellairs[1], M McDowell[1], C Humphrey, M Daws[2] & P Christophersen[3]

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 mine site. 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), Greening Australia and Top End Seeds (TES).

ERA are required to sustainably establish a range of local native species on rehabilitation areas at the Ranger mine site. 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 their nursery operations.

The aim of the project has been to investigate seed collection, 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.

---

[1] School of Environmental & Life Sciences, Charles Darwin University, NT 0909.
[2] Earth Water Life Sciences, GPO Box 518, Darwin, NT 0801
[3] Kakadu Native Plant Suppliers, PO Box 319, Jabiru NT 0886

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 more difficult than originally anticipated. TES collection data indicate that seeds for some species has not been able to be collected in sufficient quantities to make up a seed lot for seven years.

Testing of seeds is being carried out by CDU research staff under standard conditions in laboratory incubators to control for any light, temperature and moisture variations to which seed responses are highly susceptible. In this manner, the most effective treatments may be accurately identified without having to use large quantities of seeds.

Testing is based on the International Seed Testing Association guidelines and methodologies, with modifications to enable fewer seeds to be utilised. The methodologies used by the Australian Millennium Seed Bank Projects and other published studies are both guiding treatment selection and being used so that results can be compared with findings elsewhere.

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.

In 2008–09 an undergraduate student funded by the CDU UTROP research experience scheme investigated dormancy of *Persoonia falcata*.

Progress to date

CDU propagation trials

The seed biology project commenced in July 2006. At a meeting held at Jabiru with ERA, ERISS, EWLS, KNPS and CDU project staff in that year, fifty priority species were chosen 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 when established on the rehabilitation areas. This list was reviewed in December 2008 and modified slightly in early 2009, omitting those species for which sufficient seeds were unlikely to be able to be collected, thus reducing the priority list to 37 species. A major review and re-prioritisation were also carried out after the July 2009 project meeting (see section ‘Re-prioritisation of species for propagation’ below).

Seed lots of 26 of the original 50 priority species and an additional three species from the revised (early 2009) priority list have been received by CDU. Project progress on those species is summarised in Table 1. KNPS supplied five seed lots in August 2006, two in October 2006, four in January 2007, fourteen in July 2007, five in August 2009 and one in September 2009, including some additional seed lots of previously-supplied species. Thirty-two seed lots have been supplied by TES or Greening Australia NT between November 2006 and September 2008 and an additional six seed lots have been collected by CDU. As well as the priority species, some testing has occurred for 21 other species (Table 2). 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. In some cases, other species that also occur on the

Ranger mine lease have been tested for student projects or to provide a more detailed assessment of germination and dormancy trends.

The second annual report on the viability, germination and dormancy present in thirty species was provided to project sponsors in October 2008 (Bellairs & McDowell 2008). Additional species to those included in the 2007 report were: *Cymbopogon* sp, *Ectrosia* sp, *Eragrostis* sp, *Eriachne glauca*, *Eriachne schultziana*, *Eriachne triseta*, *Eulalia* sp, *Fimbristylis* sp, *Gomphrena canescens*, *Haemodorum coccineum*, *Livistona inermis*, *Schizachyrium fragile* and *Spermacoce stenophylla* (species identification to be confirmed). Test procedures have been developed for these species and initial viability, germination and dormancy testing has been conducted.

In addition, the germination and viability of seeds lots to be used in the trial landform plot were tested. Seventeen species had germination tests carried out and fifteen species had viability tests carried out. EWLS have developed an Access Database to store the seed biology data.

Table 1 Summary of progress investigating seed biology of the priority species

Species	# Lots received	Weight / Number	Viability	Imbibition	Germination	
Alloteropsis semialata	2	C	C	C,N	C	1
Aristida inaequiglumis	2	C	C,N	C,N	C	2
Brachychiton diversifolius	1	C	C	C	C	3
Brachychiton megaphyllus	1	C	C	C	C	4
Buchanania obovata	2	C	C	I,C	C	5
Chrysopogon fallax	2	C	C	C,N	C	6
Clerodendrum floribundum	1	C	C	N	P	7
Denhamia obscura	3	C	C	C,C,N	C,C,P	8
Eragrostis sp TBI	1	C	C	C	C	9
Eriachne obtusa	2	C	C	C,N	C	10
Eriachne schultziana	1	C	C	C	C	11
Eriachne triseta	1	C	C	N	C	12
Gomphrena spp TBI	3	C	C	N	C,C,P	13
Haemodorum coccineum	2	C	C	I,N	C,P	14
Heteropogon triticeus	1	C	N	N	C	15
Livistona humilis	2	C	C	C	P	16
Livistona inermis	2	C	C	C	P	17
Owenia vernicosa	5	C,C,C,P,P	I,C,C,N,N	I,N,N,N,N	I,N,N,N,N	18
Persoonia falcata	2	C	C	C	C	19
Petalostigma pubescens	1	C	C	N	P	20
Petalostigma quadriloculare	1	C	P	N	P	21
Schizachyrium fragile	1	C	C	C	C	22
Setaria apiculata	1	C	C	C	C	23
Setaria sp TBI	1	C	C	N	C	24
Spermacoce sp1 TBI	2	C	C	N	C,P	25
Spermacoce sp2 TBI	1	C	C	N	P	26
Terminalia carpentariae	3	C	C	C,I,N	C,C,P	27
Terminalia ferdinandiana	4	C	C,C,N,N	C,N,C,N	C,P,P,N	28
Terminalia pterocarya	1	C	C	N	N	29
Verticordia cunninghamii	1	C	C	C	C	30

C Completed; P In progress; N Not started, I insufficient seeds.

Table 2 Investigations of species that are related to the priority species or other species that occur on the Ranger area, including student projects

Species	# Lots received	Weight / Number	Viability	Imbibition	Germination
Chrysopogon latifolius	1	C	P	N	C
Crotalaria novae-hollandiae	1	C	C	N	P
Cymbopogon bombycinus	1	C	C	N	C
Cymbopogon sp TBI	1	C	C	C	C
Dichanthium sericeum	1	C	C	N	C
Ectrosia leporina	1	C	C	N	C
Ectrosia sp TBI	1	C	C	N	C
Eriachne burkittii	1	C	C	C	C
Eriachne glauca	1	C	C	C	C
Eulalia aurea	1	C	N	N	C
Eulalia sp TBI	1	C	C	C	C
Ficus benjamina	1	I	C	C	C
Ficus virens	1	I	C	C	C
Fimbristylis sp TBI	1	C	C	N	C
Heteropogon contortus	1	C	N	N	C
Hibiscus sabdariffa	1	C	C	N	P
Sarga intrans	1	C	P	N	C
Sarga plumosum	1	C	P	N	C
Tephrosia rosea	1	C	C	C	P
Themeda triandra	1	C	P	N	C
Triodia bitextura	1	C	C	N	C

C Completed; P In progress; N Not started, I insufficient seeds, TBI – identification to be confirmed.

The CDU researchers were successful in gaining CDU research training program funding and this was used to investigate the fruit-imposed dormancy of *Persoonia falcata*. The type of fruit treatment required to allow the seeds to germinate has been identified. Extracting the seed and applying gibberellic acid resulted in 60% germination, which was equivalent to the proportion of viable seeds. A second seed lot is being investigated to confirm this result prior to publication.

In 2009, summary reports were provided to sponsors and to KNPS. Meetings were also held in December 2008 and July 2009 to present data and discuss progress. Two other meetings with KNPS, EWLS and CDU were held in Jabiru to maximise information and technology transfer. The research work will continue to test new species and new seed lots of existing species. Considerable literature has been obtained on the taxa (although often on southern species of the genera in the priority list).

Re-prioritisation of species for propagation

At the July 2009 project meeting, a more objective prioritisation process was recommended for seed biology research, 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 quantitative abundance information for plants arising from the combined EWLS and ***eriss*** vegetation analogue sites (Humphrey et al 2007), adopting the following methodology:

Separately for the tree (>2 m height) and shrub (<2 m height) species, the combined ***eriss*** and EWLS analogue data used in Humphrey et al (2007) were used to determine the average density of each species (stems ha^{-1}). These stem density values were then ranked and plotted as a frequency distribution. For each species, it was assessed whether: (i) tubestock has been used for revegetation at Ranger and Jabiluka; (ii) whether seeds have been successfully germinated by KNPS or CDU; and (iii) whether seeds are available from KNPS. (In Figures 1 & 2, the three categories are coded 'Used in reveg?', 'Germination?' and 'Seeds available?' respectively.) Different symbol codings were assigned in the ranking graphs, depending upon whether the criterion was met, where a knowledge gap was identified or where the criterion was not met. Note that the 'Germination' or 'Seeds available' designation has been applied wherever this has 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 these species. Nevertheless, research needs for these species have a lower priority than species for which no knowledge is available.

For herb and grass species, the analogue data of Brennan (2005) relevant to sites adjacent to Ranger were used. Species were ranked according to plant biomass (g dry mass m^{-2}), and plotted similarly as a frequency distribution. For each species, it was assessed 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. (In Figure 3, the three categories are coded 'Used in reveg?', 'Germination?' and 'Seeds available?' respectively.) As for the trees and shrubs, different codings were assigned in the ranking graphs, depending upon whether the criterion was met, where a knowledge gap was identified or where the criterion was not met.

Ranked species abundances for all three categories (trees, shrubs and ground cover), together with the three categories of collective knowledge and knowledge gaps for each species, are provided in Figures 1–3 respectively. For each life-form category, the greatest priority for plant biology research would be expected to be placed on the abundant species for which there are significant knowledge gaps and/or concerns that seed stock is not readily available locally. The plots will be revised as additional information is made available but the most obvious observation at this stage is the dearth of seed biology information available for shrubs and ground cover species (Figures 2 and 3, respectively) compared to the amount of information available for trees (Figure 3). Thus a key recommendation from this review is that the focus for future seed biology research be shifted to shrubs and groundcover species. However and as noted above, we do not imply that the data in this report indicate that seed biology issues have necessarily been overcome for the other species. 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.

Where poor availability of seed stock is identified in the plots, other commercial sources of seed from outside of the ARR might be sought for research (propagation) purposes (eg www.topendseeds.com.au/price-list.php) and indeed, for rehabilitation generally noting that provenance issues will need to be resolved for these species.

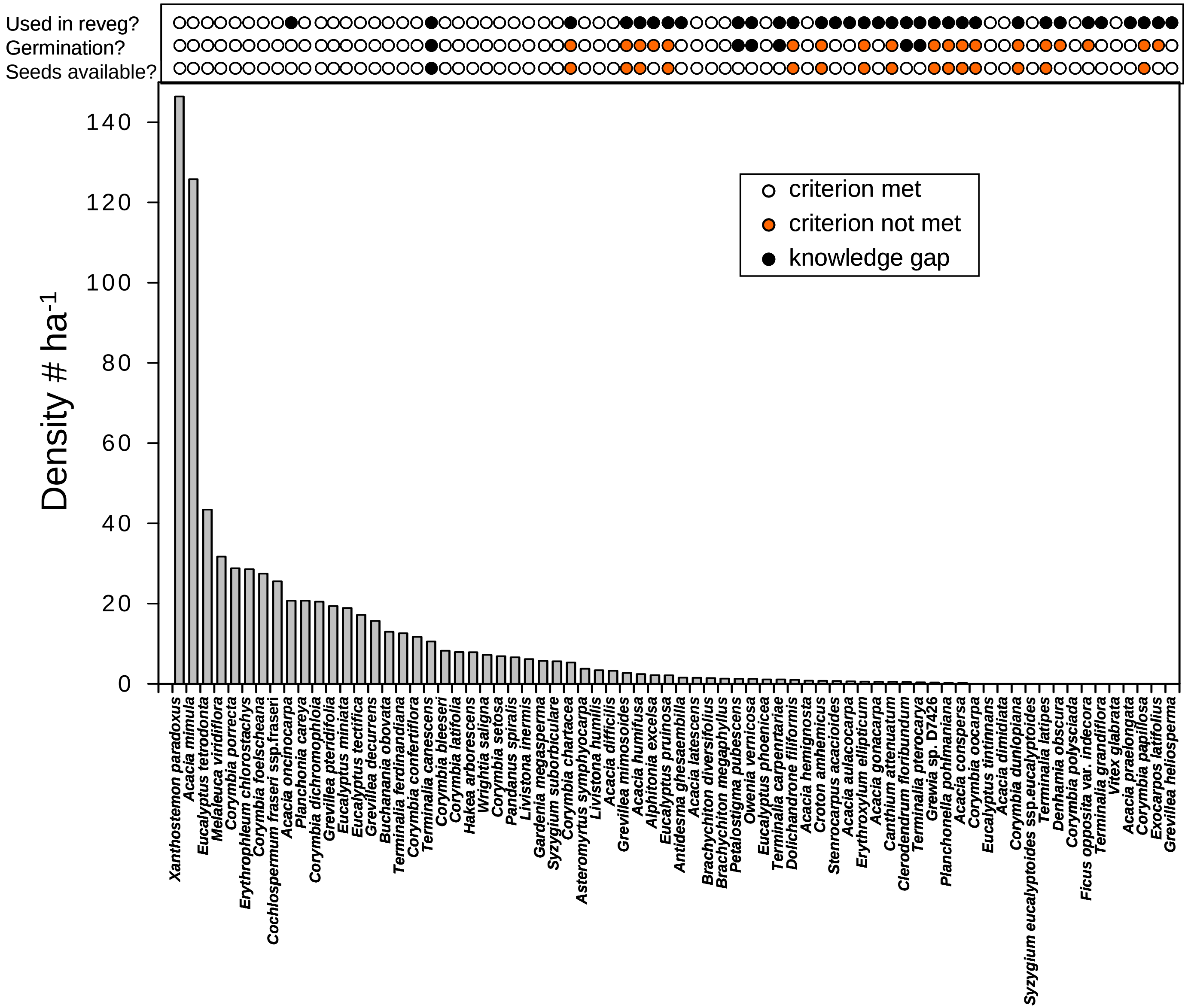


Figure 1 Relative density of trees present in ARR vegetation analogue sites (Humphrey et al 2007) and knowledge available for seed propagation and revegetation

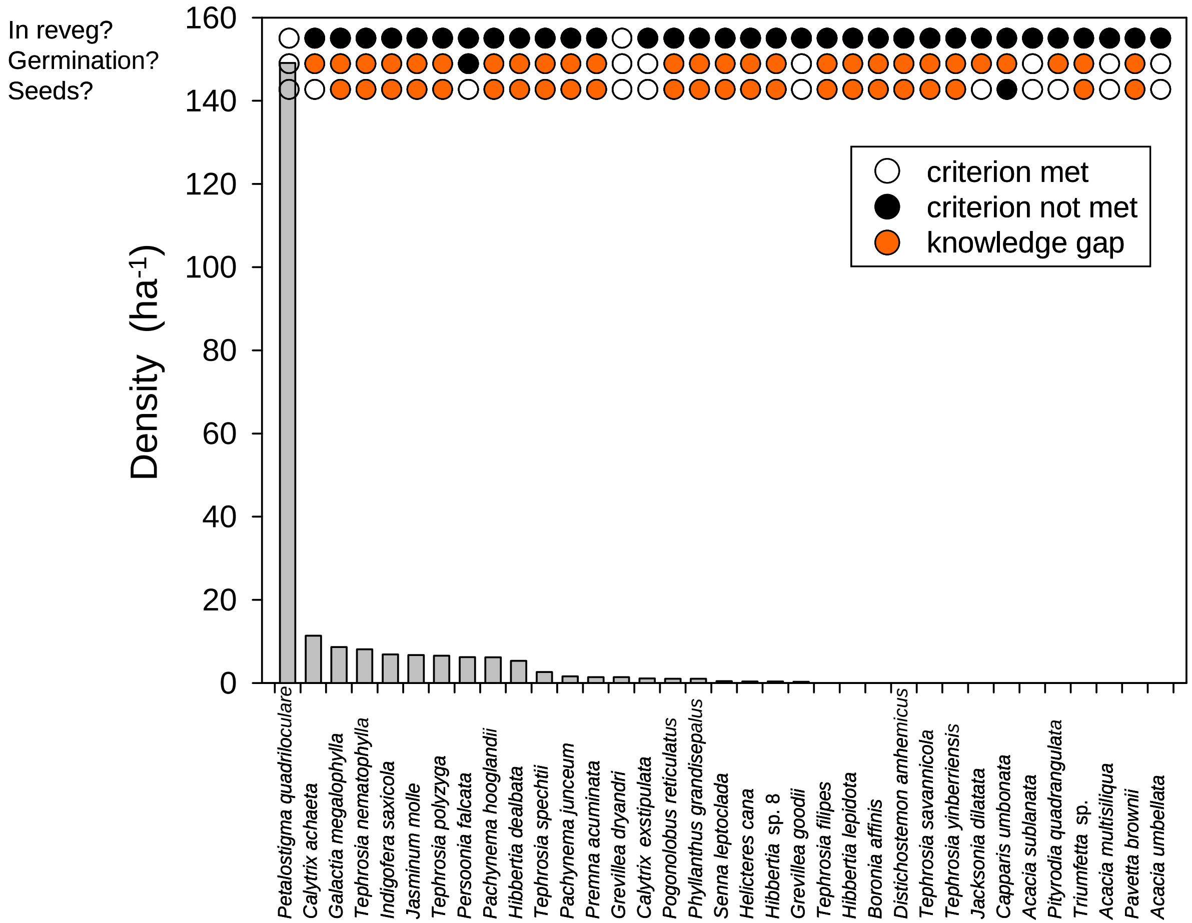


Figure 2 Relative density of shrubs present in ARR vegetation analogue sites (Humphrey et al 2007) and knowledge available for seed propagation and revegetation

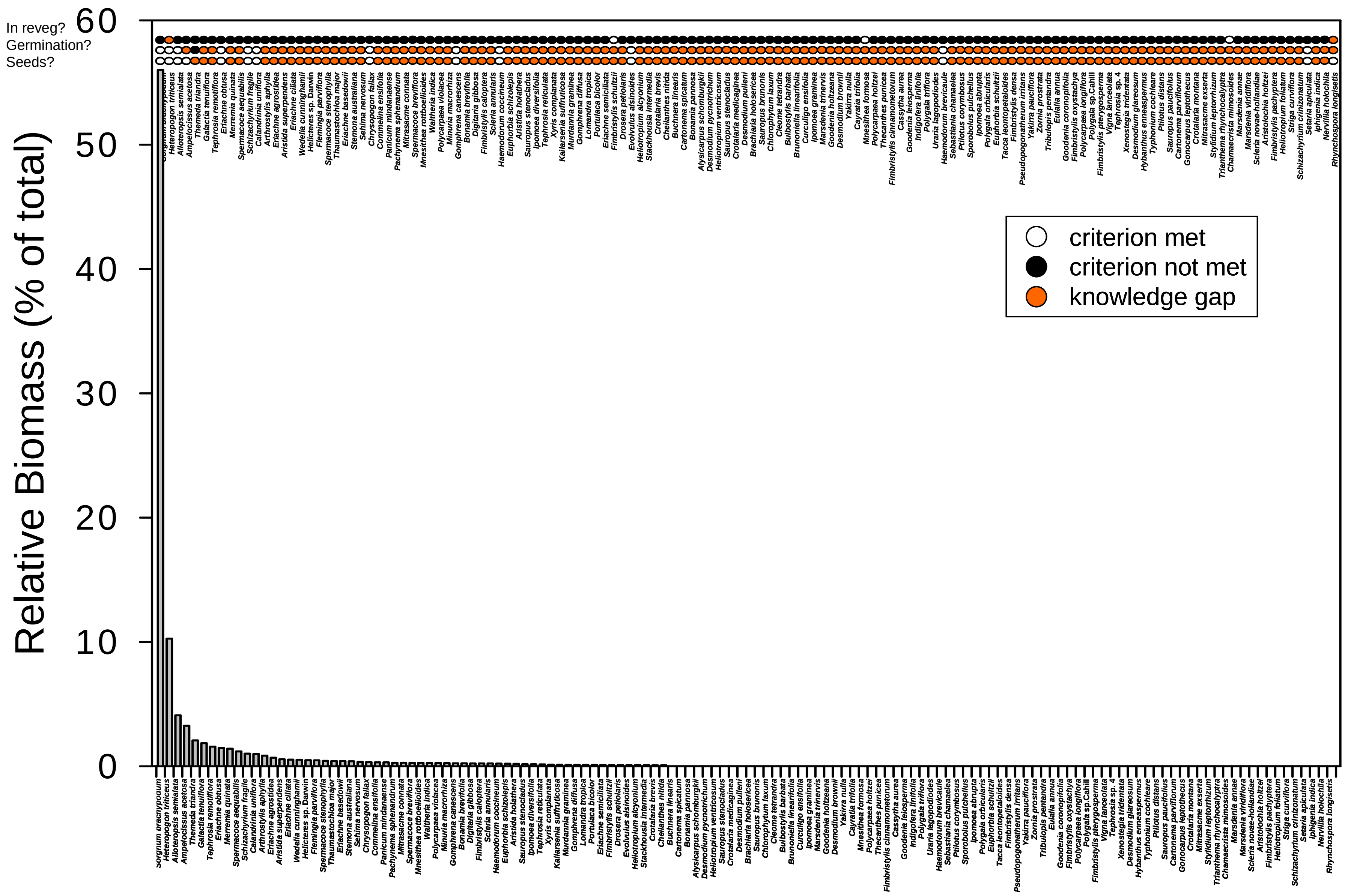


Figure 3 Relative density of herbs and grasses present at Ranger vegetation analogue sites (Brennan 2005) and knowledge available for seed propagation and revegetation

References

Bellairs SM & Ashwath N 2007. Seed biology of tropical Australian plants. In *Seeds: biology, development and ecology*. eds SW Adkins, S Ashmore & SC Navie, CABI Publishing, Oxford UK, 416–427.

