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eriss research summary 2011–2012

Editors RA van Dam, A Webb & SM Parker

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Note: Authors were Supervising Scientist staff at time of research and/or write-up unless otherwise stated.

Preface

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

eriss also applies its expertise to conducting research into the sustainable use and environmental protection of tropical rivers and their associated wetlands, and to undertaking 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 are defined by Key Knowledge Needs (KKNs) developed by consultation between the Alligator Rivers Region Technical Committee (see ARRTC membership and function in Appendix 2), the Supervising Scientist, Energy Resources of Australia Ltd (ERA) and other stakeholders. The KKNs are subject to ongoing review by ARRTC 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.

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

This report documents the monitoring and research projects undertaken by ***eriss*** over the 2011–12 financial year (1.7.11 to 30.6.12). The report is structured according to the five major topic areas in the KKN framework, noting that this year there are no papers for Jabiluka or Nabarlek.

1 Ranger – current operations
2 Ranger – rehabilitation
3 Jabiluka
4 Nabarlek
5 General Alligators Rivers Region

eriss continued its chemical and biological off-site monitoring programs for assessing impacts from the current operations at Ranger. Research associated with current operations continued its focus on water quality issues. The discrepancies in snail egg production between the downstream site relative to the upstream site for the Magela and Gulungul creek in situ toxicity monitoring programs was further studied. Snail egg production was found to be related to water temperature and electrical conductivity (EC) in a complex, interacting relationship. However, the physiological mechanism underpinning these observations is not yet understood. Other research of note included the derivation of annual solute load budgets for Gulungul Creek upstream and downstream of Ranger using the continuous monitoring data obtained over the past seven wet seasons (2005–06 and 2011–12), and the commencement of an assessment of the toxicity of manganese to local freshwater species.

Research related to rehabilitation at Ranger is a key focus at ***eriss***. The trial landform studies continued in 2011–12, with a focus on the monitoring of erosion products and water chemistry in runoff from two of the four erosion plots. The bedload being exported from these plots continues to decline linearly with time, with the 2011–12 wet season data showing, for the first time, the effects of the developing vegetation on reducing the rate of erosion. Radon exhalation flux densities from the trial landform are also being assessed. Research into the development of closure criteria continues to be a focus area, particularly in relation to billabong water quality and terrestrial vegetation.

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

Significant 2012–13 work planning activities were undertaken during 2011–12, as a result of ERA announcing an increased focus on determining closure timelines and associated knowledge needs. The outcomes of this process will be reflected in the program of research and monitoring presented in the 2012–13 Annual Research Summary.

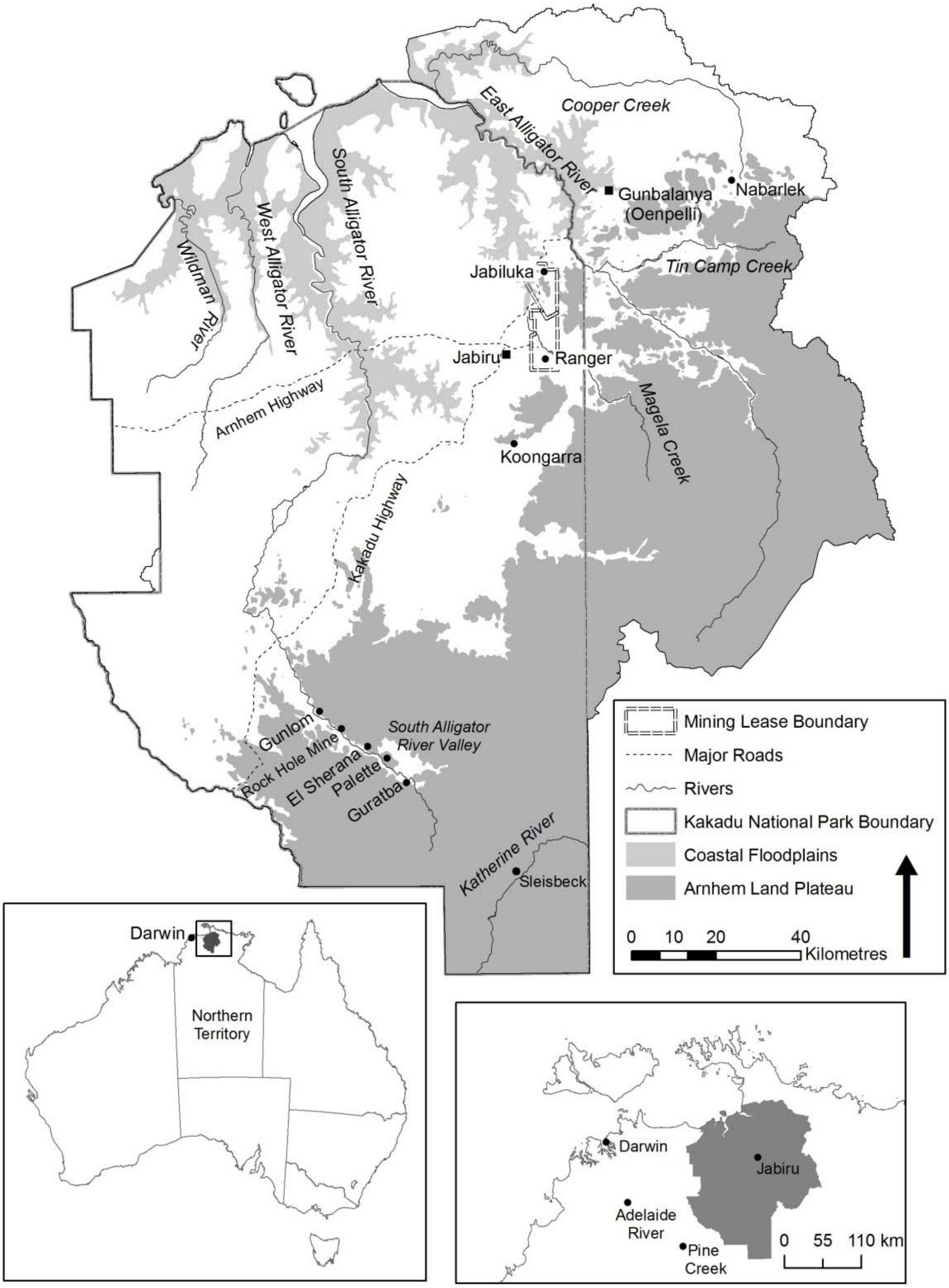
Key non-uranium mining related external activities for 2011–12 centred around two large government-funded programs: (i) the National Environmental Research Program (NERP) Northern Australia Hub, and (ii) the current revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Involvement in the NERP Hub focused on the acquisition and analysis of LiDAR data for the Alligator Rivers Region floodplains, for which the details are provided in a related core research project on the use of LiDAR data (see 'LiDAR capture for the Alligator Rivers Region, KKN 2.2.1 Land design, p116). Details of the involvement in the Water Quality Guidelines revision are provided in the Supervising Scientist 2011–12 Annual Report.