Bellairs SM 2007. Seed biology of plants of the Australian wet/dry tropics and implications for Ranger mine site rehabilitation. Internal Report 523, March, Supervising Scientist, Darwin. Unpublished paper.

Bellairs SM & McDowell M 2008. ERA Ranger Uranium Mine Seed Biology Project Annual Report 2007–2008. School of Environmental and Life Sciences, Charles Darwin University, Darwin NT.

Brennan K 2005. Quantitative descriptions of native plant communities with potential for use in revegetation at Ranger uranium mine. Internal Report 502, August, Supervising Scientist, Darwin. Unpublished paper.

Humphrey C, Hollingsworth I, Gardener M & Fox G 2007. Use of analogue plant communities as a guide to revegetation and associated monitoring of the post-mine landform at Ranger. In ***eriss** research summary 2005–2006*. Jones DR, Evans KG & Webb A (eds), Supervising Scientist Report 193, Supervising Scientist, Darwin NT, 84–86.

Investigating radium uptake in *Passiflora foetida* (bush passionfruit)

P Medley, A Bollhöfer & D Parry[1]

Introduction

Our current ability to predict radiological dose received via the ingestion of radionuclides in terrestrial plants growing on a rehabilitated minesite is limited because the uptake mechanisms of radionuclides in plants are not well understood. The most common approach to determine doses is to use concentration factors for each food item and radionuclide to assess radionuclide uptake, and a model diet to estimate the quantity of each radionuclide that is ingested. Reported concentration factors, expressed as the ratio of the radionuclide activity concentration in the food item to the activity concentration measured in the soil in the plant root zone can, however, vary by up to three orders of magnitude, even for individual soil-crop combinations (Simon & Ibrahim 1990, Ryan et al 2005, IAEA 1994; Vandenhove et al 2009). Radium uptake in particular has been reported as being highly variable in tropical environments (Velasco et al 2009). In addition, although the International Atomic Energy Agency (IAEA) provides default concentration factor values for some food items (which are currently under review), the analogues available for local foods in the Alligator Rivers Region (eg. potato as an analogue for yam) have been shown to be inaccurate. In some cases, the concentration factors are different by several orders of magnitude (Ryan et al 2005).

The need exists to develop concentration factors specific for a region or even for individual sites to enable a more reliable prediction to be made for radionuclide activity concentrations in plants growing on a rehabilitated minesite. Site-specific studies for the purpose of determining concentration factors have also more recently been suggested by Vandenhove et al (2009) and Velasco et al (2009). Of particular importance for mine site rehabilitation are ^{226}Ra, ^{210}Po and ^{210}Pb. These radionuclides, which have comparatively high ingestion dose coefficients (ICRP 1996), have previously been identified as the main contributors to radiological dose via the ingestion pathway from eating traditional terrestrial bush food items such as fruits and yams in the Alligator Rivers Region (Martin & Ryan 2004).

Various approaches have been suggested to operationally define the transfer mechanisms of radionuclides from soil to plant, in an attempt to lower variability in concentration factors and to account for differences in bioavailability of metals between different soil types and/or sites. Many of these approaches have attempted to define the 'bioavailable' fraction within the soil using chemical techniques such as sequential extraction (eg Tessier et al 1979). In our study, a soil sequential extraction procedure was developed to assess partitioning of radium between soils and plants in the Alligator Rivers Region. Correlation of ^{226}Ra/^{228}Ra activity ratios in the different extractable fractions from soil in the plant root zone with those in the edible fruit has been used to infer the specific pools of radium in soil that are available for uptake into the fruit.

[1] Australian Institute of Marine Science, Arafura Timor Research Facility (ATRF), Darwin NT

Methods

An introduced passionfruit species, *Passiflora foetida* (Figure 1), was selected for study since it is eaten by indigenous groups in the Alligator Rivers Region and is also commonly eaten by children. This is important because ingestion dose coefficients are higher for children than for adults. *Passiflora foetida* is a shallow rooted, fast growing weed that may flower and fruit at any time of the year. This makes it ideal for short-term studies, and more likely to have a higher uptake of radium when growing in areas where contaminated material is retained in the surface layers.

Figure 1 Passiflora foetida

Samples of fruit and associated soils were collected from a wide range of sites including: the rehabilitated Nabarlek uranium mine in western Arnhem Land; a historic mine area impacted by uranium mine tailings ('Rockhole residues') in the South Alligator River valley; a land application area at Ranger mine impacted by mine waters; the black plain soils to the south-west of the Ranger tailings dam; and background sites not impacted by mining. The soil was then subjected to a sequential extraction procedure and radium isotopes (^{226}Ra and ^{228}Ra) measured in the soil extracts and fruit to determine concentration factors for the various fractions. The fractions (and extraction techniques) defined by this study were: the bioavailable fraction (water followed by a 1 M $MgCl_2$ leach, with the Ra extracted by the two steps being combined); the fraction bound to iron and manganese oxides (1 M HCl leach); the fraction that could be bound by organic complexes, particularly Ra-sulfate (Willett & Bond 1995) due to the very high levels of sulfate in the irrigation waters and extremely low solubility of radium in water and acid solutions (Kirby 1964) (0.2 M EDTA in 1.7 M NH_4OH); and the residual fraction (measured via gamma spectrometry) (Medley 2007).

Results

Final results of the ^{226}Ra and ^{228}Ra activity concentration measurements are now available to identify which extract fraction may be most suitable to use to determine concentration factors for the uptake of radium in *Passiflora*.

Total ^{226}Ra activity concentrations in the soils associated with the collected samples of *Passiflora* range over three orders of magnitude (35–11700 Bq/kg dry weight). The highest values measured are from the Rockhole residues site in the South Alligator River valley where passionfruit was growing in soil that contained tailings rich in ^{226}Ra. Activity concentrations are also elevated in the soils of the Magela land application area (MLAA) at Ranger mine due to the application of pond waters containing elevated ^{226}Ra, and in soils from the Gulungul catchment. The lowest soil activity concentrations were measured at a site close to Magela Creek, downstream of but un-impacted by the minesite. It is noted that the sites sampled cover a range of different origins for the ^{226}Ra, from being applied in solution to the top of the soil profile in the MLAA to residual ^{226}Ra in mine tailings at the Rockhole site. Hence it could be expected that the Ra might be bound in different ways in each of these soil types. The ^{226}Ra activity concentrations [Bq·kg^{-1} dry weight] measured in the samples of *Passiflora* range from about 3 Bq·kg^{-1} at the Magela Creek downstream site, to 520 Bq·kg^{-1} at the Rockhole residues site.

In contrast to ^{226}Ra, the ^{228}Ra activity concentrations in the soils were similar at all sites and reflect typical values seen throughout the region, ranging from 12–84 Bq/kg. The variability of ^{228}Ra activity concentrations in the fruits is very small and ranges from 1.3–5.1 Bq/kg.

Table 1 shows the calculated concentration factors derived from the ^{226}Ra activity present in the total soils and in the different selective extraction treatments for the various sites.

Table 1 ^{226}Ra Concentration Factors for *passiflora foetid*a measured relative to the various leach fractions

Sampling site	Concentration factors relative to extraction method			
	Total soil	**Bioavailable**	**1M HCl**	**EDTA + NH_4OH**
Rockhole residues	*0.030±0.001*	*0.27±0.012*	*0.201±0.006*	*0.70±0.02*
Nabarlek	*0.086±0.005*	*1.39±0.11*	*0.316±0.017*	*1.59±0.09*
Magela LAA	*0.018±0.001*	*0.45±0.03*	*0.079±0.005*	*0.34±0.02*
Gulungul 1	*0.0050±0.0002*	*0.35±0.02*	*0.010±0.001*	*0.086±0.004*
Gulungul 2	*0.0037±0.0002*	*0.32±0.02*	*0.0068±0.0003*	*0.066±0.003*
Magela DS	*0.238±0.021*	*1.87±0.11*	*1.131±0.074*	*2.86±0.28*

The concentration factor is defined as the activity concentration in the dry fruit divided by the activity concentration in the dry soil or in the leach fraction. Activity concentration in the leach fraction is expressed as the total activity leached divided by the dry soil weight. Uncertainties are one standard deviation based on counting statistics only.

Using total soil, and the HCl and EDTA extraction steps, as the basis for the calculation there is an up to two order of magnitude range of values for the concentration factors spanned by the data shown in Table 1. However, when the bioavailable fraction only is considered, a much smaller variation is observed with values ranging from 0.27–1.87. Critically, the use of the ^{226}Ra/^{228}Ra activity ratio as a tracer of the origin of the radium in passionfruit confirmed that the bioavailable fraction is the most likely source of the radium taken up by the plant. That is, the ratio of ^{226}Ra/^{228}Ra in the plant is most similar to the ratio that is found in the bioavailable extraction. The calculated concentration factor based on the bioavailable fraction is plotted against total soil activity concentration in Figure 2. The ^{226}Ra concentration factors are highest at the lowest soil activity concentrations and approach a saturation value of approximately 0.3 at high soil ^{226}Ra activity concentrations. This suggests that Ra uptake is non-linear – this has previously been suggested for plants from more temperate regions (Martinez-Aguirre et al 1997, Madruga et al 2001).

The non-linearity of concentration factors has important implications for, and further complicates, dose models that use concentration factors to derive ingestion doses, as ^{226}Ra activity concentrations in plants may be over- or underestimated, depending on the degree of contamination of the soil. Site-specific concentration factors should thus be determined relative to the bioavailable fraction of ^{226}Ra in the substrate to increase the confidence in the modelled ingestion doses.

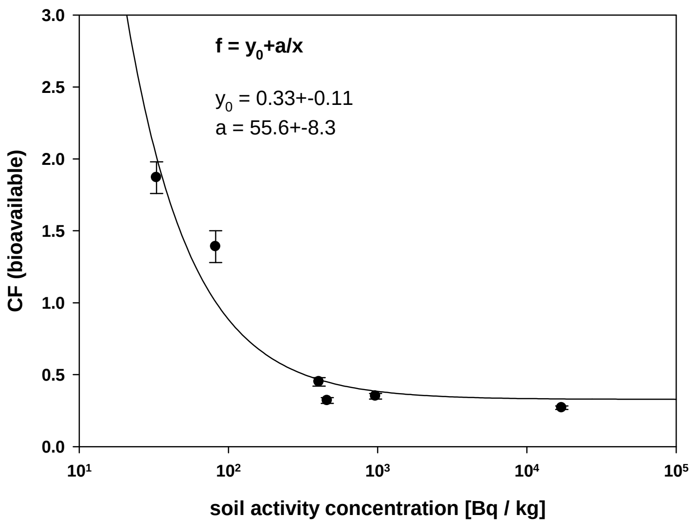


Figure 2 Concentration factors based on the bioavailable fraction plotted versus soil ^{226}Ra activity concentration (log scale)

Steps for completion

It is planned, using the selective extraction and radium isotope ratio approach described above, to extend the *Passiflora* work to other plant species commonly eaten by indigenous people in the Alligator Rivers Region.

Acknowledgments

We would like to acknowledge Anthony Sullivan and Sally Atkins from SSD for providing advice and for their help with sample collection.

References

IAEA 1994. *Handbook of parameter values for the prediction of radionuclide transfer in temperate environments*. IAEA Technical Report Series No 364, Vienna.

ICRP 1996. *Age-dependent doses to members of the public from the intake of radionuclides: part 5. Compilation of ingestion and inhalation dose coefficients*. International Commission on Radiation Protection Publication 72.

Kirby HW & Salutsky ML 1964. *The radiochemistry of radium*. National Academy of Sciences, Nuclear Science Series NAS-NS 3057.

Madruga MJ, Brogueira AG & Cardoso F 2001. ^{226}Ra bioavailability to plants at the Urgeirica uranium mill tailings site. *Journal of Environmental Radioactivity* 54, 175–188.

Martin P, Hancock GJ, Johnston A & Murray AS, 1998. Natural-series radionuclides in traditional north Australian Aboriginal foods. *Journal of Environmental Radioactivity* 40, 37–58.

Martin P & Ryan B 2004. Natural-series radionuclides in traditional Aboriginal foods in tropical northern Australia: a review. *TheScientificWorldJOURNAL* 4, 77–95.

Martínez-Aguirre A, García-Orellana I & García-León M 1997. Transfer of natural radionuclides from soils to plants in a marsh enhanced by the operation of non-nuclear industries. *Journal of Environmental Radioactivity* 35(2), 149–171.

Medley P 2007. Validation and refinement of a method for determination of ^{228}Ra, via gamma spectrometry using the ^{228}Ac daughter, or alpha spectrometry using the ^{228}Th daughter. Honours Thesis, Charles Darwin University, Darwin, Australia.

Ryan B, Martin P & Iles M 2005. Uranium-series radionuclides in native fruits and vegetables of northern Australia. *Journal of Radioanalytical and Nuclear Chemistry* 264(2), 407–412.

Simon SL & Ibrahim SA 1990. Biological uptake of radium by terrestrial plants. In *Environmental behaviour of radium*. IAEA Technical Reports Series 310, 545–599.

Tessier A, Campbell PGC & Bisson M 1979. Sequential extraction procedure for the speciation of particulate metals. *Analytical Chemistry* 51, 844–851.

Vandenhove H, Olyslaegers G, Sanzharova N, Shubina O, Reed E, Shang Z & Velasco H 2009. Proposal for new best estimates of the soil-to-plant transfer factor of U, Th, Ra, Pb and Po. *Journal of Environmental Radioactivity* 100 (9), 721–732.

Velasco H, Juri Ayub J & Snason U 2009. Influence of crop types and soil properties on radionulcide soil-to-plant transfer factors in tropical and subtropical environments. *Journal of Environmental Radioactivity* 100 (9), 733–738.

Willett IR & Bond WJ 1995. Sorption of manganese, uranium and radium in highly weathered soils. *Journal of Environmental Quality* 24, 834–845.

Storing, accessing and communicating the bushtucker project information

D Walden, R Bartolo, B Ryan & A Bollhöfer

Background

The intake of radionuclides in traditional bush food items sourced from radiologically enhanced/contaminated areas has been identified as a significant potential contributor to the radiological dose to traditional people living the Alligator Rivers Region, in particular after minesite rehabilitation (Ryan et al 2005). There are a large number of dietary items, radionuclides and exposure pathways to be considered to assess potential ingestion doses. The ***eriss***/EnRad bushtucker database contains over 1500 records (individual samples) of a wide variety of plants and animals (and water and sediments) collected from over 100 sites in the Alligator Rivers Region and as far east as Maningrida in Arnhemland. The challenge is to develop a user friendly system to store and retrieve the data, and to present this information to the local people of the area, and a wider public audience, using media such as the internet and other graphical user interfaces. Employing an interactive spatial interface will greatly simplify locating sites and presenting data spread over such a broad geographic area.

Methods

The Department of the Environment, Water, Heritage and the Arts employs Google Maps for their Heritage website where Google Maps is embedded into the web pages (see Figure 1 www.environment.gov.au/heritage/index.html).

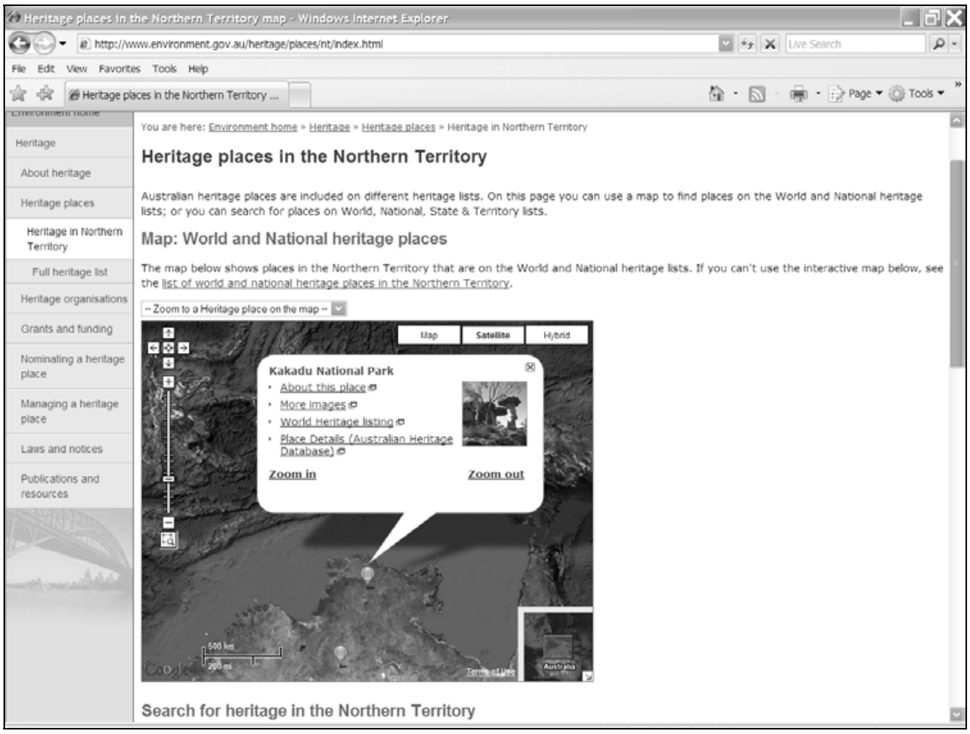


Figure 1 DEWHA website with Google Maps used to find Heritage sites in Australia

The user selects a state from a map of Australia, zooms into and clicks on any number of placemarks revealing a callout box with text, images and hyperlinks to a host of information relevant to that site.