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

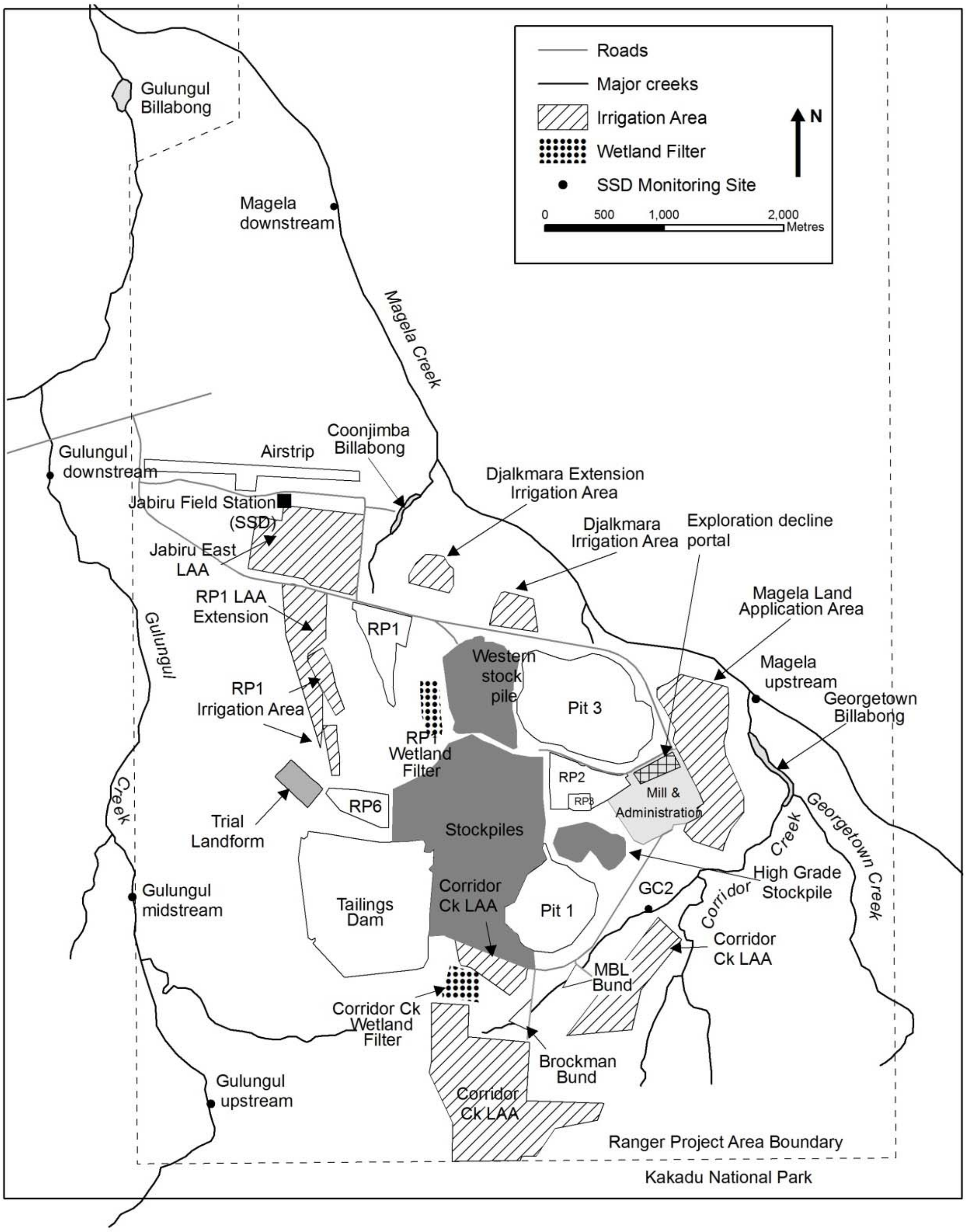
For additional information, readers are referred to the annual publications list (Appendix 1) that details all of the material published, and conference and workshop papers presented, by ***eriss*** staff in 2011–12.

Finally, I would like to take the opportunity to acknowledge the former Director of ***eriss***, Dr David Jones, for his tireless and highly committed efforts at guiding the Institute's research program and improving its business systems during his tenure.

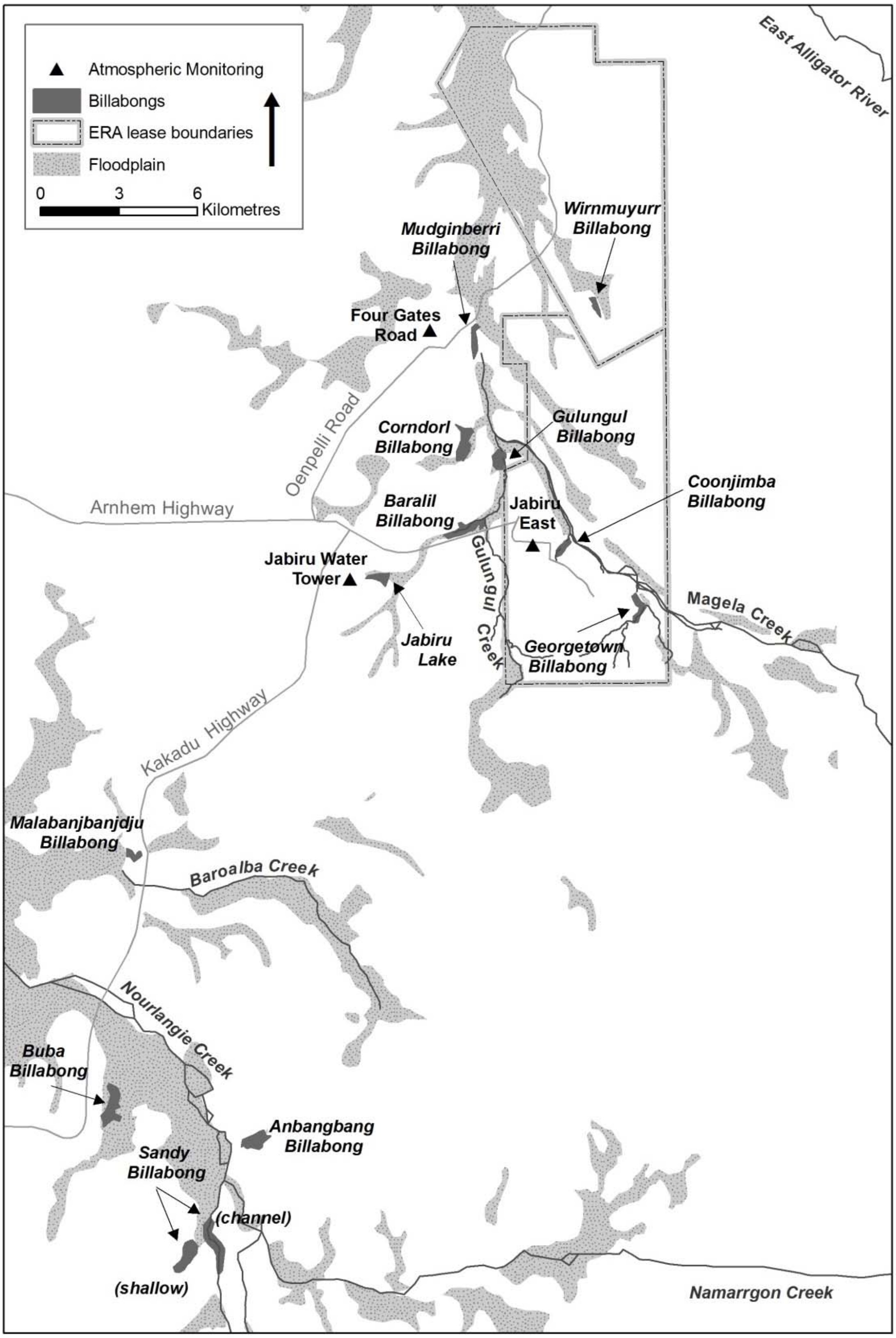
Dr RA van Dam, *Acting Director,*
Environmental Research Institute of the Supervising Scientist



Map 1 Alligator Rivers Region



Map 2 Ranger minesite



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

Part 1: Ranger – current operations

Conceptual models of stressor pathways for the operational phase

RE Bartolo, RA van Dam, SM Parker, K Kai-Nielsen, A Bollhöfer, WD Erskine & C Humphrey

Introduction

Conceptual models of potential stressor pathways associated with uranium mining in the Alligator Rivers Region (ARR) have been developed as part of the evolving ecological risk assessment framework that was started by the Supervising Scientist in the early 1980s. In response to recommendations by the World Heritage Commission Independent Scientific Panel and the Alligator River Region Technical Committee (ARRTC), a specific project was initiated to produce a comprehensive conceptual model of stressor pathways associated with the operational phase of the Ranger Uranium Mine (RUM).

Development of a new conceptual model of stressor pathways associated with the operational mining phase was commenced in 2004 (van Dam et al 2004). The primary purpose of the conceptual model was to place off-site environmental impact associated with the operational phase of mining into a risk management context. Sub-models for the multiple stressor pathways identified in the conceptual model were not finalised at that time. This project presents sub-models and supporting narratives for each of the stressor pathways identified in the conceptual model. An assessment of the relative importance of each pathway in terms of its potential to cause adverse environmental effects to the off-site environment of the ARR was conducted using an expert panel approach.

The main objective of the project was to assess ***eriss***'s knowledge base in relation to stressor pathways applicable to the operational phase at RUM and, in doing so, produce a gap analysis which can inform future Key Knowledge Need (KKN) reviews. A secondary objective of this work was to provide a screening level assessment of the importance of stressor pathways. This will support the development of a risk-based framework which could be used to prioritise future ***eriss*** mine-related research and monitoring.

Approach

Narratives were developed for each of the stressor pathway sub-models which describe the key model elements and provide supporting scientific knowledge and other information relevant to the pathway.

While each of the pathway sub-models and their related narrative are able to be read as stand-alone documents, collectively they form the current version of the conceptual model of stressor pathways for the operational mining phase at RUM.

The narratives are based on the following structure:

- a ***pathway*** section which describes the stressor pathway the narrative relates to;
- a ***relevant risk mitigation measures*** section outlining the measures in place to reduce potential risks to receptors in the ARR environment associated with the stressor pathway;

- a number of ***headline statements*** for each sub-model indicating the *relative importance* of the pathway (see below), the level of *scientific certainty* attached to this knowledge based on current research and monitoring (see below), the current *level of adverse ecological impact* resulting from the pathway based on existing monitoring data (see below), and the various ARRTC KKNs which relate to the pathway;
- a ***key stressors and their sources*** section that identifies the main mine-derived stressors involved in the pathway, their sources on the mine lease and the key transport mechanisms involved;
- an ***environmental compartments*** section that identifies the main environmental compartments/exposure media in which stressors may be distributed once they leave the mine site;
- a ***receptors*** section that identifies the main ecological receptors that may be exposed to the stressors as they move through the environmental compartment;
- a ***measurement endpoints*** section that identifies which measurement endpoints relate to each receptor involved and whether they provide a direct or indirect measure of impact on these receptors;
- a ***current level of scientific knowledge and certainty*** section which outlines current scientific knowledge regarding the pathway such as the characteristics of stressors or their sources, the loads of stressors being transported off the mine site via the pathway and the potential environmental effects of this, relevant knowledge gaps and the scientific research and monitoring being undertaken to address these.

Each stressor pathway was assessed in terms of its potential to result in adverse impacts on the environment in the ARR during the operational mining phase based on:

a) the size/potential maximum generating capacity of the relevant stressor source (low, medium or high); and

b) the potential maximum capacity (load and rate) of the relevant pathway to transport stressors from the mine site to the surrounding environment (low, medium or high).