A similar bushfoods website with these features is currently being developed whereby users can easily locate the sites sampled for bushtucker, what species were examined and other relevant information. The website will be formatted according to the Departmental template with appropriate headings, banners, logos menus and links as per other DEWHA web pages.

In addition, Keyhole Markup Language (KML) files containing site locations and associated data are being developed in parallel to the website information. The files are in a compressed format and can be made available to the public via download from the internet or on storage media such as CD/DVD. They can be used by a growing number of freeware 3D browsers such as Google Earth, ArcExplorer and Arc Globe, enabling users to zoom into sites using high resolution satellite imagery and manipulate the tilt, roll, angle, altitude and terrain of the landscape. In most cases the KML files can also be imported into web-based viewers such as Google Maps, Virtual Earth (Bing Maps), Yahoo Maps and Whereis.com. The advantage of the KML files in a full screen environment such as Google Earth, is the greater available space for images and information directly in the placemark callout box without having to hyperlink to other pages. The disadvantage is that the viewing program has to be installed manually and performance and refresh times can be limited by computer hardware.

At this stage of the project, information will be presented in general terms as outlined in Table 1. The actual results for radionuclides and other elements will not be presented for viewing or use by external stakeholders and the general public because interpretation of the radionuclide results is complex, and general users may find the numbers confusing and/or misleading.

Table 1 Information to be presented for external stakeholders and the general public

Information	Details
Differing names of species	• Common • Latin (scientific) • Bininj (local Aboriginal)
Why is the site important?	• Proximity to contamination source or control site • Used for food frequently • Special cultural significance
Why has the food been sampled/tested?	• Eaten frequently • Position on the food web and feeding habits – bioaccumulation issues • Proximity to a contamination source • Totem/cultural significance
How is the food sampled/obtained/prepared?	• Methods used by scientists – eg dredge • Traditional harvesting methods • How is the food prepared for eating? • How is the sample prepared for testing?
What parts of the animal/plant have been tested?	• Flesh • Various internal organs • Fruit, leaves, stems or roots
What has the food been tested for?	• Which radionuclides and/or heavy metals?

Progress/results

The database contents were converted to KML files early in 2008[1]. Many results are expressed both as radionuclide activity concentration and concentration factors, giving at least two separate radionuclide values for any one sample at any given site. Individual sites often have dozens of samples (Figure 2). Consequently, the default presentation method for multiple attributes for one location in Google Earth whereby the sample icons expand out from a representative site icon, while impressive, is simply overwhelming. In addition, the format in which the data translates from the GIS files is far from meaningful to the lay person. Names are truncated and the default callout box reflects the original database structure (Figure 2). Consequently, the database content that will be presented in the KML files has been 'streamlined' to suit the presentation method. A number of positional errors have also been corrected.

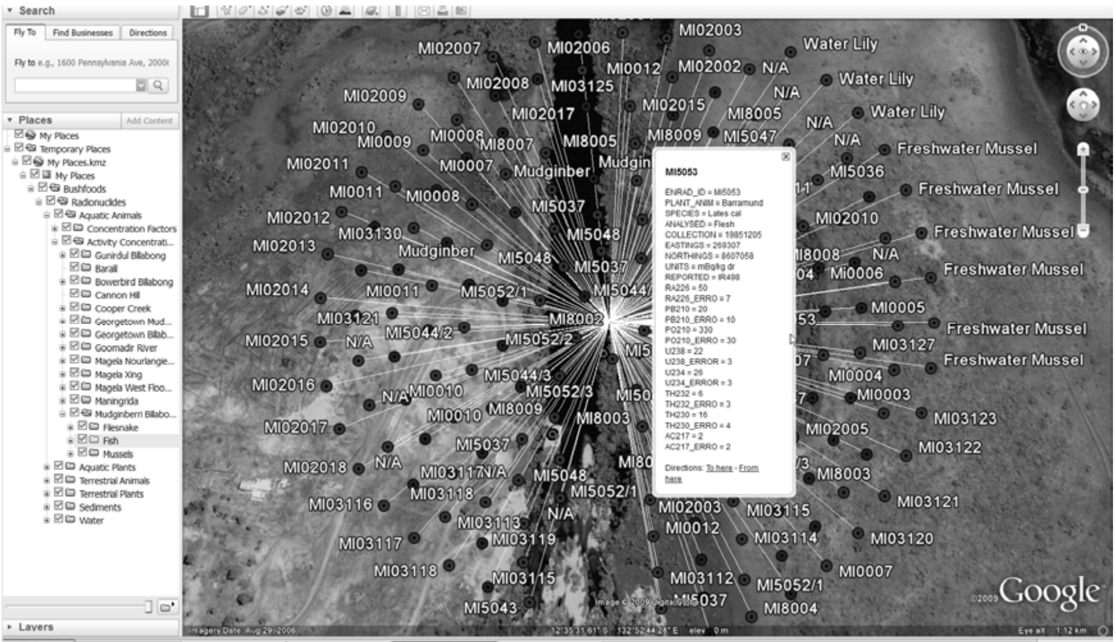


Figure 2 A single sampling site at Mudginberri Billabong selected in Google Earth showing all samples

Early in 2009, work recommenced on the KML files in the Google Earth environment. As the actual values are generally not being presented it is possible to present summarized information directly in the callout boxes as free text along with images and hyperlinks to either other KML files or web pages if necessary. A number of KML based templates have been designed to display information in a more user friendly and visually aesthetic way. The template content can be easily edited to describe different sites and samples. A collection of customized icons has been created representing most of the species sampled. These were created from photographs owned by SSD and some were provided by ***eriss*** staff. Examples of the templates and icons can be seen in Figures 3 and 4.

Liaison has commenced with the Web and Intranet Management Section (WIMS) of the Department of the Environment, Water, Heritage and the Arts to assess the feasibility of uploading all of the bushfoods information as outlined in this document, onto the Department's internet site.

---

1 This follows on from earlier work by Bartolo et al 2007, who created ArcGIS shapefiles from the original database to develop a Bushtucker Spatial Information System (SIS).

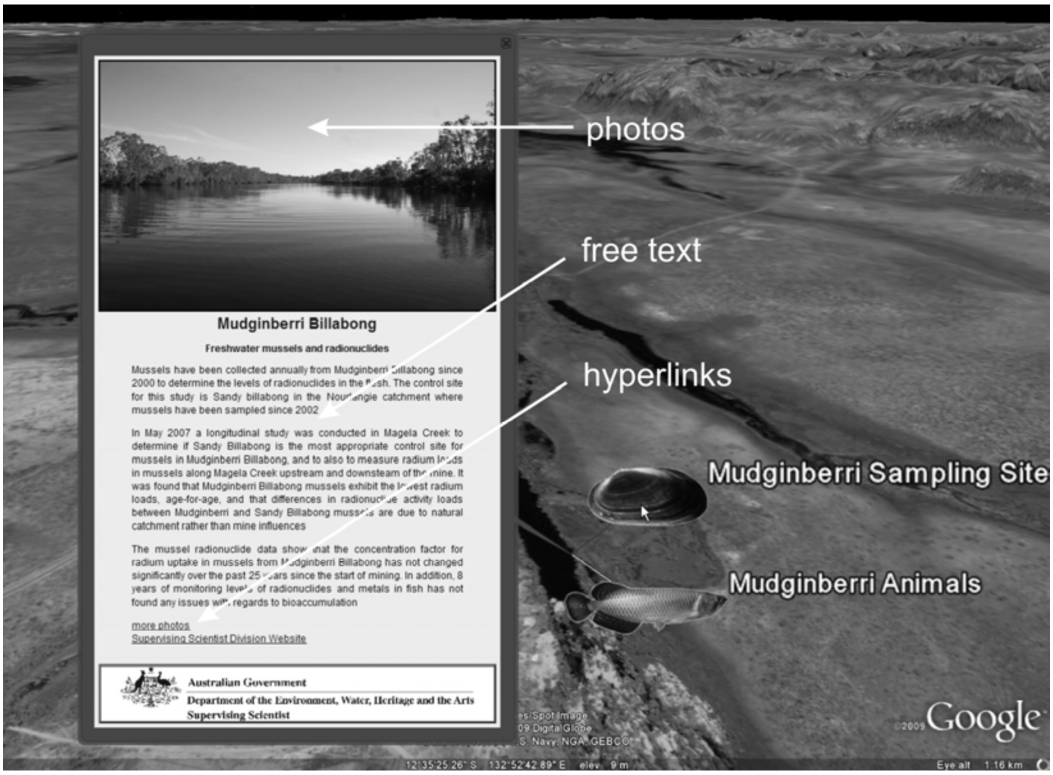


Figure 3 An example of a large callout box showing an image of the site, a description of the sampling and species and hyperlinks to more images and relevant websites for example

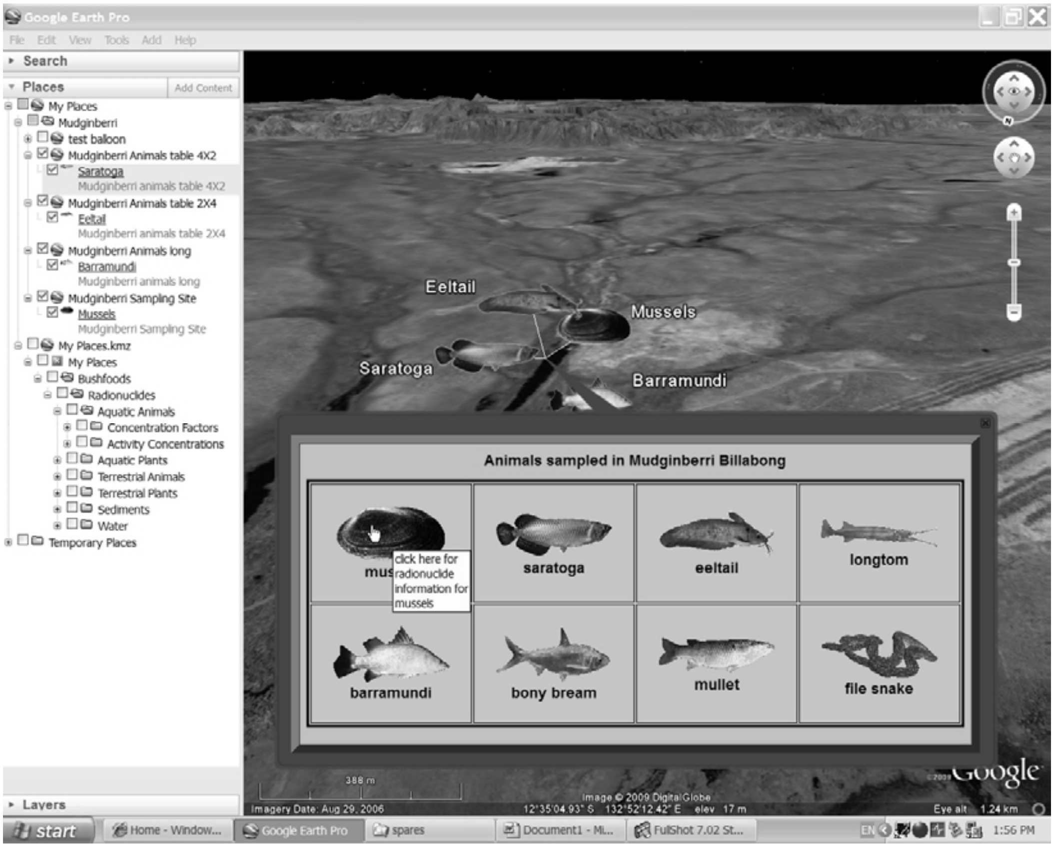


Figure 4 An example of a callout box representing species at a site. The icons can be links to further information about that species. Note the mouse pointer over the mussel icon. The view is looking south along the Magela Ck to Ranger in the background.

Conclusions

The use of this technology represents a new phase of communication of SSD research. Although there is abundant SSD information on the intranet and internet, to date there has been no real spatial presentation of data nor any spatial searching facilities. Internet accessibility will reach the widest audience searching for information on radionuclides and bushfoods. If each site is represented by a web page, there will be 100 or so pages to draft and it may become necessary to group sites and reduce detail to get the project on the web.

The KML products, although time consuming to produce initially, once completed and the content approved, can be distributed on CD or simply posted (and easily updated) on an existing SSD web page as a download for users with compatible browsers. These browsers are free, easy to install and use and are in general popular use.

Once the methods and procedures have been established for this project, it is anticipated that other SSD databases and indeed a host of general information will be presented using this technology.

Steps for completion

- Continue work on the summaries of site and species information.
- Complete the design of the species icons.
- Obtain photos of all sites.
- Populate the KML templates with the above information.
- Test the KML product prior to public access.
- Work with the WIMS to agree on how the internet product will be hosted. The task at hand is to draft some content, test its performance under a variety of conditions and seek approval to proceed under the Department guidelines.
- Once the above is completed, work can commence on drafting the web pages.

References

Bartolo R, Ryan B & Bollhöfer A 2007. Traditional diet in a modern world: Implementing a bushtucker SIS to communicate radiological issues. Paper presented at Spatial Sciences Institute Biennial International Conference 2007, Hobart Tasmania, 14–18 May.

Ryan B, Martin P & Iles M 2005. Uranium-series radionuclides in native fruits and vegetables of northern Australia. *Journal of Radioanalytical and Nuclear Chemistry* 264(2), 407–412.

Radio- and lead isotopes in sediments from the Nourlangie and Koongarra catchments (PhD project)

A Frostick, A Bollhöfer & D Parry[1]

Introduction

This project aims at developing an innovative and sensitive methodology to investigate and monitor sources, pathways and deposition of materials eroded from past, present and future uranium mining activities in the wet-dry tropics. Funded through the ARC Linkage Projects scheme, the project is a collaboration with researchers from Charles Darwin University. The objective of the project is to characterise sources and pathways of pollutants in catchments in the Alligator Rivers Region at the decommissioned and rehabilitated Nabarlek mine site (Frostick et al 2008), the operating Ranger mine, and at natural analogues, in order to develop a joint lead isotope/radionuclide approach for monitoring transport and dispersion of erosion products from a (rehabilitated) uranium mine site.

Due to the source-specific lead isotope signatures and the fact that no physical or chemical fractionation of lead isotopes occurs during transport and deposition, stable lead isotopes are ideally suited as a contaminant source tracer. Lead isotopic fingerprinting relies on the fact that three of the four stable lead isotopes, lead-204, lead-206, lead-207 and lead-208, are produced by the decay of uranium-238 ($^{238}U \rightarrow {}^{206}Pb$, $t_{1/2} = 4.5 \cdot 10^9$ yrs), uranium-235 ($^{235}U \rightarrow {}^{207}Pb$, $t_{1/2} = 0.7 \cdot 10^9$ yrs) and thorium-232 ($^{232}Th \rightarrow {}^{208}Pb$, $t_{1/2} = 14 \cdot 10^9$ yrs), respectively. ^{204}Pb is of primordial origin only.

In uranium and thorium rich minerals, radiogenic lead is continuously produced over time. For example, monazites present in many sands with high Th/U exhibit $^{208}Pb/^{207}Pb$ and $^{206}Pb/^{207}Pb$ ratios much higher than the present day average crustal (PDAC) lead (Bosch et al 2002) which has $^{206}Pb/^{207}Pb \approx 1.20$ and $^{208}Pb/^{207}Pb \approx 2.48$, respectively. On the other hand, uranium ore bodies show elevated $^{206}Pb/^{207}Pb$ ratios but are low in $^{208}Pb/^{207}Pb$, as ^{208}Pb is formed by the radioactive decay of thorium rather than uranium. Gulson et al (1992), for example, measured $^{206}Pb/^{207}Pb$ ratios in particulates from uranium tailings at Ranger as high as 9.69, whereas $^{208}Pb/^{207}Pb$ is as low as 0.0494, in agreement with results from a study of airborne dispersion of Ranger mine origin dust (Bollhöfer et al 2006). There have also been several studies utilising the lead isotope fingerprinting technique elsewhere in the Alligator Rivers Region including: the use for mineral exploration at the Jabiluka and Koongarra ore bodies (Gulson & Mizon 1980, Gulson 1986); lead isotopes in soils and plants from the Koongarra deposit for biogeochemical prospecting (Dean & Gulson 1987); an assessment of sediments from Ngarradj (Swift Creek), a catchment potentially influenced by the Jabiluka mineral lease (Bollhöfer & Martin 2003); and an investigation of sediments from Cooper Creek potentially influenced by the rehabilitated uranium mine at Nabarlek (Frostick et al 2008).

Coupled with the measurement of radionuclide activity and trace metal concentrations, the lead isotope fingerprinting technique enables identification and quantification of the

---

[1] Australian Institute of Marine Science, Arafura Timor Research Facility (ATRF), Darwin NT

depositional patterns of solids eroded due to past and present mining activities (Munksgaard et al 2003, Frostick et al 2008). This part of the PhD project aims at investigating natural, undeveloped uranium mineralised analogues in the Alligator Rivers Region. Anomaly 2 to the south of the Ranger lease and the undeveloped Koongarra uranium deposit in the Nourlangie Creek catchment were investigated. This present report focusses on the results from the Nourlangie Creek catchment.

Location and methods

The Koongarra uranium deposit was discovered in 1970 by Noranda Australia Ltd. The lease covers approximately 1197 hectares and is located next to the Arnhem Land escarpment, 30 km south of Ranger uranium mine and 3 km east of Nourlangie Rock in a valley bounded by the Mount Brockman outlier and the Arnhem Land plateau. The orebody lies entirely within the catchment of Koongarra Creek which enters Nourlangie Creek and into the South Alligator River. Anbangbang Billabong is located approximately 5 km downstream of Koongarra mineral deposit, whereas Sandy Billabong is located in the Nourlangie Creek system upstream of the confluence of Koongarra with Nourlangie Creek, and is thus unaffected by runoff from Koongarra.

Surface sediments were collected from the Koongarra Mineral Lease during the dry season in 2007. Approximately 2 cm of the top sediment (< 2 mm) was placed in acid-cleaned poly-ethylene bags before drying at 60°C then grinding in an automated agate mortar and pestle. A fraction of the sample was taken for heavy metal and lead isotope analysis via inductively coupled plasma mass spectrometry (ICPMS). Around 150 g of the ground sediment was pressed into a standard geometry for direct gamma spectroscopy.

Sediment cores were retrieved from Anbangbang and Sandy Billabongs in late 2007. Cores were taken either by boat or direct bank access using lengths of PVC pipe driven into the sediment. Cores were cut into 10 mm sections using an acid cleaned stainless steel knife. Each section was then centrally cored using a stainless steel ring cutter and the external sediment, which had been in contact with the PVC, was discarded. Core samples were then dried and ground as described above. A fraction of the sample was taken for heavy metal and lead isotope analysis via ICPMS. For gamma spectroscopy samples from progressive depths were combined until approximately 15 g total of material was obtained. This mass was then pressed into a standard geometry and counted for 1–2 days.