Based on the scores assigned in relation to (a) and (b) above, a total importance score was calculated for each stressor pathway using a standard 3 × 3 scoring matrix (Table 1).

The current level of scientific certainty based on research and monitoring (*high*, *medium* or *low*) and the current level of adverse ecological impact on receptors based on monitoring (*yes*, *no* or *unknown*) associated with each stressor pathway were also determined.

Table 1 Scoring matrix for assessment of relative importance of stressor pathways

		Maximum size/generating capacity of source		
		Low	Medium	High
Max capacity of pathway	Low	Low	Low	Medium
	Medium	Low	Medium	Medium
	High	Medium	Medium	High

Stressor pathways and sub-models

It is important to provide some context to the narratives for the stressor pathways and sub-models:

Assessment endpoint: The assessment endpoint for each pathway is *protection of the environment in the ARR from the effects of uranium mining operations*. This assessment endpoint has been directly derived from the functions of the Supervising Scientist in relation to uranium mining in the ARR, as specified in the *Environment Protection (Alligator Rivers Region) Act 1978*.

Assessment of importance and certainty: The outcomes of the assessment of importance and certainty of the stressor pathways are shown in Table 2. The results of the assessment of relative importance are generally consistent with current scientific knowledge and assumptions regarding the potential risks to the environment associated with the various stressor pathways.

Three of the five pathways assessed as being of high importance relate to the transport of stressors via the surface water to surface water pathway, which would be expected given that the surrounding surface water systems are the primary receptors of water and hence waterborne stressors transported from the mine site. The majority of the stressor pathways (20 out of 31) were assessed as being of medium importance during the operational mining phase.

Relative importance of the pathways was determined based on the 'unmitigated' potential of pathway processes to transport stressors from the mine site into the surrounding environment. However, this does not mean that high importance pathways are resulting in, or are likely to result in, impact on receptors within the ARR. The actual loads and/or concentrations of stressors transported by these pathways at any time (and therefore the level of potential risk to receptors) vary depending on a range of chemical, biological, physical, radiological and other factors. Moreover, the main factor that determines the actual level of environmental risk associated with these stressor pathways is the various mitigation measures implemented by ERA. These measures are designed to reduce risks to the environment to acceptable levels (accepting that all mining involves some level of impact on the environment) either by containing stressors on the mine site or minimising the loads and concentrations of stressors that may be transported via the various pathways.

Given this, details of the risk mitigation measures relevant to each stressor pathway are highly relevant and, as such, are included in the model narratives.

Maps depicting each pathway have been developed. For example, the surface water to surface water pathway involves the potential transport of stressors from the mine lease to the surrounding environment via active (controlled) and passive releases of water from the mine site direct to Magela, Gulungul, Coonjimba, or Corridor/Georgetown creeks or via their tributaries (see Figure 1). Conceptual models for each pathway and stressor have also been developed. The model for the potential transport of inorganic toxicants through the surface water to surface water pathway is shown by Figure 2.

Table 2 Assessment of relative importance of stressor pathways based on size/maximum of sources and maximum capacity of pathway

Pathway	Size/max generating capacity of source (H, M, L)	Max capacity of pathway (H, M, L)	Scientific Certainty (H, M, L)	Impact No = N Yes = Y Unknown = U	Importance in operational phase
Inorganic toxicants surface water to surface water pathway	H	H	H	N	**High**
Inorganic toxicants airborne emissions pathway	H	H	H	U	**High**
Radionuclides surface water to surface water pathway	H	H	H	N	**High**
Radon-222 and radon decay products pathway	H	H	H	N – Human U – Biota	**High**
Transported sediments surface water to surface water pathway	H	H	H	N	**High**
Inorganic toxicants surface water to groundwater pathway	H	L	M	N	**Medium**
Inorganic toxicants LAA infiltration and runoff pathway	M	H	H	N	**Medium**
Inorganic toxicants stormwater runoff from non-mine site areas of lease pathway	L	H	H	N	**Medium**
Inorganic toxicants human and non-human vectors pathway	H	M	H	N	**Medium**
Inorganic toxicants bioaccumulation and trophic transfer pathway	H	L	M	N	**Medium**
Inorganic toxicants airborne dust & other particulates pathway	H	M	M	U	**Medium**
Organic toxicants surface water to surface water pathway	*M*	*H*	*L*	*U*	***Medium***
Organic toxicants stormwater runoff from non-mine site areas of lease pathway	*L*	*H*	*M*	*U*	***Medium***
Organic toxicants airborne dust & other particulates pathway	*M*	*M*	*M*	*U*	***Medium***
Organic toxicants airborne emissions pathway	*M*	*H*	*M*	*U*	***Medium***
Radionuclides surface water to groundwater pathway	H	L	M	N	**Medium**
Radionuclides stormwater runoff from non-mine site areas of lease pathway	L	H	M	N	**Medium**
Radionuclides human and non-human vectors pathway	H	M	H	N – Human U – Biota	**Medium**
Radionuclides bioaccumulation and trophic transfer pathway	H	L	H	N – Aquatic U – Terrestrial	**Medium**
Radionuclides airborne dust & other particulates pathway	H	M	H	N	**Medium**
Transported sediments stormwater runoff from non-mine site areas of lease pathway	L	H	H	N	**Medium**
Weed propagules surface water to surface water pathway	M	H	L	U	**Medium**
Weed propagules stormwater runoff from non-mine site areas of lease pathway	L	H	L	U	**Medium**
Weed propagules human and non-human vectors pathway	M	M	L	U	**Medium**
Weed propagules airborne dust & other particulates pathway	M	M	L	U	**Medium**
Organic toxicants surface water to groundwater pathway	*M*	*L*	*M*	*U*	***Low***
Organic toxicants LAA infiltration and runoff pathway	*L*	*L*	*M*	*U*	***Low***
Organic toxicants human and non-human vectors pathway	*L*	*M*	*L*	*U*	***Low***
Organic toxicants bioaccumulation and trophic transfer pathway	*L*	*L*	*L*	*U*	***Low***
Radionuclides airborne emissions pathway	L	H	H	N	***Low***
Radionuclides LAA infiltration and runoff pathway	M	L	H	N	**Low**

Figure 1 The surface water to surface water pathway

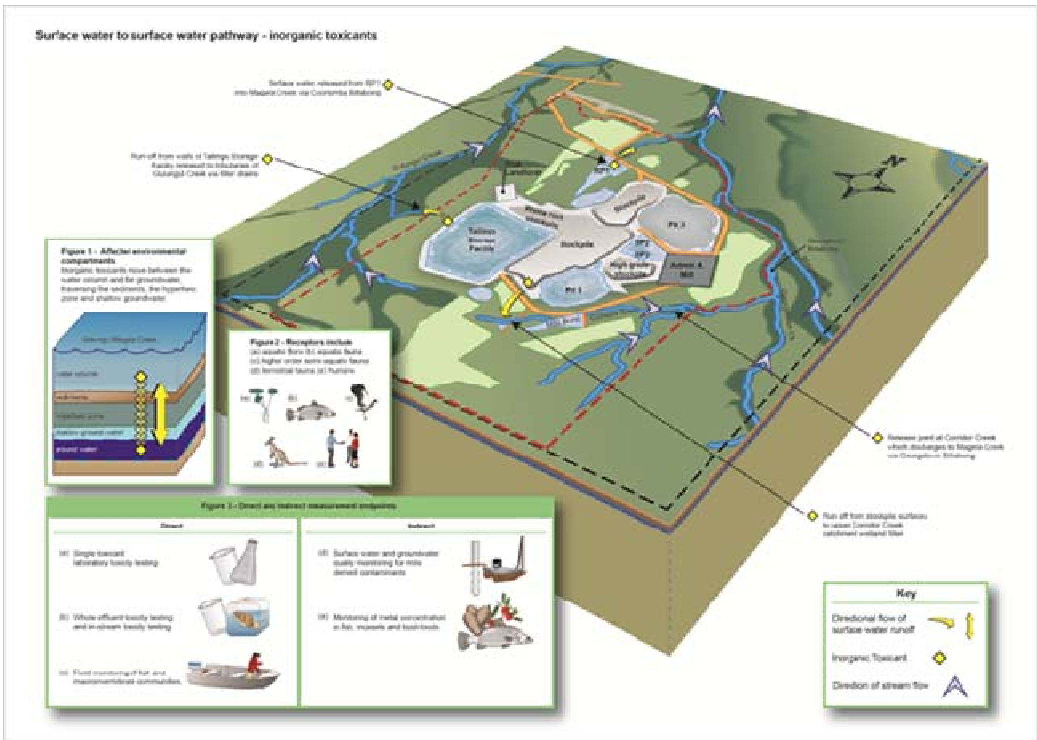


Figure 2 Sub-model for the transport of inorganic toxicants via the surface water to surface water pathway

Discussion

As a result of this project, a number of existing and new/emerging knowledge needs in relation to various stressor pathways associated with the operational mining phase at RUM were identified. Most of the existing knowledge needs have been identified as a result of recent research and monitoring, and are included in ARRTC's 2008–2010 KKN document.

A number of new and emerging knowledge needs were also identified as part of this work (see Tables 3 & 4). For example, there is currently limited knowledge regarding the extent to which various organic toxicants on the mine site, particularly hydrocarbons and volatile organic compounds, may be being transported off the mine site by surface water and groundwater pathways, and also the potential effects of these toxicants on biological receptors. The level of organic toxicants potentially being transported from the mine site is not currently measured by the Supervising Scientist. Areas of significant organic contamination (mainly hydrocarbons) are known to exist on the mine site (Hollingsworth 2006). While issues such as soil contamination on the mine site are generally seen as being more relevant to the mine rehabilitation phase, there could be potential for organic toxicants from these areas to be transported off the mine site by various pathways during the operational mining phase, and this may warrant further investigation.

A further emerging knowledge need identified during this work relates to the measurement of potential impacts of radiological stressors on non-human biota within the ARR. While previous work has been done by the Supervising Scientist in relation to the bioaccumulation of radionuclides in mussels and a limited number of other fauna, this work has mainly focused on potential dose effects on human receptors. Historically, radiation protection issues in the uranium mining sector have been focused on impacts to mine workers on the mine site and relevant off-site human receptors. However, the issue of potential radiological impacts on the environment in general is receiving increased attention at the international level mainly resulting from policy and technical guidance work by the International Commission on Radiological Protection (ICRP 2003, 2009). The need to better understand the potential impacts on biota associated with mine derived radiological stressors within the ARR has been recognised by the Supervising Scientist and ARRTC. As a result, the Supervising Scientist and the Australian Radiation Protection and Nuclear Safety Agency (ARPANSA) are jointly examining implications of this issue in relation to uranium mining in the ARR.

A number of new knowledge needs were also identified in relation to the transport of weed propagules via surface water, atmospheric and human and non-human pathways. Weeds are a key issue for the operational mining phase as well as the mine rehabilitation and post-mining phases. The mine operator undertakes regular chemical control measures to limit the spread of weeds on the mine lease as well as annual surveys of the distribution and density of a number of priority weed species on the mine lease. However, currently there are no quantitative data on the level of weed propagules being transported off the mine lease by the various transport pathways. There are also limited data available on the levels of weed propagules being transported onto the mine lease from the surrounding environment. The ongoing management of weeds on the mine lease is an important issue for both the operational and post-mining phases, in particular the establishment of the final landform and associated ecosystem development.

It is anticipated that these existing and new knowledge needs will be assessed as part of the review and revision of the ARRTC knowledge needs scheduled for 2012–13.

Table 3 Gaps in knowledge about the importance of stressor-pathway combinations. '✖' and '✖✖' denote knowledge gaps associated with moderate uncertainty and high uncertainty in knowledge, respectively (summarised from Table 1).

	Stressors					
Pathways	**Inorganic toxicants**	**Organic toxicants**	**Radionuclides**	**Radon-222 and decay products or progeny**	**Transported sediments**	**Weed propagules**
Surface water to surface water		✖✖				✖✖
Airborne emissions		✖				
Stormwater runoff from non-mine site areas of the lease		✖	✖			✖✖
Airborne dust and other particulates	✖	✖				✖✖
Exhalation and atmospheric transport of Radon-222 and decay products or progeny						
Surface water to groundwater	✖	✖	✖			
Human and non-human vectors		✖✖				✖✖
Land Application Areas infiltration and runoff		✖				
Bioaccumulation and trophic transfer	✖ (terrestrial)	✖✖				

NB: shaded cells represent combinations of stressors and pathways that do not occur.