Samples were analysed for radionuclides at ***eriss*** using high resolution High Purity Germanium (HPGe) gamma detectors in 2008-09. Procedures for sample collection, preparation and measurements of radionuclide activities via gamma spectrometry at ***eriss*** are described in Murray et al (1987), Marten (1992) and Esparon and Pfitzner (2010).

Results

Radionuclide activity concentrations of scrape samples taken on the Koongarra lease and from the Nourlangie Creek bed are relatively low and comparable with concentrations at other creeks within the Alligator Rivers Region. ^{210}Pb activity is higher than ^{226}Ra and ^{238}U, and ^{228}Ra and ^{228}Th are in equilibrium in all surface samples. The ^{40}K activities follow similar trends to ^{228}Th. Activities of ^{226}Ra and ^{210}Pb are highest at the two downstream sites before Koongarra Creek exits the lease area.

The top of the two billabong cores exhibit higher concentrations of radionuclides and heavy metals, probably related to the higher content of clay and organics compared to the sandier nature of the sediment in deeper sections of the cores. This has also been observed in sediment cores taken along Cooper Creek in Western Arnhem Land (Frostick et al 2008). Both Anbangbang and Sandy Billabong cores exhibit higher ^{210}Pb activity concentrations in the top sections as compared to ^{226}Ra (Figure 1) but they are generally in equilibrium (to within 2σ) in the bottom sections of the cores. ^{210}Pb excess dating of the two sediment cores reveals a sedimentation rate at the Sandy Billabong site of approximately 0.15 ± 0.01 cm yr^{-1}, whereas the sedimentation rate for Anbangbang Billabong is higher at 0.34 ± 0.12 cm yr^{-1}. ^{137}Cs activity concentration profiles measured support these sedimentation rates (Figure 1).

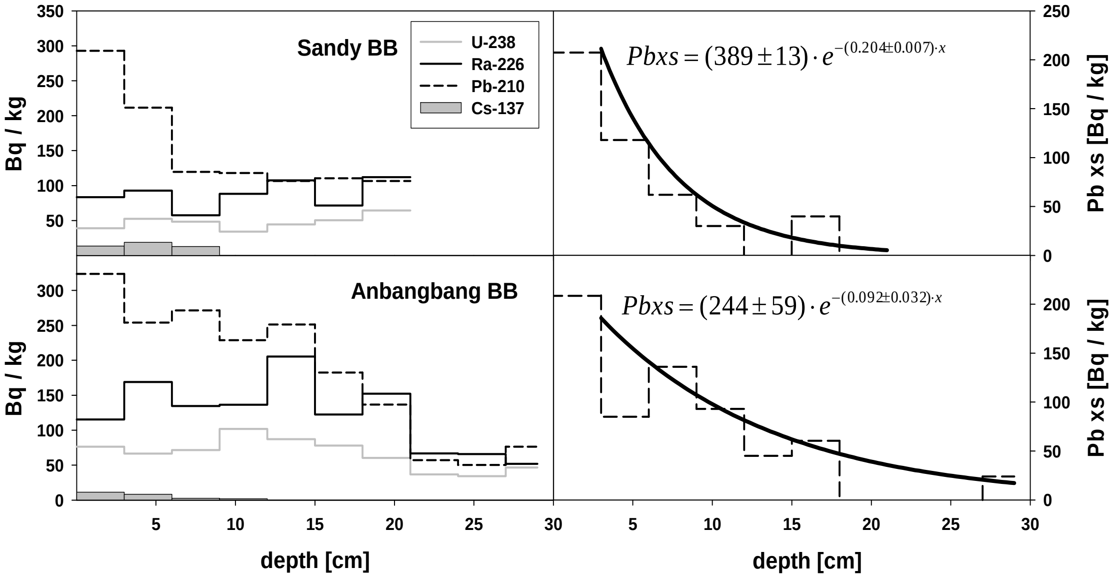


Figure 1 ^{238}U, ^{210}Pb, ^{226}Ra and ^{137}Cs activity concentrations, and ^{210}Pb excess versus depth in Sandy and Anbangbang Billabongs. Exponential fits to the ^{210}Pb excess (Pb_{xs} = ^{210}Pb – ^{226}Ra) activity concentration data are also shown (from Frostick et al 2010).

Variations in metal concentrations through the cores are due to changes in core lithology. The Sandy Billabong core showed a distinct layering of dark humic sections alternating with sandier layers at 10-14 cm (~65 yrs BP) and 19-21 cm (~125 yrs BP), respectively. Anbangbang Billabong contained only one major sand dominated layer from 22-29 cm (~65 yrs BP). The presence of sand layers is reflected in the lower trace metals concentrations and also radionuclide activities of the horizons. It appears that both cores recorded a sand layer deposited about 65 years before present.

The sediment samples from Sandy Billabong located at Nourlangie Creek, upstream of the confluence with Koongarra Creek, exhibit lead isotope ratios similar to the general background within the Alligator Rivers Region area, as reported in Frostick et al (2008). Trends in ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb are comparable to trends seen in Cooper Creek upstream of Nabarlek mine (Frostick et al 2008) and Ngarradj (Bollhöfer & Martin 2003) although the variations are much smaller (Figure 2). At Cooper Creek it was assumed that sands bearing heavy minerals with comparatively low Pb concentrations and high ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb ratios, are mixed with natural dust, clays and silts that contribute Pb with a significantly different lead isotopic composition, closer to present day average crustal (PDAC). The contribution of the heavy minerals to total Pb within the samples may be small but can be seen in the isotope ratios in the Sandy Billabong sediment samples, which have higher than PDAC ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb.

Anbangbang Billabong is located ~5 km downstream of the Koongarra mineral deposit and shows higher radionuclide activity concentrations in the sediment than Sandy Billabong. The samples also exhibit consistently more radiogenic $^{206}Pb/^{207}Pb$ but less radiogenic $^{208}Pb/^{207}Pb$ ratios than sediments from Sandy Billabong. Although higher ^{238}U activity concentrations can also be associated with an increase in the amount of silts and clays deposited in a sediment core, common silts and clays usually exhibit lead isotope ratios more similar to PDAC lead which would shift the ratio in the Anbangbang core to lower $^{208}Pb/^{207}Pb$ and lower $^{206}Pb/^{207}Pb$ isotope ratios. This is only the case in the top sediment sample. Throughout the remainder of the core $^{208}Pb/^{207}Pb$ is reasonably constant at about 2.57, whereas a rise in the $^{206}Pb/^{207}Pb$ ratio can be observed from about 80 - 60 years before present to an average of ~1.375 (Figure 2). The lower $^{208}Pb/^{207}Pb$ and higher $^{206}Pb/^{207}Pb$ compared with Sandy Billabong may be caused by uraniferous material originating from the Koongarra Mineral deposit.

Scrape samples collected for our study exhibit $^{206}Pb/^{207}Pb$ ratios up to ~ 1.80 and $^{208}Pb/^{207}Pb$ ratios of 2.53 (sample KML1). If we assume an end member ratio for uranium rich material from the Koongarra deposit of $^{206}Pb/^{207}Pb$ = 7.364 (data from Dickson et al 1985) and a background ratio of 1.328 (see Figure 2) this would indicate a relative contribution of about 1 percent of uraniferous sediment originating from the Koongarra mineral lease to the top sediments in Anbangbang Billabong.

The input of sediments with a higher $^{206}Pb/^{207}Pb$ but little different $^{208}Pb/^{207}Pb$ ratio in the upper section of the Anbangbang Billabong core is not apparent in Sandy Billabong.

Steps for completion

The method using trace metals and stable lead isotope ratio analysis is now being investigated for its potential to assess impacts from non-uranium mine sites. Samples have been taken from the vicinity of the former Mount Todd gold mine and are currently being analysed.

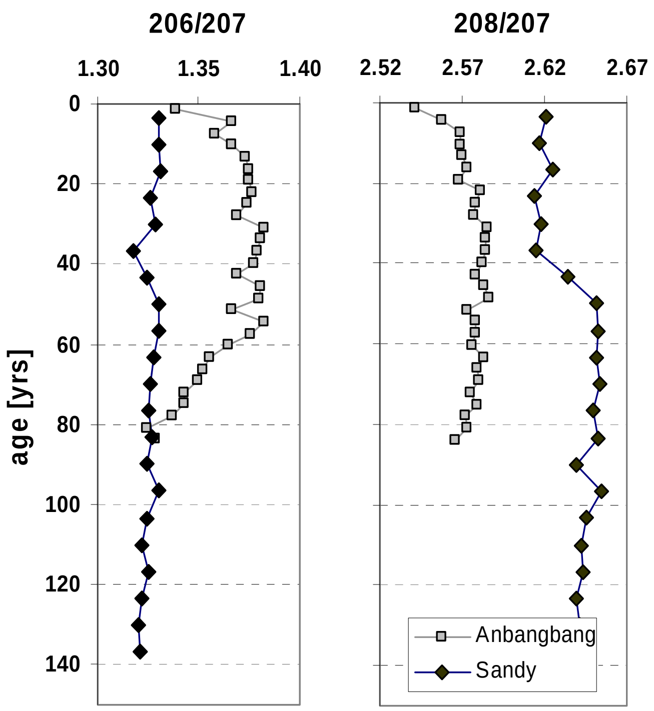


Figure 2 $^{206}Pb/^{207}Pb$ and $^{208}Pb/^{207}Pb$ ratios plotted versus age in both, Sandy and Anbangbang Billabong cores (from Frostick et al 2010)

References

Bollhöfer A, Honeybun R, Rosman KJR & Martin P 2006. The lead isotopic composition of dust in the vicinity of a uranium mine in northern Australia and its use for radiation dose assessment. *Sci. Total Environ.* 366, 579–589.

Bollhöfer A & Martin P 2003. Radioactive and radiogenic isotopes in Ngarradj (Swift Creek) sediments: a baseline study. Internal Report 404, February, Supervising Scientist, Darwin. Unpublished paper.

Bosch D, Hammor D, Bruguier O, Caby R & Luck JM 2002. Monazite 'in situ' $^{207}Pb/^{206}Pb$ geochronology using a small geometry high-resolution ion probe. Application to Archaean and Proterozoic rocks. *Chem. Geol.* 184, 151–165.

Dean JA & Gulson BL 1987. Biogeochemical prospecting using lead isotopes. *J. Geochem. Explor.* 29, 391–392.

Dickson BL, Gulson BL & Snelling AA 1985. Evaluation of lead isotope methods for uranium exploration, Koongarra area, Northern Territory, Australia. *J. Geochem. Explor.* 24, 81–102.

Esparon A & Pfitzner J 2010. Visual Gamma 2009, Gamma Analysis Manual. Internal Report 539. Supervising Scientist, Darwin. Unpublished paper.

Frostick A, Bollhöfer A, Parry D, Munksgaard N & Evans K 2008. Radioactive and radiogenic isotopes in sediments from Cooper Creek, Western Arnhem Land. *J. Environ. Radioact.* 99, 468–482.

Frostick A, Bollhöfer A & Parry D 2010. A study of radionuclides, metals and stable lead isotope ratios in sediments and soils in the vicinity of natural U-mineralisation areas in the Northern Territory, Australia. submitted to the *Journal of Environmental Radioactivity*.

Gulson BL 1986. *Lead isotopes in mineral exploration*. Elsevier, Amsterdam.

Gulson BL & Mizon KJ 1980. Lead isotope studies at Jabiluka. In *Uranium in the Pine Creek Geosyncline. Proceedings of the International Uranium Symposium on the Pine Creek Geosyncline.* eds Ferguson J & Goleby AB, International Atomic Energy Agency, Vienna, 439–455.

Gulson BL, Mizon KJ, Korsch MJ, Carr GR, Eames J & Akber RA 1992. Lead isotope results for waters and particulates as seepage indicators around the Ranger tailings dam: A comparison with the 1984 results. Open file record 95, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Marten R 1992. Procedures for routine analysis of naturally occurring radionuclides in environmental samples by gamma-ray spectrometry with HPGe detectors. Internal report 76, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Munksgaard NC, Lim K & Parry DL 2003. Rare earth elements as provenance indicators in North Australian estuarine and coastal marine sediments. *Est. Coast. Shelf Sci.* 57, 399–409.

Murray AS, Marten R, Johnston A & Martin P 1987. Analysis for naturally occurring radionuclides at environmental concentrations by gamma spectrometry. *J. Radioanal. Nucl. Chem. Art.* 115, 263–288.

Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on Gulungul Creek catchment, Ranger mine site and Nabarlek area

K Evans & D Moliere[1]

Introduction

Severe category 5 tropical cyclone Monica crossed the Northern Territory coastline near Maningrida on 24 April 2006 and passed through the Magela Creek catchment near the Ranger Mine on 25 April 2006 (Figure 1), approximately 9 hours after landfall having reduced intensity to a Category 2 cyclone. Treefall damage, caused by high wind velocities, occurred around lower regions of the Magela Creek catchment, including the area surrounding the Ranger mine (Staben & Evans 2008). The treefall damage occurred primarily in areas with saturated soils (ie the terraces and palaeochannels which flank the creek channels).

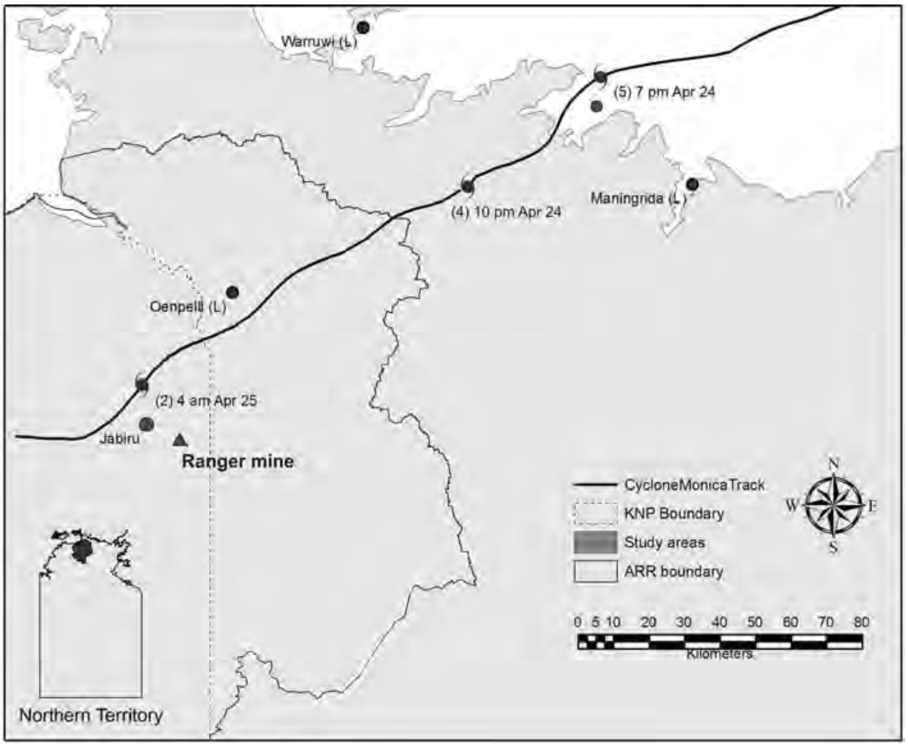


Figure 1 Track of Cyclone Monica over the Top End

The cyclone contributed to high stream suspended sediment concentrations in Magela Creek *during* the event (Figure 2). However, it is also important to assess the subsequent changes to the catchment sediment transport characteristics as a result of the substantial treefall

---

1 Water Division – Bureau of Meterology, NT Regional Office

associated with this event, since such information can inform the risk assessment process for determining possible effects of future similar events on the catchment lines containing the rehabilitated Ranger minesite.

Suspended sediment data collected at Magela Creek Upstream (MCUS) and Magela Creek Downstream (MCDS) (Map 2) during 2006–07 and 2007–08 have been compared to suspended sediment data collected prior to the cyclone (2005–06 data) to assess the post-cyclone behaviour of the system. In particular, an assessment was made to determine whether or not a change in sediment transport characteristics has occurred in Magela and Gulungul Creeks and the duration of such impacts.

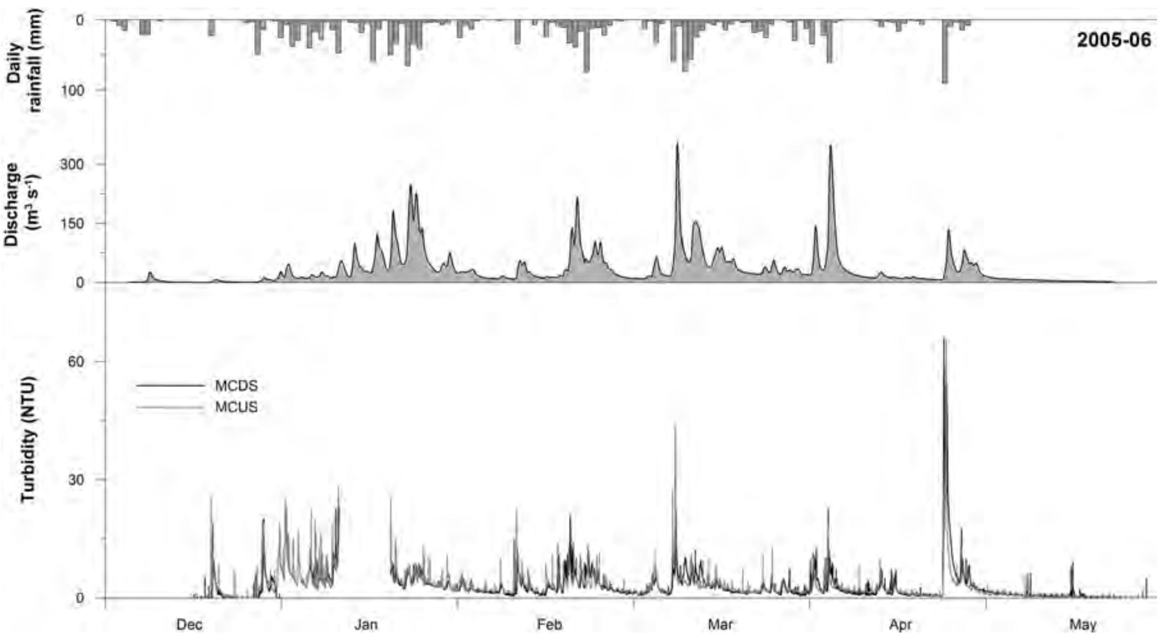


Figure 2 Annual hydrograph and sedigraph for Magela Creek (location) during 2005–06 highlighting the annual peak in turbidity on 25 April 2006 associated with the cyclone

Results

Fine suspended sediment (FSS – <63 µm and >0.45 µm diameter) was monitored at MCUS and MCDS using as a surrogate, turbidity data arising from field-calibrated turbidity sensors. A statistically significant relationship between laboratory-determined FSS concentration and field-derived turbidity was used to indirectly measure continuous FSS concentration.

The continuous turbidity data were used to quantify the FSS loads transported by an FSS pulse event, rather than assessing instantaneous FSS concentrations in the stream, an approach which is typically complicated by hysteresis effects[2] (Moliere et al 2005). FSS load during a pulse is defined as the area under the event sedigraph, where FSS concentration rises above, and then returns to, approximate base-flow levels (2–5 mg L^{-1}).