Table 4 Gaps in knowledge about the ecological impacts of stressor-pathway combinations. '✖' denotes a knowledge gap (summarised from Table 1).

	Stressors					
Pathways	**Inorganic toxicants**	**Organic toxicants**	**Radionuclides**	**Radon-222 and decay products or progeny**	**Transported sediments**	**Weed propagules**
Surface water to surface water						✖
Airborne emissions	✖	✖				
Stormwater runoff from non-mine site areas of the lease						✖
Airborne dust and other particulates	✖	✖				✖
Exhalation and atmospheric transport of Radon-222 and decay products or progeny				✖ (biota)		
Surface water to groundwater						
Human and non-human vectors		✖	✖ (biota)			✖
Land Application Areas infiltration and runoff						
Bioaccumulation and trophic transfer		✖	✖ (terrestrial)			

NB: shaded cells represent combinations of stressors and pathways that do not occur.

Conclusions

While knowledge gaps exist for some pathways and stressors, there is no evidence to suggest that any of these pathways are currently resulting in adverse ecological impacts on the environment within the ARR. Results of ongoing physical, chemical, radiological and biological monitoring undertaken by the Supervising Scientist continue to show that the environment of the ARR remains protected from uranium mining-related impacts via the aquatic pathway (the dominant potential vector) and from airborne radionuclides in the case of human health protection. This work will also inform the future development of conceptual models of stressor pathways associated with the mine closure and rehabilitation phases of mine life. The closure pathways conceptual model will inform and assist the development of closure criteria and the specifying of the monitoring framework needed to address them.

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Characterisation of contamination at land application areas at Ranger

R Akber[1], A Bollhöfer & P Lu[2]

Introduction

A collaborative project led by Energy Resources of Australia Ltd (ERA) was started in 2007–08 between the Department of Environmental Strategy (DES) within ERA, *Safe Radiation* (Brisbane) and ***eriss***. The objectives of this project were to (i) characterise the magnitude and extent of radiological contamination at each of the Ranger LAAs and (ii) on the basis of site-specific measurements, develop parameters that enable radiation dose estimates during future occupancy of these areas due to the applied radioactivity during land irrigation. ***eriss*** has been involved in planning and scoping the project from the early stages. A major contribution to the project involved radioanalyses by ***eriss*** of different types of samples (soils, leaf litter, dust) from the Ranger LAAs, and provision of assistance and radioanalyses for the assessment of radon exhalation from the LAAs.

The assessment of the LAAs was finalised in 2012, and a series of reports have been produced on their radiological status (Akber et al 2011a–e). A final report (Akber & Lu 2012) summarises the findings and provides a model for dose estimates, including the ingestion pathway. It was found that external gamma radiation and inhalation of dust are the two main exposure pathways in the LAAs, responsible for 86% of the above background radiation dose. As these two pathways strongly depend on the vertical distribution of applied radioactivity, various rehabilitation options were suggested to reduce exposure of people potentially accessing the footprint of the LAAs after rehabilitation. These options include removal of the surface 10 cm of contaminated soil and placing it into the pit, tilling of the soil, or a mixture of both.

In late September 2011, a rehabilitation trial including 4 different rehabilitation treatments was initiated at the Magela B LAA (Figure 1). The objective of the rehabilitation trial was to investigate whether the theoretically predicted reductions in dose rates can be achieved.

- Treatment 1: Baseline – no soil removal or redistribution within the area.
- Treatment 2: Soil redistribution – tilling to 30–50 cm depth within 7 m radial distance from the sprinkler heads.
- Treatment 3: Soil removal – removal of surface 10 cm of soil within 5 m radius of location of the sprinkler head.
- Treatment 4: Soil removal (as per treatment 3) and redistribution – removal of 10 cm of soil followed by tilling.

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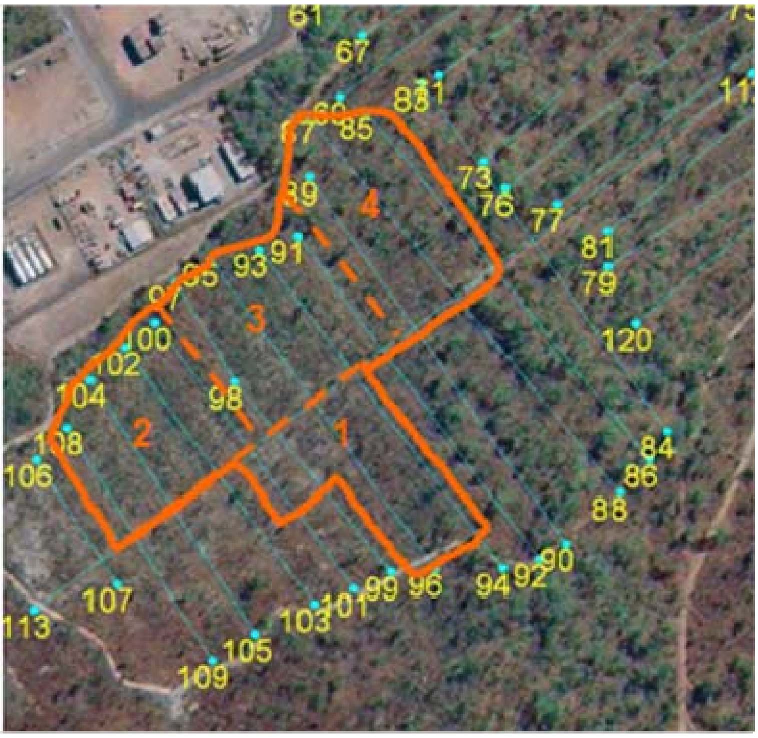

Figure 1 Location of the four treatments at the Magela B land application area

Experimental work was conducted in late 2011–12 in different sections of the trial plots, both before and after the earth works including (i) characterisation of vertical and horizontal distribution of radioactivity, (ii) measurement of external gamma dose rate, and (iii) measurement of radon activity flux density. The report provided here summarises the results of soil radionuclide activity concentrations measured in the ***eriss*** laboratories.

Methods

Thirty-five soil samples (0–10 cm) were collected by ERA at 1 m distance from selected sprinklers perpendicular to the irrigation lines before and after rehabilitation, to investigate the effects of the three rehabilitation treatments on the spatial distribution of radioactivity. To investigate the effects of the three rehabilitation treatments on the vertical distribution of radioactivity, 0–10 cm soil core samples were collected on either side of 34 different sprinklers. These cores were sliced into 7 sections (0–10, 10–20, 20–30, 30–40, 40–50, 50–70 and 70–100 mm). Samples from the same depth and rehabilitation trial treatment were combined resulting in 28 individual samples (7 depths × 4 treatments) from before and after rehabilitation.

All soil samples were dried and sieved to <16 mm fraction and delivered to ***eriss*** for radioanalysis. The samples were milled and weighed, and analysed for radionuclide activity concentrations of the different radioisotopes using the ***eriss*** high purity germanium (HPGe) gamma spectrometers. Methods for sample preparation and measurement are described in Murray et al (1987), Marten (1992) and Esparon and Pfitzner (2009).

Results

Figure 2 shows the activity concentrations of ^{226}Ra and ^{238}U from applied radioactivity in samples collected from 0 to 10 cm depth, both before and after the rehabilitation trial.

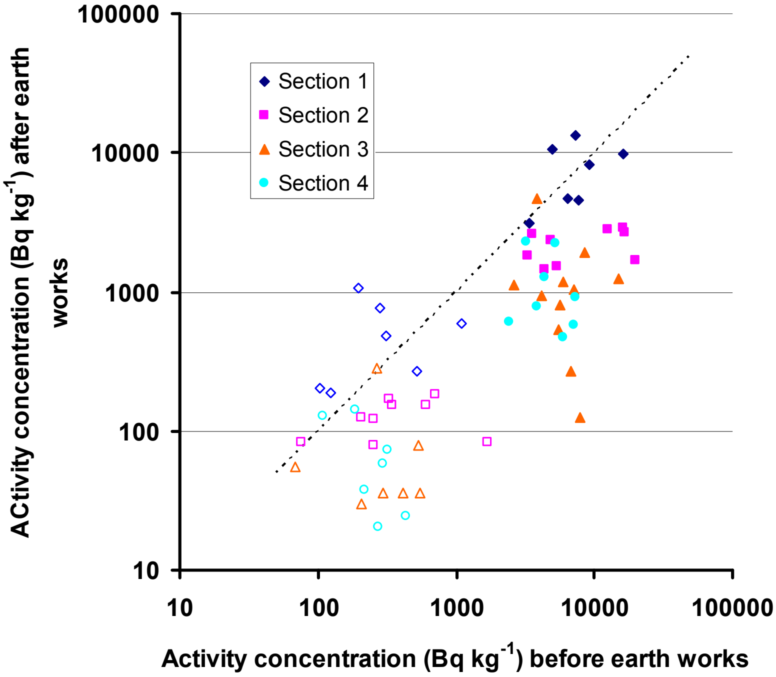

Figure 2 Applied ^{238}U (filled symbols) and ^{226}Ra (hollow symbols) activity concentrations in the top 0–10 cm of the soil, before and after the rehabilitation trials. The dotted line is the line of equality.

Figure 1 shows that treatment 1 (control) of the rehabilitation trial does not show systematic deviations from the line of equality when comparing pre- with post-rehabilitation activity concentrations. In contrast, post-rehabilitation activity concentrations in 0–10 cm soil from treatments 2–4 are, as expected, systematically lower than the pre-rehabilitation values. Reductions in radionuclide activity concentrations in the top 10 cm of the soils ranged from about 70% in treatment 2 to 85% in treatment 3.

Figure 3 shows the vertical profiles of ^{226}Ra and ^{238}U measured in the soil cores from 0 to 10 cm depth, before and after rehabilitation. Again, as expected, post-rehabilitation activity concentrations are generally lower in treatments 2 to 4 than those measured before rehabilitation, whereas there is no change in treatment 1.

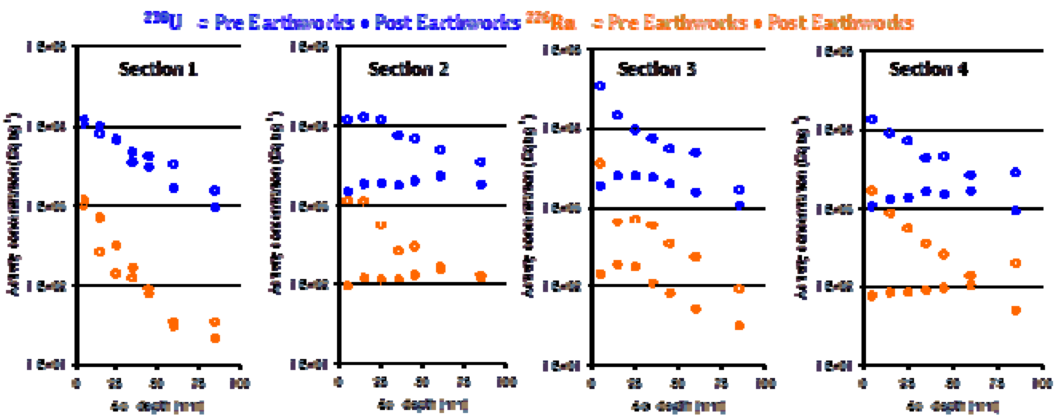

Figure 3 Vertical distribution of applied ^{226}Ra and ^{238}U activity concentrations from 0–10 cm depth, before and after rehabilitation trials

Treatments 2 and 4, which were both mixed to a depth of approximately 0.3–0.5 m as part of the rehabilitation treatment, show the expected near uniform distribution in the top 0–10 cm of the soil after rehabilitation. Treatment 3, however, which had 10 cm of the top soil removed and was analysed after soil removal without further tilling, shows a further decrease of activity concentration with depth for both ^{226}Ra and ^{238}U.

More results and in depth discussion of the project is given in Akber et al (2013).