[2] The hysteresis effect occurs when the same FSS concentration value is measured during the rising stage and falling stage of an event hydrograph but for different instantaneous discharge values. This effect is a significant complicating issue when developing regression relationships between instantaneous FSS concentration and instantaneous discharge.

Two methods based on event FSS loads were used to determine whether there has been a change in sediment transport characteristics within Magela Creek as a result of Cyclone Monica:

1 A student *t*-test based on a Before-After-Control-Impact Paired difference design (BACIP) (Stewart-Oaten et al 1986, 1992, Humphrey et al 1995)

2 A regression relationship between event FSS load and corresponding event discharge characteristics.

Student *t*-test based on BACIP

For this work the BACIP analysis uses the differences between event FSS loads measured at MCDS and MCUS and between two time periods, to test for significant change between the two intervals. In particular, the difference data distributions observed for events prior to the cyclone (2005–06 wet season) were compared to those events observed post-cyclone (2006–07 and 2007–08 wet seasons). A student *t*-test (assuming equal variances) showed that the differences in the event FSS loads for post-cyclone events during 2006–07 were not significantly different to those for events observed prior to the cyclone (2005–06 data). Similarly, post-cyclone events during 2007–08 were also not significantly different from the 2005–06 event data indicating that the relative difference in event FSS loads between the two stations has not changed as a result of tree damage associated with Cyclone Monica along the Magela Creek channel.

Relationship between FSS load and discharge characteristics

Event FSS load and corresponding event discharge characteristics for MCUS and MCDS collected during the 2005–06 wet season were used to fit a relationship of the form (Moliere & Evans 2009, 2010):

$$\text{Total FSS load} = K\left(Q_T\right)^a Ri^b \qquad (1)$$

where Q_T is total discharge during the FSS pulse, Ri is maximum periodic rise in discharge over 10 minutes and a, b and K are fitted parameters.

Observed event FSS loads measured for rainfall/discharge events for each site were then plotted against the predicted FSS loads, derived using equation (1) where Q_T and Ri were applied to the observed discharge for the event for which observed FSS load was measured (Figure 3). The fitted line (1:1) represents the ratio between observed and fitted event loads.

Event FSS load data observed during 2006–07 and 2007–08 at each station were plotted against the loads predicted using the fitted relationships derived for pre-cyclone conditions (Figure 3). Data points that plot above the 1:1 line indicate an observed event load greater than the expected (predicted) load for a particular discharge event and data points below the 1:1 line represent observed FSS loads less than expected for that discharge event. Observed event loads that plot above the +2 standard deviation (SD) line have a significantly higher FSS load than would be expected for the corresponding event discharge characteristics for the pre-cyclone condition.

All FSS load events observed during the first wet season after the cyclone (2006–07), at MCUS and MCDS respectively, plotted above the 1:1 line (Figure 3) even though only a few events at both stations fell outside +2 SD of the fitted relationship. An *F*-test showed that for both sites, a relationship fitted using 2006–07 data was significantly different to that fitted using 2005–06 event data (at the 95% level). This indicates a significant increase in FSS loads relative to discharge characteristics along Magela Creek during 2006–07 compared with that observed prior to the cyclone.

All observed events in 2007–08 (except for one at each station) plotted within +2 SD and around the 1:1 line for the 2005–06 fitted relationship (Figure 3). An *F*-test showed that a relationship fitted using 2007–08 data was not significantly different from the relationship fitted using pre-cyclone (2005–06) event data (at the 95% level). Therefore, event FSS loads for 2007–08 relative to the event discharge characteristics are not significantly different to pre-cyclone conditions at both stations.

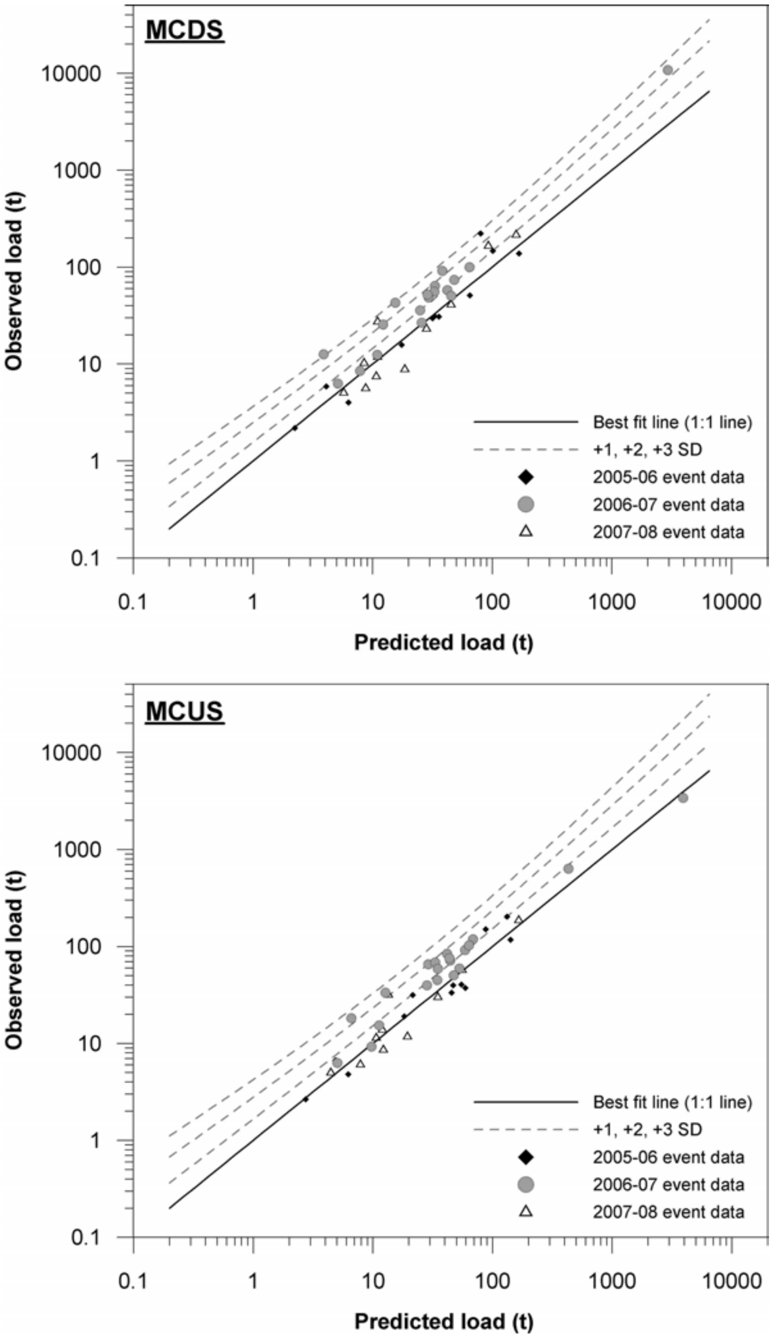


Figure 3 Fitted lines for relationship between observed and predicted event FSS loads and associated +1, +2, and +3 standard deviation lines for MCDS (Top) and MCUS (Bottom). Discrete event data collected during 2005–06 (used to fit the relationships) and the post cyclone (2006–07 and 2007–08) event data are also shown.

This analysis indicates that the effects of Cyclone Monica caused a significant impact on sediment transport characteristics within the catchment during the first year after the cyclone. There was a general increase in stream suspended sediment concentrations and suspended sediment loads relative to discharge measured at both stations along Magela Creek. By the second year after the cyclone (2007–08) the system had returned to pre-cyclone FSS transport conditions.

Comparison of approaches

The student *t*-test using BACIP results shows that there was no significant change in the relative differences in event FSS loads between the two stations as a result of the cyclone. However, using this approach it is not possible to distinguish if there has been a similar change in sediment transport characteristics at both sites, which was indicated to have occurred by the regression model. It is considered that while the BACIP approach may be a good method to assess an impact that occurs *between* the two stations (i.e. a mining-related impact), it is not an appropriate method for assessing the impact of a catchment-wide impact (such as cyclone, fire or extreme flood event) where both upstream and downstream sites are significantly and similarly impacted.

The regression approach detected a drainage system (ie broad scale catchment) impact since suspended sediment concentrations and loads significantly increased along Magela Creek as a result of the cyclone. This approach also clearly illustrated the subsequent recovery of the system by returning of the fitted regression line (for 2007–08 data) to its original position, indicating that sediment transport in the creek system had returned to pre-cyclone conditions. It is likely that much of sediment made available by treefall was washed away during the major flood event in late February and early March 2007 (Moliere et al 2008). This event completely inundated the terraces and palaeochannels alongside the main channel where most of the treefall occurred.

Application of the regression analysis method has shown that riparian zone treefall associated with a Category 2 cyclonic event can have a significant, although relatively short-lived. effect on sediment transport characteristics within a catchment,.

This was the only progress made on analysis of the Cyclone Monica data base this year due to changes in ***eriss*** staff structure and competing priorities with the trial landform. However, all field survey data and soil survey data were published as SSD Internal Reports in March 2009 (Saynor et al 2009a, b).

Future work

Future work will include finalising the statistical analysis of impacts on vegetation and sediment transport rates in the Gulungul Creek catchment, and other sites where data were collected such as Ranger and Nabarlek. No further work will be specifically undertaken to assess impacts of Cyclone Monica on sediment loads in Magela Creek. Monitoring will continue in the 'Turbidity and suspended sediment management guidelines and trigger values for Magela Creek' project under KKN 2.5.2 (Off-site monitoring during and following rehabilitation) – and ongoing BACIP and regression analysis analyses undertaken to detect any minesite impacts and to track any systematic changes in the overall sediment delivery behaviour of the catchment.

References

Humphrey CL, Faith DP & Dostine PL 1995. Baseline requirements for assessment of mining impact using biological monitoring. *Australian Journal of Ecology* 20, 150–166.

Moliere DR & Evans 2009. Cyclone Monica and the resulting impacts of treefall on stream sediment transport. In *Adapting to change*, Proceedings of the 32nd Hydrology and Water Resources Symposium, 30 November – 3 December 2009, Newcastle, Engineers Australia (CD).

Moliere DR & Evans KG 2010. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research* DOI: 10.1111/j.1745-5871.2010.00641.x

Moliere DR, Evans KG & Turner KF 2008. Effect of an extreme storm event on catchment hydrology and sediment transport in the Magela Creek catchment, Northern Territory. In *Water Down Under 2008, Proceedings of the 31st Hydrology and Water Resources Symposium and 4th International Conference on Water Resources*, 15–17 April 2008. Adelaide, South Australia, Engineers Australia (CD).

Moliere DR, Saynor MJ & Evans KG 2005. Suspended sediment concentration-turbidity relationships for Ngarradj – a seasonal stream in the wet-dry tropics. *Australian Journal of Water Resources* 9(1), 37–48.

Saynor MJ, Houghton R, Hancock G, Staben G, Smith B & Lee N 2009b. Soil sample descriptions – Gulungul Creek, Ranger mine site and Nabarlek: Cyclone Monica fieldwork. Internal Report 558, Supervising Scientist, Darwin. Unpublished paper.

Saynor MJ, Staben G, Hancock G, Fox G, Calvert G, Smith B, Moliere DR & Evans KG 2009a. Impact of Cyclone Monica on catchments within the Alligator Rivers Region – Data. Internal Report 557, Supervising Scientist, Darwin. Unpublished paper.

Staben GW & Evans KG 2008. Estimates of tree canopy loss as a result of Cyclone Monica, in the Magela Creek catchment northern Australia. *Austral Ecology* 33, 562–569.

Stewart-Oaten A, Bence JR & Osenberg CW 1992. Assessing effects of unreplicated perturbations: No simple solutions. *Ecology* 73, 1396–1404.

Stewart-Oaten A, Murdoch WW & Parker KR 1986. Environmental impact assessment: ‘Pseudoreplication’ in time? *Ecology* 67, 929–940.

Assessment of suspended sediment movement upstream and downstream of Ranger

K Evans & D Moliere[1]

Background

Event loads of fine suspended sediment (FSS) obtained during the three wet seasons from 2005–06 to 2007–08 from upstream (MCUS) and downstream (MCDS) of Ranger along Magela Creek were used to derive proposed guideline trigger values for suspended sediment. The trigger values were based upon attributes of the statistical distribution of FSS data derived from the creek in periods of no net inputs from the mine (ie a 'reference' condition), using two methods (1) using a Before-After-Control-Impact-Paired differences (BACIP) design and (2) using regression analysis. The results were reported at ARRTC 22 (see Moliere et al 2007).

Progress

In 2008–09 the above analysis was written up as an SSD internal report and a journal paper. The internal report is being reviewed and the paper has been accepted for publication in *Geographical Research* (Moliere & Evans 2010).

The 2005–08 data set has also been used to analyse the likely impact that Cyclone Monica (2005–06 wet season) had on suspended sediment transport in Magela Creek in the two wet seasons following this event. The findings from this work are reported in this volume (see 'Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on Gulungul Creek catchment, Ranger mine site and Nabarlek area', pp 179–184, in this volume).

Continuous rainfall, stream discharge, electrical conductivity and turbidity data were collected for the 2008-09 wet season (Figure 1) but have not yet been analysed or reported due to staffing changes and competing priorities.

Future work

Continuous monitoring data will continue to be collected. The data are used to update and revise the trigger values, and to detect and assess possible mine-associated changes in FSS in Magela Creek, using a combination of BACIP and regression model analysis. Work to date has identified one possible mine-related change in FSS event loads measured downstream of Ranger since 2005. It is recommended that impact assessment of FSS loads continue to use both statistical approaches, given their complementary strengths and weaknesses (see 'impact of Cyclone Monica' paper, pp 179–184, in this volume, for a description of how the two approaches may be applied).

[1] Water Division – Bureau of Meterology, NT Regional Office

Future measured event load data which plot above the 'action' trigger levels derived from these methods of sediment load analysis should prompt further investigation and management action, if required. The investigation should include chemical analysis of the suspended sediment to ascertain if it has come primarily from the minesite or from adjacent un-impacted subcatchment areas. This distinction needs to be made since elevated sediment loads could arise from an intense local storm event on land, unassociated with mining activity itself, yet lying within the mine lease boundary. Water samples for this analysis will be provided by the event-triggered autosamplers located at the downstream site in Magela Creek.

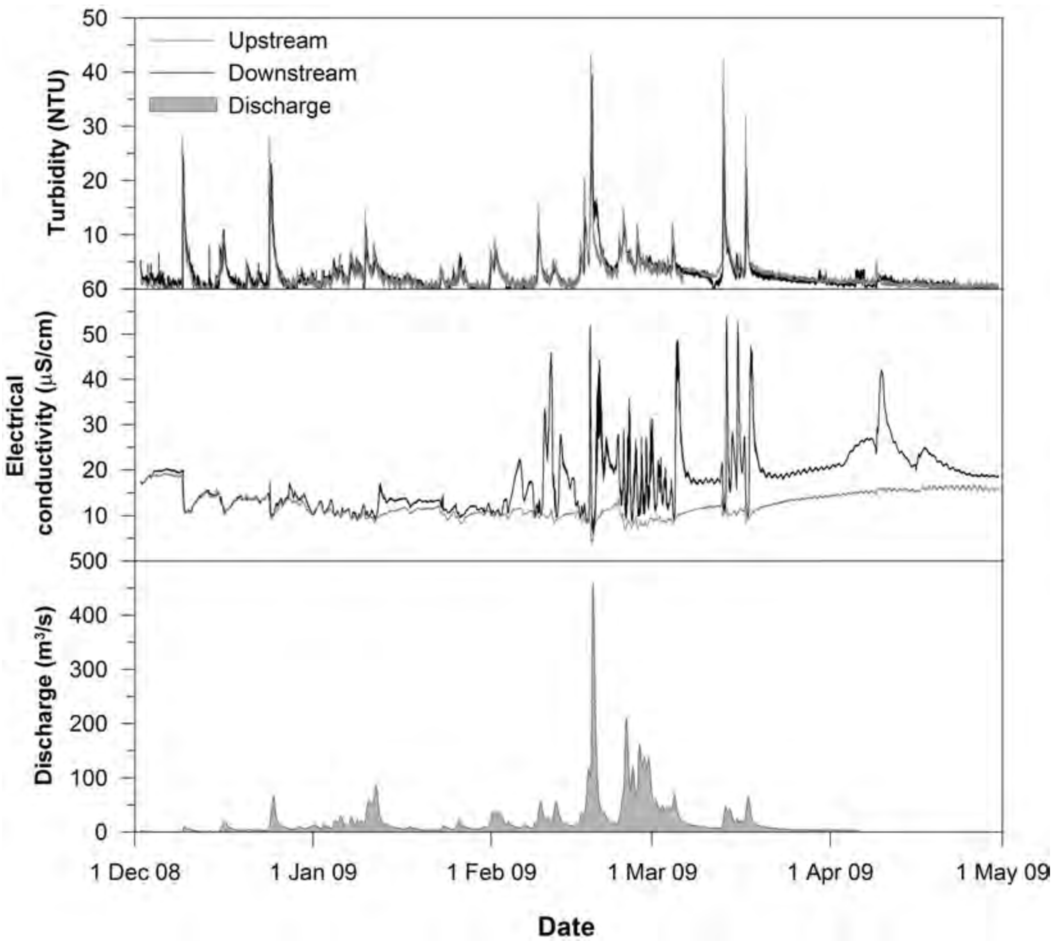


Figure 1 Magela Creek turbidity and electrical conductivity measured upstream and downstream of the mine and discharge measured at G8210009

References

Moliere DR & Evans KG 2010a. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research* DOI: 10.1111/j.1745-5871.2010.00641.x

Moliere D, Evans K & Turner K 2007. Assessment of continuous Magela Creek turbidity data upstream and downstream of Ranger. In ***eriss*** *research summary 2006–2007*. Supervising Scientist Report 196, Supervising Scientist, Darwin NT, 124–130.

Part 3: Jabiluka

Monitoring sediment movement in Ngarradj

K Evans, K Turner & M Saynor

Background

During the 2008–09 wet season, continuous rainfall, stream discharge, turbidity and electrical conductivity (EC) data were collected from the Swift Creek (SC) continuous monitoring station, located immediately downstream of the Jabiluka project area within the Ngarradj catchment. Baseline suspended sediment and hydrology data have now been monitored at this site over a 11 year period. Data collection at two stations upstream of Jabiluka (ET and UM) ceased after the 2006–07 wet season, following discussion and agreement with both the Jabiluka Minesite Technical Committee and ARRTC. The locations of the three monitoring stations are shown in Figure 1.

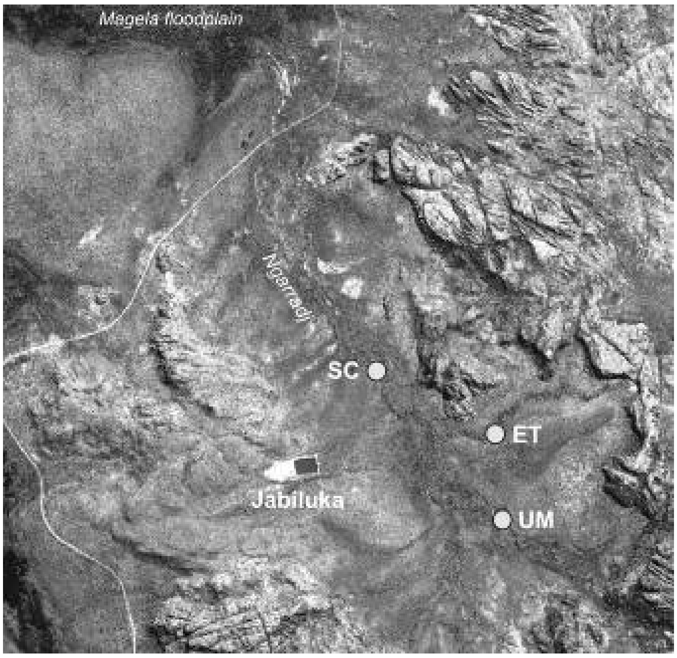


Figure 1 Location of the monitoring stations along Ngarradj

The fine suspended sediment (FSS) data, measured indirectly from turbidity, together with hydrology data collected from the sites, will be used to derive indicators for minesite impact in the event that the Jabiluka project proceeds. Data collected since the 2006–07 wet season are especially important in the context of Cyclone Monica, which occurred in April 2006. In particular, comparison between the suspended sediment load data collected before and after this event may indicate whether there has been a change in 'pre-mining' sediment transport characteristics as a result of this catchment-wide impact.

Progress

An extraordinary rainfall event occurred over a 3-day period between 27 February and 2 March 2007, which resulted in the highest flood levels recorded within the Ngarradj catchment since monitoring commenced in 1998. The SC station was submerged by floodwaters, resulting in damage to monitoring equipment and loss of data during the peak and subsequent recession of the flood. Consequently, the SC continuous monitoring station underwent an upgrade which included installation of new instruments and elevation of the instrument platform to at least 1 m above the peak water level recorded during the March 2007 flood. This upgrade was similar to that described for the Gulungul Creek stations in Moliere et al (2009). As a result of the upgrade, the SC station is now integrated into SSD's continuous monitoring program which also includes Gulungul and Magela Creeks. Due to the location of the SC station, dial up telemetry access is not possible up to an including 2008–2009 as it lies outside mobile network coverage.

The continuous monitoring data collected at SC during the 2008–09 wet season have not yet been analysed, owing to staffing changes and competing commitments. However, all indications from the water quality grab sampling program are that measured parameters remained within the baseline range of values for this wet season. The hydrograph and rainfall over the 2008–09 wet season are shown in Figure 2.

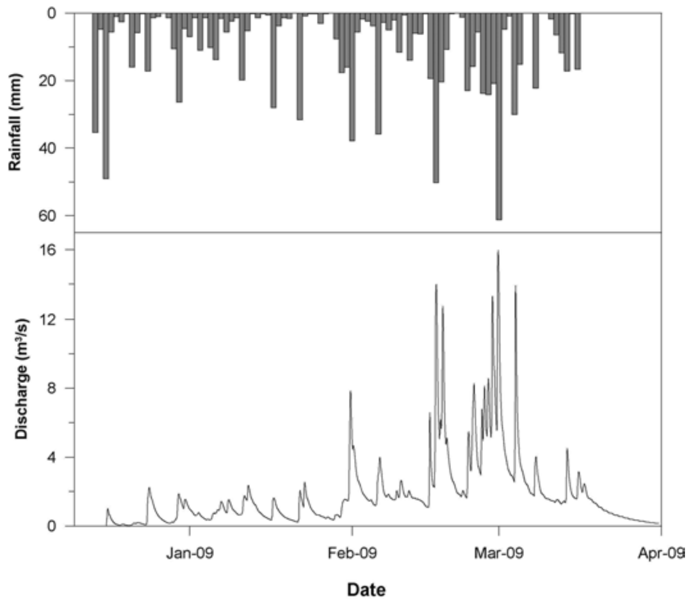


Figure 2 Discharge (m³/s) and rainfall (mm) measured at the SC continuous monitoring station during the 2008–09 wet season

Future work

Event FSS load data collected during the 2008–09 wet season will be plotted against the relationship between FSS load and discharge characteristics derived using data from previous years (Moliere & Evans 2010) to assess the sediment load characteristics for the 2008–09 wet season.

References

Moliere DR & Evans KG 2010a. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research* DOI: 10.1111/j.1745-5871.2010.00641.x

Moliere D, Staben G, Saynor M & Houghton R 2009. Development of catchment geomorphic characteristics of Gulungul Creek – gauging station upgrades. In ***eriss*** *research summary 2007–2008*. Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 170–171.

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

Remediation of the remnants of past uranium mining activities in the South Alligator River Valley

A Bollhöfer, B Ryan, M Fawcett[1], K Turner & D Jones

Introduction

The upper South Alligator River valley in the south of Kakadu National Park is both a popular tourist destination and a region in which past uranium exploration, mining and milling activities have occurred. The locations of these former uranium mine sites are marked on Figure 1.

Mining in the area started with the discovery of the Coronation Hill deposit in 1953, and continued through to 1964. During that time, approximately 877 tonnes of U_3O_8 were produced from 13 small scale uranium mines (Waggitt 2004). When mining ceased, no substantial effort was made to clean up and rehabilitate the mine and mill areas or camps.

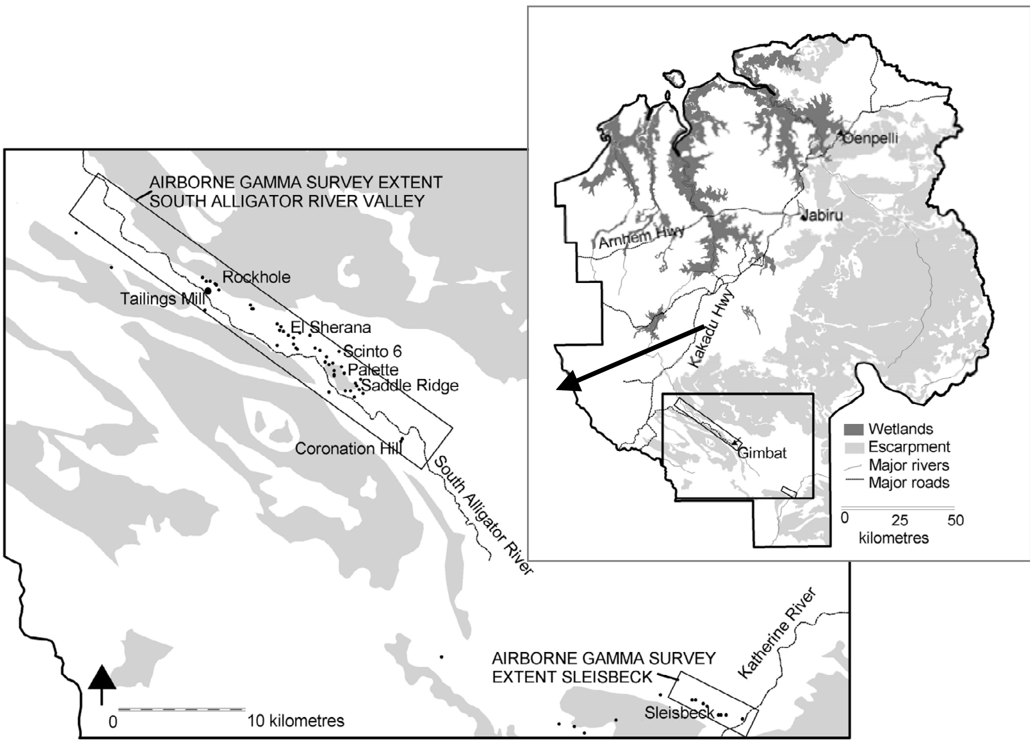


Figure 1 Alligator Rivers Region, with a detailed excerpt of the southern area showing the extent of two airborne gamma surveys conducted in 2000 and 2002, the location of known uranium anomalies (from MODAT database) and some historic mining and milling areas (Supervising Scientist 2003)

Radioactive tailings were discovered by staff from SSD in 1984 next to the road to the Gunlom waterfall, and close to the South Alligator River, during a ground gamma radiation survey. The fine-grained tailings originated from the Rockhole mill, where over 13 400

[1] Fawcett Mine Rehabilitation Services, Darwin

tonnes of high-grade uranium ore were processed in the 1950s and 1960s. Subsequently, rehabilitation works were conducted in 1990–92, with most of the tailings removed. Residuals of the tailings subsequently found to be located on erodable terrain adjacent to the river (Tims et al 2000) were covered with rock armour (in 2000) to prevent erosion of the material into the river. Other small historic mining sites were, at that stage, not considered a priority for rehabilitation because they were either largely inaccessible to the public, relatively stable and/or did not contain radioactive tailings.

In 1996, land granted to the Gunlom Aboriginal Land Trust was leased back to the Director of National Parks to be managed as part of Kakadu National Park. The lease agreement required the Director of National Parks to implement an environmental rehabilitation plan for the historic minesites and associated workings in the South Alligator River valley. This plan is managed by Parks Australia. SSD is providing specialist assistance with the radiological assessment of the sites.

Airborne gamma surveys were flown over the South Alligator River valley in 2000 (Pfitzner & Martin 2000, Pfitzner et al 2001) and over the Sleisbeck area in 2002 (Bollhöfer et al 2008). The results from these surveys were used to identify the location, extent and magnitude of residual radiological contamination. Areas exhibiting radiation levels above local background values were subsequently surveyed in more detail by ground measurements (Bollhöfer et al 2002, Bollhöfer et al 2007). The results of these investigations have aided the development of a rehabilitation strategy for the South Alligator River valley. The works for this are nearing completion.

Radiological assessment of the area continued through 2008–09 to provide final details of those sites that may require additional attention to remove remaining above-background materials. Two historic minesites, Palette and El Sherana, were investigated by grid-surveys. In addition, a post-remediation radiation survey was conducted at the Sleisbeck mine in the Katherine River catchment to document the success of the works that were carried out during the 2007 dry season. The Sleisbeck site is approximately 30 km south-east of Guratba. Figure 1 shows the location of these sites.

Results

Status of Palette mine

Palette mine was worked from 1956 to 1961 and produced ~120 tonnes U_3O_8 from high grade uranium ore (Fisher 1968). While mining occurred mainly in open stopes, there are also a number of adits in the area. The mine area is difficult to access. It is located less than 1 km to the east of the Koolpin access track, approximately 220 m above sea level.

The highest pre remediation gamma dose rate at Palette was 5 $\mu Gy \cdot hr^{-1}$, measured on the top bench, with typical values ranging between 1.4 and 1.7 $\mu Gy \cdot hr^{-1}$. During a meeting between Parks Australia, the Supervising Scientist and consultants involved in the rehabilitation works, a guideline value for the gamma dose rate applicable to the rehabilitation of historic mining and milling sites in the South Alligator River valley was set at 1.25 ± 0.25 $\mu Gy \cdot hr^{-1}$ (which is approximately 10 times higher than background levels). This guideline value was derived purely on the basis of being able to distinguish the radiological signal from the regional background, and it should be regarded as a screening value. Application of this value as a cleanup threshold will result in annual effective doses to members of the public being well below the 10 mSv dose constraint recommended by the International Commission for Radiological Protection (ICRP 2007) for the rehabilitation of existing exposure situations.

This applies even in the unlikely case that the remediated areas were permanently occupied for a couple of months per year.

About two thirds of the surveyed area at Palette mine, in particular the top bench, showed gamma dose rates above the screening guideline value. The top bench has been remediated and scraped in September 2009. The material has been buried at the new containment at the El Sherana Airstrip.

Status of El Sherana mine

The El Sherana mine area was worked from 1956 to 1964 and produced ~400 tonnes U_3O_8 (Fisher 1968). The ore grade was lower than at Palette but was still comparatively high at up to 0.82%. Two areas were worked: the El Sherana pit on the hill top and El Sherana West in the valley located approximately 500 m north-east of the El Sherana camp. The airborne gamma survey from 2000 indicated that the El Sherana pit was the main source of above-background radiation in the area and so ground surveys focused on that area.

The highest dose rate on top of the El Sherana pit was 14 µGy/hr, 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. Figure 2 shows a contour plot of dose rates measured on top of the pit. It appears that some radiological material was eroding towards the northwest, coincident with flow lines established from the local topography. Approximately 7000 m^2 were surveyed within the fenced area; approximately 4800 m^2 was found to exceed the 1.25 µGy/hr threshold value.

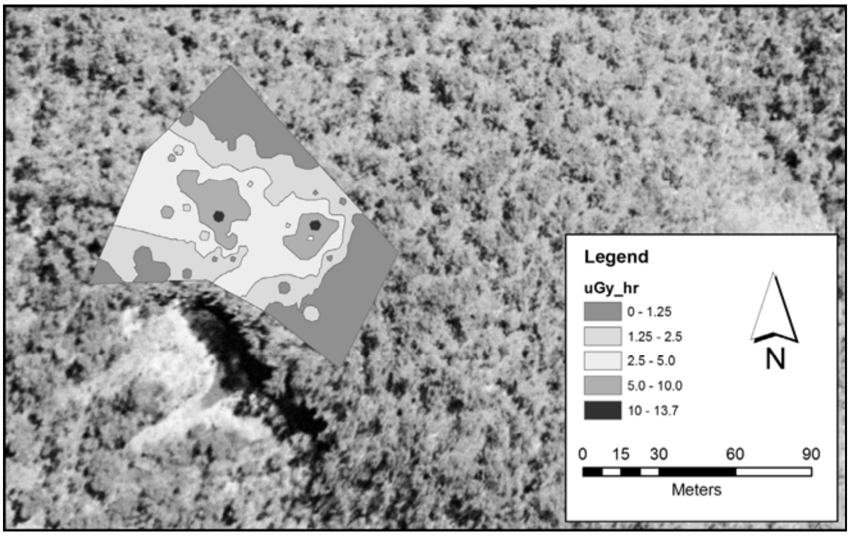


Figure 2 Dose rate contours [µGy/hr] on top of the El Sherana pit

In addition, the bottom of the El Sherana pit and associated workings, consisting of two waste piles and four benches in the pit wall to the south-east of the pit, were surveyed in December 2008. The bench on top of the pit wall exhibited gamma dose rates of approximately 1 µGy/hr. The remaining three benches, the bottom of the pit, and the two waste piles exhibited average gamma dose rates of about 2 µGy/hr and above. There was a small area of mineralisation accessible from one of the benches that exhibits gamma dose rates of above 7 µGy/hr. To cover the mineralisation and to reduce average gamma dose rates in the area, the material from the two waste piles was shifted and pushed against the benches to the south-east of the pit, and subsequently covered with background material. The area has now been remediated and the contaminated material has been transported to the new containment at the El Sherana Airstrip.

Assessment of the rehabilitated Sleisbeck minesite

The Sleisbeck mine was worked in 1957 but only a little over 2 tonnes of U_3O_8 was produced before the mine was abandoned. The rehabilitation of this site, comprising a water-filled open pit and surface waste dumps with a substantially above background radiological signature was undertaken in the dry season of 2007. The waste rock and low grade material from the truck dumps to the south of the pit were removed and placed into the pit. The pit backfill was shaped to cover a mineralised area in the pit wall that exhibited very high external gamma dose rates of above 30 µGy/hr.

Top cover material with background radiological signature was sourced from old spoil piles located to the east of the Sleisbeck pit. This material was spread over the surface of the backfilled pit in a single layer to a nominal depth of 700 mm. The second source of cover material was from a disused track to the north-east of the pit, which provided material for the final upper 300 mm cover layer (Waggitt & Fawcett 2008). Rehabilitation works were finalised in December 2007.

A detailed ground survey of the rehabilitated footprint was carried out in 2008 to confirm that the radiological objectives of the works had been achieved (Bollhöfer & Fawcett 2009). Figure 3 shows a probability plot of the gamma dose rates. Geometric and arithmetic averages measured across the 7.6 ha surveyed are 0.14 and 0.23 µGy/hr, respectively. Assuming a lognormal distribution, the plot shows that 99% of the area surveyed has gamma dose rates below the screening value of 1.25 µGy/hr. There is a small area immediately to the east of the old access track to the rehabilitated pit where gamma dose rates of above 3 µGy/hr were measured. This area is part of an old access track to the pit, and mineralised material may have been used as road fill. It comprises less than 0.01% of the area.

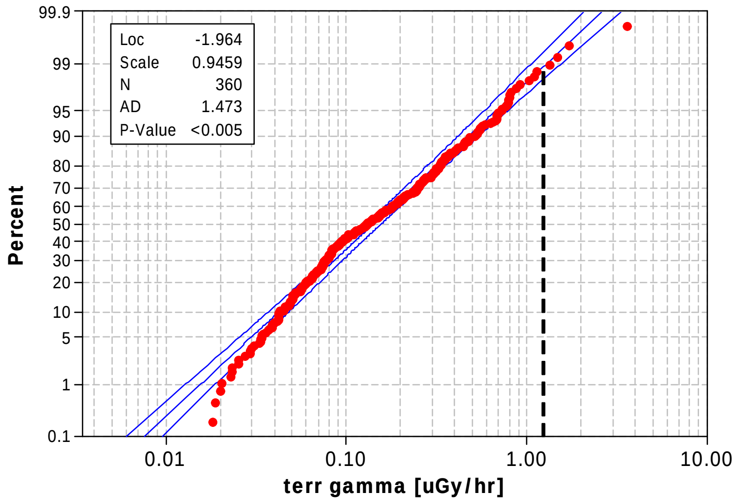


Figure 3 Probability plot of terrestrial gamma dose rates at Sleisbeck post rehabilitation. The probability plot shows that ~99 % of the area surveyed exhibits terrestrial gamma dose rates below 1.25 µGy/hr.

The successful rehabilitation of the old truck dumps and pit area at the Sleisbeck mine has reduced the average terrestrial gamma dose rates in the area by about threefold. Assuming the unlikely scenario that the site is occupied for one month per year, the average terrestrial gamma dose rate on site will lead to an effective dose from exposure to terrestrial gamma radiation of ~ 0.1 milli Sievert. Approximately half of this dose will originate from exposure to natural background radiation. These doses are well below the annual dose constraint for the public for existing exposure situations of 10 mSv, and even lower than the 0.3 mSv dose constraint recommended in current ICRP (2007) guidelines for prolonged exposure from planned exposure situations.

Assessment of rehabilitation requirements for Rockhole Mine Creek

In contrast to the situations described above where the primary rehabilitation requirements relate to solid materials produced by mining, Rockhole Mine Creek is a case of a receiving waterway receiving contaminated drainage originating from abandoned mine workings.

Rockhole Mine Creek (RMC) is a small tributary of the upper South Alligator River that receives low level inputs of acidic and metal-rich seepage water from the former Rockhole minesite. The water is flowing at a low rate (0.2–0.4 L/s) from the lower adit draining the abandoned Rockhole mine workings (see Figure 4).

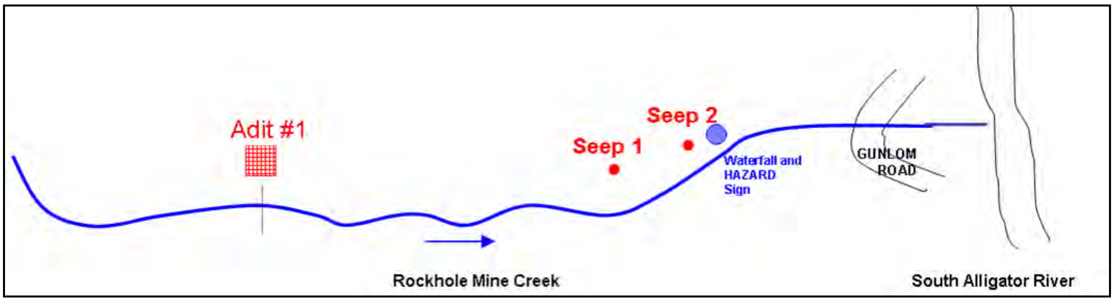


Figure 4 Schematic (not to scale) of Rockhole Mine Creek showing the location of Adit 1 and two downstream seeps

SSD has completed an assessment of the downstream effects of this input to advise Parks Australia on whether specific remediation is needed. The review draws upon a long history (since 1988) of field investigations, an earlier SSD review prepared for the NT World Heritage Ministerial Council in 2000, the results of field investigations conducted for Parks in 2000 and 2001 by Earth Water Life Sciences (EWLS) Pty Ltd, and a subsequent program of stream monitoring undertaken by SSD between 2002 and 2009.

The earlier reviews and reports by SSD and EWLS in the early 2000s concluded that: (i) there were no significant radiological issues in the creek; (ii) although there was substantial iron staining (an aesthetic issue) along the channel of the creek, this iron was also coming from seeps further downstream of the adit (see Figure 4); (iii) though there were detrimental effects on the ecology of RMC, these effects did not extend to the South Alligator River; and (iv) RMC was not considered to be of significant cultural value to Traditional Owners. Given this background, it was concluded that unless the risk to the downstream environment could be shown to be increasing through time (viz increasing loads of potentially toxic metals, or inputs of radionuclides), there would be no justification for carrying out specific remediation works at the adit. In particular, and given the multiple inputs of iron to RMC, there was no guarantee that remedial works such as plugging the adit or treating the adit waters would necessarily lead to removal of the iron staining and deposition of iron precipitates.

To determine if contaminant loads were increasing through time, an extended program of monitoring to track the composition of the adit water was carried out by SSD between 2002 and 2009.

The recently-completed review (Turner et al 2009) found that over the 20-year water quality record (1988–2009), iron and manganese are the only contaminants that have systematically increased (manganese only slightly) in concentration in the RMC adit outflow. In contrast, the concentrations of the potentially toxic metals, aluminium, copper, lead, zinc and uranium have, overall, decreased significantly on a year by year basis. Subsequent and complementary work conducted by SSD (Ryan et al 2008) has also shown no significant bioaccumulation of ^{226}Ra in mussels in the South Alligator River as a result of input of adit water to RMC.

Given that concentrations of metals of greatest concern to ecosystem health have declined over time, this finding supports earlier recommendations that no remedial action is required in RMC. While iron concentrations have increased in the adit water, this needs to be considered in the context of substantial amounts of iron also being contributed by seeps downstream of the adit. Given that it will not be possible to stop the flow from these distributed downstream sources, there would be no benefit to be gained by remediating the adit source of iron. SSD has recommended future opportunistic sampling of adit waters to confirm on an ongoing basis that iron continues to be the only contaminant in the water that is significantly affecting RMC.

Steps for completion

Post remediation surveys need to be conducted at the Rockhole Residues site in 2009, and at the remediated Palette and El Sherana sites in the dry season 2010.

Acknowledgments

Therese Fox, Nadine Riethmuller and Rob Muller are thanked for support before and during the fieldwork.

References

Bollhöfer A, Ryan B, Pfitzner K, Martin P & Iles M (2002). A radiation dose estimate for visitors of the South Alligator River valley, Australia, from remnants of uranium mining and milling activities. In *Uranium Mining and Hydrogeology III*, eds BJ Merkel, B Planer-Friedrich & C Wolkersdorfer. Technical University, Bergakademie Freiberg, 931–940.

Bollhöfer A, Dunn L, Ryan B & Sellwood J 2007. Radiological field investigations at the Rockhole Residue Site, South Alligator River Valley, Australia. Internal Report 529, September, Supervising Scientist, Darwin. Unpublished paper.

Bollhöfer A, Pfitzner K, Ryan B, Martin P, Fawcett M & Jones DR 2008. Airborne gamma survey of the historic Sleisbeck mine area in the Northern Territory, Australia, and its use for site rehabilitation planning. *Journal of Environmental Radioactivity* 99, 1770–1774.

Bollhöfer A & Fawcett M 2009. Results of a gamma dose rate survey at the rehabilitated Sleisbeck mine, Northern Territory, Australia. Internal Report 561, April, Supervising Scientist, Darwin. Unpublished paper.

Fisher WJ 1968. Mining practice in the South Alligator Valley. In: *Proceedings of a Symposium 'Uranium in Australia'*. AusIMM Rum Jungle Branch, 16–21 June 1968, AusIMM, Melbourne.

ICRP 2007. *The 2007 Recommendations of the International Commission on Radiological Protection*. Publication 103, International Commission on Radiological Protection, Elsevier Ltd.

Pfitzner K & Martin P 2000. Airborne gamma survey of the South Alligator River valley: First report. Internal Report 353, Supervising Scientist, Darwin. Unpublished paper.

Pfitzner K, Martin P & Ryan B 2001. Airborne gamma survey of the upper South Alligator River valley: Second Report. Internal Report 377, Supervising Scientist, Darwin. Unpublished paper

Ryan B, Bollhöfer A & Martin P 2008. Radionuclides and metals in freshwater mussels of the upper South Alligator River, Australia. *Journal of Environmental Radioactivity* 99, 509–526.

Supervising Scientist 2003. *Annual report 2002–2003*. Supervising Scientist, Darwin NT.

Tims S, Ryan B & Waggitt P 2000. γ Radiation survey of exposed tailings in the area around Rockhole mine. Internal report 332, Supervising Scientist for the Alligator Rivers Region, Darwin.

Turner K, Jones D & Humphrey C 2009. Changes in water quality of Rockhole Mine Creek associated with historic mining activities. Internal Report 560, June, Supervising Scientist, Darwin. Unpublished paper.

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

Waggitt P & Fawcett M 2008. Small scale uranium mine remediation in northern Australia. In *Uranium mining and hydrogeology V*, eds BJ Merkel & A Hasche-Berger, Springer-Verlag Berlin & Heidelberg New York, 243–250.

Research consultancies

This section contains a summary of non-uranium mining related research consultancies carried out by ***eriss*** during 2008–2009. Most of these reports are commercial-in-confidence and are not available for public release.

List of non-uranium mining related research consultancies

Humphrey C, Buckle D & Camilleri C 2009. A macroinvertebrate survey of stream sites associated with Territory Resources' Frances Creek iron ore project, April 2008. Commercial-in-Confidence Report for Earth Water Life Sciences Pty Ltd, March 2009.

Harford A, van Dam R & Hogan A 2009. Ecotoxicological Assessment of Seepage Water from the Savannah Nickel Mines. Commercial-in-Confidence Report for Panoramic Resources Ltd, April 2009.

Humphrey C, van Dam R, Storey A, Chandler L, Hogan A & Buckle D 2008. Assessment of the effects of $MgSO_4$-rich wastewater discharges from Argyle Diamond Mine on downstream aquatic ecosystems: Synthesis of a three year (2006–08) study. Commercial-in-Confidence Report for Argyle Diamonds Ltd, November 2008.

Ryan B & Bradley F 2009. Preliminary report into the characterisation of groundwater at the Rum Jungle mine site. Report prepared for Department of Resources, Energy and Tourism, February 2009, Supervising Scientist Division, Darwin NT.

van Dam R & Harford A 2009. Review of the revision of the nitrate water quality trigger value for fresh surface waters. Commercial-in-Confidence Report for Environment Canterbury, April 2009.

Ecotoxicological assessment of mine site seepage water

A Harford, R van Dam, A Hogan & A Storey[1]

An ecotoxicological assessment was undertaken on seepage waters from a mining project in north-west Western Australia, to address knowledge gaps regarding potential effects of mine seepage waters on downstream aquatic ecosystems. The study was done in conjunction with a field study by Wetland Research and Management of the aquatic biota of the creeks on and adjacent to the mine. The objectives of the ecotoxicological study were to:

1 assess the toxicity of seepage water to five native tropical freshwater species;

2 compare the ecotoxicological results to water quality downstream of SNM in order to predict the risks of the aquatic ecosystems being impacted; and

3 where possible, recommend site-specific water quality Trigger Values (TVs) for key toxicants or water quality parameters based on the laboratory ecotoxicological results or approaches detailed by ANZECC/ARMCANZ (2000).

The toxicity of the seepage water ranged from non-toxic to moderately toxic. Chemical analyses of the seepage water demonstrated that sulfate (SO_4) and magnesium (Mg) were the major constituents. Metals were below default ANZECC/ARMCANZ (2000) guideline TVs. A likely cause of observed toxicity was the combination of all the elevated major ions, resulting in an ion imbalance that led to osmotic stress in the organisms (ie a salinity effect).

As SO_4 was the clearly dominant ion in the seepage water, and hence a good indicator of seepage water salinity, a preliminary TV for SO_4 of 70 mg/L, based on the toxicity data, was proposed. Based on comparisons of the preliminary SO_4 TV (ie as an indicator of the seepage water salinity) with existing limited environmental monitoring data, the risk of seepage water having a significant impact on the downstream aquatic ecosystem appeared to be low. The results of the field study confirmed this conclusion. There were no significant adverse effects observed on fish, zooplankton or hyporheic-zone fauna, and detectable but minor effects on macroinvertebrate communities, at mine-exposed sites. Moreover, the data obtained from the field study showed an ecological response consistent with the laboratory-derived SO_4 TV of 70 mg/L for protection of more sensitive macroinvertebrate species.

In the event the mining project wished to identify the cause/s of toxicity of the seepage water, additional toxicity studies were recommended, using a 'synthetic' seepage water that simulates the major ion composition and concentrations in the seepage water. Various manipulations of the synthetic effluent, whereby ions are excluded individually (and, if necessary, in combination), would aid in identifying the key major ion/s contributing to toxicity.

References

ANZECC/ARMCANZ 2000. *Australian and New Zealand guidelines for fresh and marine water quality*. National Water Quality Management Strategy Paper No 4, Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

[1] Wetland Research & Management, Glen Forrest, WA 6071, Australia

TRaCK 4.1 – Flood inundation mapping for the Mitchell and Daly River catchments

R Bartolo, D Ward[1] & D Jones

TRaCK (Tropical Rivers and Coastal Knowledge) is a research hub under the Commonwealth Environmental Research Facilities (CERF) scheme, managed by the Department of Environment, Water, Heritage and the Arts. ***eriss*** is a project collaborator in Theme 4 (Material budgets- water, carbon, sediment and nutrients), Project 4.1 – catchment water budgets and water resource assessment. The project aims to start measuring and calculating the different elements of water budgets in three of the TRaCK focus catchments (Daly River – NT, Mitchell River – Queensland, and Fitzroy River – NT). The specific task ***eriss*** is contributing to Project 4.1 is flood inundation mapping for the Mitchell and Daly River catchments.

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. In the past year, historical rainfall data for the Mitchell and the Daly catchments and discharge data for the Mitchell was analysed and 2009 was identified as a suitable year for the mapping of maximum inundation limit for both the Mitchell and Daly catchments. For both the Mitchell and Daly catchments, optical remote sensing imagery (MODIS and Landsat) were captured during the 2009 wet season, coincident with ALOS PALSAR ScanSAR scenes.

Additional ALOS imagery covering the Daly River floodplain was acquired to compare PALSAR modes (ScanSAR, Polarimetric and Fine Beam Double imagery). Figure 1 shows basic visual imagery interpretation of a component of the ScanSAR data for the Daly River. This information will be used to produce inundation classifications using Geographic Object Based Image Analysis (GEOBIA) approaches.

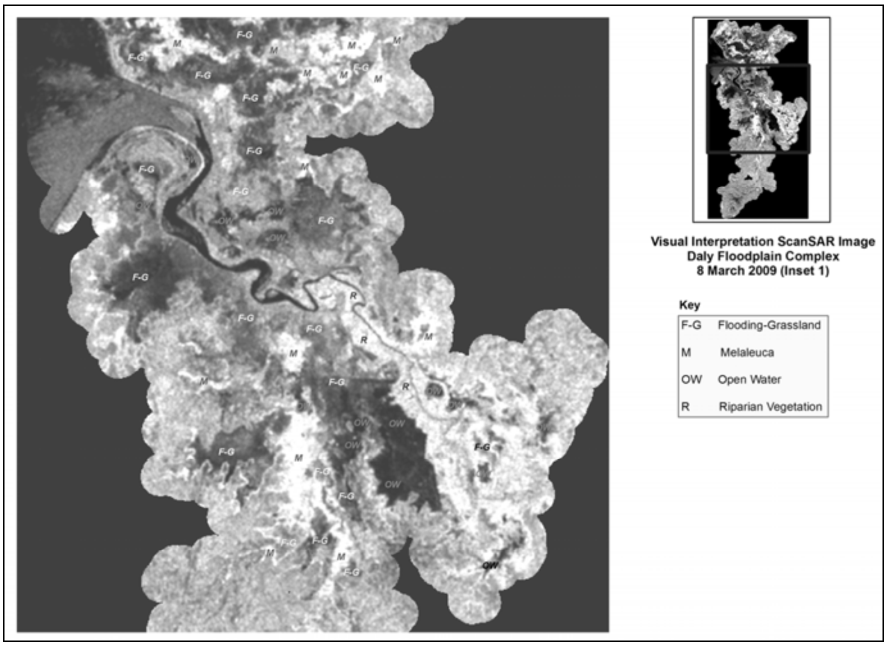


Figure 1 Visual interpretation of PALSAR (ScanSAR mode) image of a subset of the Daly River Floodplain (captured 8 March 2009) for the major cover types associated with inundation mapping

1 Australian Rivers Institute, Nathan Campus, Griffith University 170 Kessels Road, Nathan, QLD 4111 Australia

Appendix 1 SSD publications and presentations for 2008–09

Journal papers (in press or published)

Bollhöfer A, Pfitzner K, Ryan B, Martin P, Fawcett M & Jones DR 2008. Airborne gamma survey of the historic Sleisbeck mine area in the Northern Territory, Australia, and its use for site rehabilitation planning. *Journal of Environmental Radioactivity* 99, 1770–1774.

Cheng KL, Parry DL, Hogan AC, Markich SJ, Harford AJ & van Dam RA (in press). Uranium toxicity and speciation following chronic exposure to the tropical freshwater fish, *Mogurnda mogurnda*. *Aquatic Toxicology*. doi:10.1016/j.chemosphere.2010.02.017

Collins J, Hutley L, Williams R, Boggs G, Bell D & Bartolo R 2009. Estimating landscape-scale vegetation carbon stocks using airborne multi-frequency polarimetric synthetic aperture radar (SAR) in the savannahs of north Australia. *International Journal of Remote Sensing* 30, 1141–1159.

Finlayson CM, Eliott I & Eliott M 2009. A strategic framework for monitoring coastal change in Australia's wet-dry tropics: concepts and progress. *Geographical Research* June 2009 47(2), 109–123.

Hahn T, Stauber J, Dobson S, Howe P, Kielhorn J, Koennecker G, Diamond J, Lee-Steere C, Schneider U, Sugaya Y, Taylor K, van Dam R & Magelsdorf I 2009. Reducing uncertainty in ERA: Clearly defining acute and chronic toxicity tests. *Integrated Environmental Assessment and Management – Learned Discourse* 5(1), 175–177.

Hancock GR, Lowry JBC, Moliere DR & Evans KG 2008. An evaluation of an enhanced soil erosion and landscape evolution mode: a case study assessment of the former Nabarlek uranium mine, Northern Territory, Australia. *Earth Surface Processes and Landforms* 33(13), 2045–2063.

Hancock, GR, Lowry JBC, Coulthard TJ, Evans KG & Moliere DR 2009. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models. *Earth Surface Processes and Landforms.* (in press)

Hogan AC, van Dam RA, Houston MA, Harford AJ, Nou S 2009. Uranium exposure to the tropical duckweed, *Lemna aequinoctialis*, and pulmonate snail, *Amerianna cumingi*: fate and toxicity. *Archives of Environmental Contamination and Toxicology.* (in press)

Johansen K, Phinn S, Lowry J & Douglas M 2008. Quantifying indicators of riparian condition in Australian tropical savannas: integrating high spatial resolution imagery and field survey data. *International Journal of Remote Sensing* 29 (23), 7003–7028.

Lawrence CE, Akber RA, Bollhöfer A & Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Moliere DR & Evans KG (in press). Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research* DOI: 10.1111/j.1745-5871.2010.00641.x

Moliere DR, Lowry JBC & Humphrey CL 2009. Classifying the flow regime of data-limited streams in the wet-dry tropical region of Australia. *Journal of Hydrology* 367 (2009), 1–13.

Ryan B, Bollhöfer A & Martin P 2008. Radionuclides and metals in freshwater mussels of the upper South Alligator River, Australia. *Journal of Environmental Radioactivity* 99, 509–526.

van Dam RA, Harford AJ, Houston MA, Hogan AC & Negri AP 2008. Tropical marine toxicity testing in Australia: A review and recommendations. *Australasian Journal of Ecotoxicology* 14(2/3), 55–88.

van Dam RA, McCullough C, Hogan A, Houston M, Harford A & Humphrey C (in press). Aquatic toxicity of magnesium sulphate, and the influence of calcium, in very low ionic concentration water. *Environmental Toxicology & Chemistry*.

Wilson GDF, Humphrey CL, Colgan DJ, Gray KA & Johnson RN 2009. Monsoon-influenced speciation patterns in a species flock of Eophreatoicus Nicholls (Isopoda; Crustacea) *Molecular Phylogenetics and Evolution* 51(2), 349–364.

Warne M & van Dam R 2008. NOEC and LOEC data should no longer be generated or used. *Australasian Journal of Ecotoxicology* 14(1), 1–5.

Conference papers – published

Bartolo R 2008. Women in remote sensing and photogrammetry – results of the SSI women in spatial survey. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Bollhöfer A 2009. The importance of conducting an early baseline radiation study. In *Radiation in Mining and Exploration Workshop: Presentations.* Darwin NT 12 June 2009, Darwin NT, The Australian Institute of Mining and Metallurgy, Carlton South, Vic, CD.

Bollhöfer A, Martin P, Ryan B, Pfitzner K, Frostick A, Evans K & Jones D 2008. Radiological Assessment of the rehabilitated Nabarlek Uranium Mine, Northern Territory, Australia. In *Uranium Mining and Hydrogeology V conference of the Technische Universität Bergakademie Freiberg*. 14–18 September 2008, eds BJ Merkel & A Hasche-Berger, Freiberg, Saxony, Germany, 363–364.

Bollhöfer A, Pfitzner K, Martin P, Ryan B & Jones DR 2008, The use of remotely sensed radiometric data for the assessment of the radiological status of historic uranium mine sites in the Northern Territory, Australia. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Boyden J, Bartolo R, Bayliss P, Christophersen P, Lawson V, McGregor S & Kennett R 2008. Initial assessment of high-resolution remote sensing to map and monitor change in wetland vegetation on Boggy Plains, World Heritage Area, Kakadu National Park, Australia. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Bush M 2009. Assessment of new uranium mines under the *Environmental Protection and Biodiversity Conservation Act 1999*. In *AusIMM International Uranium Conference: Presentations.* 10–11 June 2009, Darwin NT, The Australian Institute of Mining and Metallurgy, Carlton South, Vic, CD.

Hancock GR, Lowry JBC, Coulthard TJ & Evans KG 2008. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference, Darwin NT*, 30 September – 2 October 2008. USB2.0

Harford AJ, Negri A, Wilson S, Hogan AC, Parry DP, Orr J, Houston M & van Dam RA 2008. The development of robust ecotoxicological protocols for tropical marine environments: Knowledge gaps, progress and future needs. Paper presented at Coast to Coast '08, 18–22 August 2008, Darwin. Department of Natural Resources, Environment & the Arts, Northern Territory Government. Available at https://www.coast2coast.org.au/presentation-files.html. Last accessed 13 October 2008.

Houston M, Ng J, Noller B, Markich SJ & van Dam R 2008. The influence of Suwannee River fulvic acid on the speciation and toxicity of uranium to Australian tropical freshwater species. In *From molecular understanding to innovative applications of humic substances. Proceedings of the 14th meeting of International Humic Substances Society*. Moscow, 13–19 September 2008, eds IV Perminova & NA Kulikova, 421–424.

Jones DR, Humphrey C, van Dam R, Harford A, Turner K & Bollhoefer A 2009. Integrated chemical, radiological and biological monitoring for an Australian uranium mine – a best practice case study. In *Proceedings International Mine Water Conference*, 19–23 October, Pretoria, South Africa, 95–104. ISBN 978-0-9802623-5-3.

Lowry JBC, Evans KG, Coulthard TJ, Hancock GR & Moliere DR 2009. Assessing the impact of extreme rainfall events on the geomorphic stability of a conceptual rehabilitated landform in the Northern Territory of Australia. In *Mine Closure 2009. Proceedings of the Fourth International Conference on Mine Closure*, 9–11 September 2009, Perth, Australia, eds A Fourie & M Tibbett, 203–212.

Lowry J, Hess L & Rosenqvist A 2008. Mapping and monitoring wetlands around the world using ALOS PALSAR: the ALOS Kyoto and Carbon Initiative wetland products. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Lowry J, Staben G & Saynor M 2008. Mapping the distribution of landslides in Arnhem Land using ALOS AVNIR imagery and object-based classification techniques. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Pfitzner K, Esparon A & Bollhöfer A 2008. SSD’s spectral library database. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Tayler K 2009. The regulation of uranium mining in the Northern Territory. In *Radiation in Mining and Exploration Workshop: Presentations.* Darwin NT 12 June 2009, Darwin NT, The Australian Institute of Mining and Metallurgy, Carlton South, Vic, CD.

Turner K & Tayler K 2009. Research to routine monitoring – best practice water quality monitoring. In *AusIMM International Uranium Conference: Presentations.* 10–11 June 2009, Darwin NT, The Australian Institute of Mining and Metallurgy, Carlton South, Vic, CD.

Presentations[1]

Akber R, Lu P & Bollhöfer A 2009. Uranium series disequilibrium in soils in areas used for water disposal through spray irrigation at ERA Ranger Uranium Mine. Paper presented at 2009 Australasian Radiation Protection Society (ARPS) Conference, Fremantle 26–28 October 2009.

Bartolo R 2008. Spatial information and climate change in a North Australia context. Paper presented at the Asia Pacific Spatial Innovation Conference, 18–19 November 2008, Canberra.

Bollhöfer A 2008. Die Nabarlek Uranmine. The Nabarlek Uranium mine – Studies of the Environmental Research Institute of the Supervising Scientist. Seminar held as part of a seminar series '*Current Topics in Radiation Protection'*, German Radiation Protection Agency. 18 September 2008, Berlin, Germany.

Bollhöfer A, Brazier J, Ryan B, Humphrey C & Esparon A 2008. A study of radium bioaccumulation in freshwater mussels, *Velesunio angasi*, in the Magela Creek catchment, Northern Territory, Australia. Paper presented at the 10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Bollhöfer A, Esparon A, Ryan B & Pfitzner K 2009. Investigating a pre-mining radiological analogue for Ranger Uranium mine, Northern Territory, Australia. Paper presented at 2009 Australasian Radiation Protection Society (ARPS) Conference, Fremantle 26–28 October 2009.

Brazier J, Ryan B, Humphrey C & Bollhöfer A 2008. Uranium bioaccumulation and lead isotope ratios in freshwater mussels downstream of Ranger uranium mine, Australia. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Buckle D, Humphrey C & Davies C 2009. *Fish community monitoring in tropical shallow billabongs and the influence of aquatic vegetation*. Paper presented at Australian Society for Limnology, Alice Springs Convention Centre, 28 September – 2nd October 2009.

Cheng K, Parry D, Hogan A & van Dam R 2008. Chronic toxicity of uranium to the tropical freshwater fish, Mogurnda mogurnda. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Cheng K, Parry D, Hogan A, Markich S & van Dam R 2009. Uranium speciation and toxicity following chronic exposure to the tropical freshwater fish, *Mogurnda mogurnda*. Paper presented at the 13th Australasian Society for Ecotoxicology Conference, 20–23 September 2009, University of Adelaide, Adelaide.

Esparon A & Pfitzner J 2008. Visual gamma – gamma spectrometry analysis software. Paper presented at the 10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Esparon A, Pfitzner K, Bollhöfer A & Ryan B 2008. Determination of an analogue site for Ranger uranium mine to extrapolate pre-mining gamma dose rates. Paper presented at the

[1] Presentations to conferences and symposia that have been externally published in 2008–2009 are included in section 'Conference papers – published'.

10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Frostick A, Bollhöfer A & Parry D 2008. Investigating potential natural analogues for Ranger Uranium Mine. Paper presented at the 10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Frostick A, Bollhöfer A, Parry D, Munksgaard N & Evans K 2008. Spatiotemporal assessment of sediments from Magela Creek, Northern Australia to evaluate the impacts of uranium mining. Paper presented at the 10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Hahn T, Stauber J, Dobson S, Howe P, Kielhorn J, Koennecker G, Diamond J, Lee-Steere C, Schneider U, Sugaya Y, Taylor K & van Dam R 2008. Sources of variation in environmental hazard assessment of chemicals in aquatic systems: an international analysis. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Harford AJ, Cheng KL, Costello CE, Hogan AC & van Dam RA 2008. Ecotoxicological assessment of flocculant blocks and their individual constituents. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Harford A, Saynor M, Hogan A, Cheng K, White D & van Dam R 2009. The development of water quality guidelines for suspended sediments in Magela Creek. Paper presented at the 13th Australasian Society for Ecotoxicology Conference, 20–23 September 2009, University of Adelaide, Adelaide.

Hogan A, van Dam R, Harford A, Cheng K & Turner K 2009. Effects of Mg pulse exposures on tropical freshwater Australian species. Paper presented at the 13th Australasian Society for Ecotoxicology Conference, 20–23 September 2009, University of Adelaide, Adelaide.

Houston M, Ng J, Noller B, Markich SJ & van Dam R. 2008a. The influence of dissolved organic carbon (DOC) on the speciation & toxicity of uranium to Australian tropical freshwater species. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Houston M, Ng J, Noller B, Markich S & van Dam R 2009. Amelioration of uranium toxicity by dissolved organic carbon from a tropical Australian billabong. Paper presented at the 13th Australasian Society for Ecotoxicology Conference, 20–23 September 2009, University of Adelaide, Adelaide.

Hughes A 2008. Regulation of uranium mining in the Northern Territory, Australia. Presentation to an International Atomic Energy Agency technical meeting on Implementation of sustainable global best practices in uranium mining and processing, 15–17 October 2008, Vienna, Austria.

Humphrey C & Chandler L 2009. Taxonomic resolution for biological assessment of mining impacts in tropical streams of the Northern Territory. Paper presented at the Combined Australian Entomological Society's 40th AGM & Scientific Conference and Society of Australian Systematic Biologists and 9th Invertebrate Biodiversity & Conservation Conference, 25–28 September 2009, Darwin.

Humphrey C, Davies C, Buckle D & McGuinness K 2009. *Development of cost-effective, 'early warning' techniques to monitor and assess changes in water quality*. Paper presented at Australian Society for Limnology, Alice Springs Convention Centre, 28 September – 2nd October 2009.

Humphrey C & McGuinness K 2008. Experimental design considerations for monitoring and assessment of mining impacts in tropical seasonally-flowing streams. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Humphrey C, McGuinness K & Douglas M 2009. *Experimental design considerations for monitoring of macroinvertebrate communities in tropical seasonally-flowing streams*. Paper presented at Australian Society for Limnology, Alice Springs Convention Centre, 28 September – 2nd October 2009.

Jones D 2008. Effects of extreme events in the Kakadu region. Paper presented at Kakadu Landscape Management Symposia Series: Climate Change Workshop. 6–7 August 2008, Jabiru.

Jones D 2009. Key issues for 'best practice' regulation of uranium mining. Paper presented at Securing the Future (Mining, Metals and the Environment in a Sustainable Society) and 8th International Conference on Acid Rock Drainage, 23–26 June 2009, Skelleftea, Sweden.

Jones D, Humphrey C, van Dam R, Harford A, Turner K & Bollhöfer A 2009. Integrated chemical, radiological and biological monitoring for an Australian uranium mine – a best practice case study. Paper presented at International Minewater Conference, Pretoria, October 2009.

Lu P, Akber R & Bollhöfer A 2009. Challenges in estimating public radiation dose resulting from land application of waters of elevated natural radioactivity at Ranger Uranium Mine, Australia. Paper presented at the International IAEA Conference on Remediation of Land Contaminated by Radioactive Material Residues, 18–22 May 2009, Astana, Kazakhstan.

Medley P, Bollhöfer A, Ryan B & Sellwood J 2008. Derivation of regional concentration factors for radium in bush passionfruit (*Passiflora foetida*) from the Alligator Rivers Region, Northern Territory, Australia. Paper presented at the 10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Ryan B, Bollhöfer A & Martin P 2008. A radiation dose assessment for Aboriginal inhabitants downstream of Ranger Uranium Mine in the Northern Territory of Australia. Paper presented at the 10th South Pacific Environmental Radioactivity Conference, SPERA 2008, 24–27 November 2008, Christchurch, New Zealand.

Staben G, Saynor M, Lowry J 2009. Mapping landslides in northern Australia using object based classification techniques. Paper presented at Surveying and Spatial Sciences Institute Biennial International Conference, 28 September – 2 October 2009, Adelaide Convention Centre, Adelaide, Australia.

Turner K, Humphrey C, Brazier J, Jones D 2008. Sediment influences on the derivation of mine closure water quality criteria in a tropical waterbody. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

van Dam R, Hogan A, Harford A & Markich S 2008. Toxicity, metal speciation and risk characterisation of waste-water from a legacy gold mine in northern Australia. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

van Dam R, Humphrey C, Storey A, Samaraweera S, Hogan A, Buckle D & Chandler L 2008. Assessment of the effects of $MgSO_4$-rich wastewater discharges from Argyle Diamond Mine on downstream aquatic ecosystems. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

van Dam R, Negri A, Harford A, Hogan A, Adams M, Stauber J, Parry D & Orr J 2008. Using the tropical species *Nitzschia closterium* & *Acropora tenuis* to assess site-specific issues of a tropical marine discharge. Paper presented at 5th SETAC World Congress, 3–7 August 2008, Sydney Convention and Exhibition Centre, Sydney. Society of Environmental Toxicology and Chemistry.

Warne M StJ & van Dam R 2009. 101 reasons to stop generating and using hypothesis based toxicity estimates – NOECs and LOECs should be banned. Paper presented at the 13th Australasian Society for Ecotoxicology Conference, 20–23 September 2009, University of Adelaide, Adelaide.

Supervising Scientist Reports

Jones DR & Webb A (eds) 2009. ***eriss*** *research summary 2007–2008*. Supervising Scientist Report 200, Supervising Scientist, Darwin NT.

de Groot R, Finlayson M, Verschuuren B, Ypma O & Zylstra M 2008. *Integrated assessment of wetland services and values as a tool to analyse policy trade-offs and management options: A case study in the Daly and Mary River catchments, northern Australia*. Supervising Scientist Report 198, Supervising Scientist, Darwin NT.

Internal Reports

ARRAC 2008. Alligator Rivers Region Advisory Committee 27th Meeting, April 2007, Darwin, Meeting papers. Internal Report 540, August, Supervising Scientist, Darwin. Unpublished paper.

ARRAC 2008. Alligator Rivers Region Advisory Committee 28th Meeting, August 2007, Darwin, Meeting papers. Internal Report 541, August, Supervising Scientist, Darwin. Unpublished paper.

Alligator Rivers Region Advisory Committee 2009. Alligator Rivers Region Advisory Committee 29th Meeting, April 2008, Darwin, Meeting papers. Internal Report 542, June, Supervising Scientist, Darwin. Unpublished paper.

Bollhöfer A & Fawcett M 2009. Results of a gamma dose rate survey at the rehabilitated Sleisbeck mine, Northern Territory, Australia. Internal Report 561, April, Supervising Scientist, Darwin. Unpublished paper.

Bollhöfer A, Fawcett M, Staben G, Sellwood J, Fox G, Ryan B & Pfitzner K 2009. Radiation surveys of the historic Palette and El Sherana mines, South Alligator River Valley, Australia. Internal Report 556, March, Supervising Scientist, Darwin. Unpublished paper.

Brazier J & Humphrey C 2009. Ranger stream monitoring program: relocation of surface water chemistry grab monitoring sites in Magela Creek. Internal Report 563, June, Supervising Scientist, Darwin. Unpublished Paper.

Buckle D, Storey A, Humphrey C & Chandler L 2010. Fish and macroinvertebrate assemblages of the upper Ord River catchment. Internal Report 559, January, Supervising Scientist, Darwin.

Esparon A 2008. Jabiru Field Station radiation dose and surface contamination survey. Internal Report 538, December, Supervising Scientist, Darwin. Unpublished paper.

Hogan A, Houston M, Nou S, Harford A & van Dam 2008. Chronic toxicity of uranium to the tropical duckweed *Lemna aequinioctialis* and the gastropod *Amerianna cumingi*. Internal Report 549, November, Supervising Scientist, Darwin. Unpublished paper.

Jones DR (ed) 2009. ***eriss*** communication and planning workshop – 08/09 workplan and proposed 09/10 directions. Internal Report 550, June, Supervising Scientist, Darwin. Unpublished paper.

Madon E 2009. Potential impacts of ammonia on the natural environment from the land application of effluent water at the Ranger mine. Internal Report 562, November, Supervising Scientist, Darwin. Unpublished Paper.

Medley P 2009. A review of chemical storage and handling protocols in the Environmental Radioactivity laboratories of ***eriss***. Internal Report 548, September, Supervising Scientist, Darwin. Unpublished Paper

Office of the Supervising Scientist, Supervision & Assessment Unit 2009. Ranger mine Routine Periodic Inspections 2002–2004. Internal Report 551, June, Supervising Scientist, Darwin. Unpublished paper.

Office of the Supervising Scientist, Supervision & Assessment Unit 2009. Ranger mine Routine Periodic Inspections 2006. Internal Report 553, June, Supervising Scientist, Darwin. Unpublished paper.

Office of the Supervising Scientist, Supervision & Assessment Unit 2009. Ranger mine Routine Periodic Inspections 2005. Internal Report 552, June, Supervising Scientist, Darwin. Unpublished paper.

Office of the Supervising Scientist, Supervision & Assessment Unit 2009. Ranger mine Routine Periodic Inspections 2007. Internal Report 554, June, Supervising Scientist, Darwin. Unpublished paper.

Office of the Supervising Scientist, Supervision & Assessment Unit 2009. Ranger mine Routine Periodic Inspections 2008. Internal Report 555, June, Supervising Scientist, Darwin. Unpublished paper.

Pfitzner K & Bollhöfer A 2008. Status of the vegetation plots for the spectral library project. Internal Report 546, December, Supervising Scientist, Darwin. Unpublished paper.

Saynor MJ, Houghton R, Hancock G, Staben G, Smith B & Lee N 2009. Soil sample descriptions – Gulungul Creek, Ranger mine site and Nabarlek: Cyclone Monica fieldwork. Internal Report 558, Supervising Scientist, Darwin. Unpublished paper.

Saynor MJ, Staben G, Hancock G, Fox G, Calvert G, Smith B, Moliere DR & Evans KG 2009. Impact of Cyclone Monica on catchments within the Alligator Rivers Region – Data. Internal Report 557, Supervising Scientist, Darwin. Unpublished paper.

Supervising Scientist Division 2008. Consolidated list of publications, reports and conference presentations by staff of and consultants to the Supervising Scientist 1978–30 June 2008. Internal Report 547, August, Supervising Scientist, Darwin. Unpublished paper.

Turner K, Jones D & Humphrey C 2009. Changes in water quality of Rockhole Mine Creek associated with historic mining activities. Internal report 560, June, Supervising Scientist, Darwin. Unpublished paper.

Consultancy reports

Humphrey C, Buckle D & Camilleri C 2009. A macroinvertebrate survey of stream sites associated with Territory Resources' Frances Creek iron ore project, April 2008. Commercial-in-Confidence Report for Earth Water Life Sciences Pty Ltd, March 2009.

Harford A, van Dam R & Hogan A 2009. Ecotoxicological Assessment of Seepage Water from the Savannah Nickel Mines. Commercial-in-Confidence Report for Panoramic Resources Ltd, April 2009.

Humphrey C, van Dam R, Storey A, Chandler L, Hogan A & Buckle D 2008. Assessment of the effects f $MgSO_4$-rich wastewater discharges from Argyle Diamond Mine on downstream aquatic ecosystems: Synthesis of a three year (2006-08) study. Commercial-in-Confidence Report for Argyle Diamonds Ltd, November 2008.

Ryan B & Bradley F 2009. Preliminary report into the characterisation of groundwater at the Rum Jungle mine site. Report prepared for Department of Resources, Energy and Tourism, February 2009, Supervising Scientist Division, Darwin NT.

van Dam R & Harford A 2009. Review of the revision of the nitrate water quality trigger value for fresh surface waters. Commercial-in-Confidence Report for Environment Canterbury, April 2009.

Other

Bartolo R, Bayliss P & van Dam R 2008. *Ecological risk assessment for Australia's northern tropical rivers. Sub-project 2 of Australia's Tropical Rivers – an integrated data assessment and analysis (DET18)*. A report to Land & Water Australia. Environmental Research Institute of the Supervising Scientist, National Centre for Tropical Wetland Research, Darwin NT. www.environment.gov.au/ssd/tropical-rivers/triap-sp2.html.

Further information

SSD publications on the web

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

SSD annual reports

http://www.environment.gov.au/ssd/about/corporatedocs.html

Supervising Scientist Report series

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

Consolidated list of publications, reports and conference presentations by staff of and consultants to the Supervising Scientist 1978 – 30 June 2008

http://www.environment.gov.au/ssd/publications/ir/547.html

Supervising Scientist Division brochure

http://www.environment.gov.au/ssd/about/brochure.html

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:

- an independent Chairperson;
- seven independent scientific members 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; and
- six members representing key stakeholder organisations.

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 the Minister for the Environment, Heritage and the Arts] 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 (current) 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 mine site. 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 mine site. 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 mine site 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 mine site 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 mine sites 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.