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

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Background

Information concerning the toxicity of uranium (U) in sediments has been identified as an important knowledge gap for both the operation and closure phases of the Ranger mine, and internationally (van Dam et al 2010). Since 2009, we have been reporting on the progress of a field study that aims to address this paucity of information and determine a site-specific sediment quality guideline for U (van Dam et al 2010, Harford et al 2011, 2012). Two pilot-scale experiments have been completed to date based on fine-grained (mostly < 63 µm) billabong sediments. The field experiments involved spiking the sediments with U, placing these in mesh containers, then setting these into the littoral sediments of a backflow billabong over a full wet season to observe and measure natural colonisation of biota. The final experiment was postponed last (2011–12) wet season in order to complete the analysis of the 2nd pilot study and thereby better inform the design of the final experiment. Here we report the progress made in the analysis of the pilot study data and the plans for a full concentration-response experiment for the 2012–13 wet season.

The first pilot-scale experiment (Pilot 1) was undertaken during the 2009–10 wet-season. This study aimed to determine the most appropriate sediment spiking and deployment methods, the U concentration range and the replication required for a full scale experiment (see Harford et al 2011 for details of methods). Chemical analysis of spiked sediments from Pilot 1 showed that the initial binding of U to the sediment was rapid and complete but the sediment spiking method needed to be further refined to minimise the confounding physical disruption (ie compaction) of the sediment structure (Harford et al 2011). Thus, a second pilot study (Pilot 2) was undertaken during the 2010–11 wet season to investigate the effects of minimising (i) the amount of sediment manipulation, in particular, the sieving and removal of coarse inorganic and organic material (> 2 mm) and the extent of mixing, and (ii) aerial exposure prior to wet season inundation.

Apart from clay clods and obvious plant root mats, sediments used in Pilot 2 study retained the coarse fraction (> 2 mm). This study also focused on evaluating an alternative method for sediment spiking, the objective of which was to minimise both disturbance of the physical characteristics of the sediment and the duration of the storage period prior to deployment. The chosen method involved pouring a solution of uranyl sulfate over the surface of the sediment, allowing the solution to infiltrate through the undisturbed sediment profile, after which it drained through the mesh base of the container into a collecting vessel. At this point, the bulk material was mixed to facilitate the even distribution of U through the profile then the elutriated solution, along with any incidentally elutriated sediment, was re-poured over the sediment surface. This method was used to produce a control and three U treatments (0, 500,

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1000 and 2000 mg kg^{-1} U). Prior to deployment, sub-samples of the sediments were taken for chemical and limited ecogenomic analysis to define the starting chemical and biological condition (see Harford et al 2012 for further details).

Chemical analyses of the sediments prior to deployment showed that the measured concentrations were close to the nominal concentrations and that the U was well distributed through the sediments. The U was also demonstrated to be present predominantly in forms extractable in dilute acid (eg 80–100% extractable in 1 M HCl). After the wet season exposure period, the chemical composition of the sediments, which included characterisation of acid-volatile sulfide and organic carbon, indicated that little had changed over the duration of the wet-season. Preliminary analysis of the available data indicated that, despite naturally low diversity, the macroinvertebrate assemblages were responsive to the high U exposure where fewer taxa were observed compared with controls (Harford et al 2012). There were no significant differences in bacterial community diversity amongst treatments. However, closer interrogation of the dataset using multivariate analyses showed an improbable pattern of community structure. Specifically, there were two distinct groups that were > 97% different from one another. The first cluster contained all the samples from the Gulungul (site) control, low and medium U treatments, eight of the fifteen control samples and five of the fifteen high U samples. A second cluster contained all the pre-deployment samples, the remaining ten high U treatments and seven control samples (Harford et al 2012).

Activities in 2012 have focused on completing the analyses of the Pilot 2 data and using this to inform our experimental design and analytical approach for the main experiment. Specifically, the aims of the project for the reporting year were to:

1. Complete ecogenomics of eukaryote benthos and analyse the results;
2. Complete processing of the macroinvertebrate samples (from the low-uranium and medium treatments) and analyse the results;
3. Investigate the bacteria dataset anomaly (described above);
4. Determine the best design for the full concentration-response experiment, based on an investigation of statistical methods available for deriving toxicity estimates using concentration-response modelling of community assemblages.

Methods

Macroinvertebrates

The 10 remaining U treated samples (low, 500 mg kg^{-1} and medium 1000 mg kg^{-1} U treatments) were processed and macroinvertebrate communities quantified. Processing of the impoverished, fine-grained sediments was very labour intensive and for this reason the Gulungul (site) control samples were not processed. These site-controls would not provide useful information on the concentration-response relationship of the macroinvertebrate communities. The results were analysed by various statistical methods, as described below.

Microinvertebrates (eukaryotes)

The ecogenomic sequencing and bioinformatics processing were completed. Two samples had low diversity (a low number of Operational Taxonomic Units, OTUs) and were re-sequenced along with a selection of other samples in order to confirm the result. The results were analysed by various statistical methods, as described below.

Bacteria (prokaryotes)

In order to determine if the two distinct clusters noted above were an error in the sequencing or bioinformatics processing, five samples were chosen to be re-processed and re-sequenced. Two controls and two high U samples were chosen from each of the two clusters (four samples). One additional outlier, which was different from both the clusters, was also chosen for re-analysis. Archived DNA was processed at Charles Darwin University under the same conditions as the first samples. They were sent to a different service provider (University of New South Wales) for sequencing, while bioinformatic processing was completed at Charles Darwin University. The results were analysed using limited statistical methods in order to compare with the previously obtained results.

Statistical analyses

A number of statistical tools were evaluated in order to determine the method(s) most suitable to derive a sediment quality guideline for U from the field experiment data. Using the microinvertebrate and macroinvertebrate datasets, the standard tools available in Primer (V6.1.13, Primer-E, Plymouth) were first evaluated. These included PERMutational ANOVA (PERMANOVA), Analysis Of SIMilarities (ANOSIM), Canonical Analysis of Principle Coordinates (CAP), Distance-based Linear Models (DisLM) and distance-based Redundancy Analysis (dbRDA). Additionally, based on literature searches and recommendations from prominent statisticians, three R statistical software-based packages, Non-linear CAP (NCAP; Millar et al 2005), Threshold Indicator Taxa ANalysis (TITAN; Baker & King 2010) and Gradient Forests (Ellis et al 2012), were evaluated.

Results

Macroinvertebrates

The total abundance and taxa richness of macroinvertebrates in Gulungul Billabong were low across all treatment groups (Figure 1). Taxa richness was significantly lower in the highest (2000 mg kg^{-1}) treatment (ANOVA, $P = 0.042$) and, while total abundance appeared lower in the U treatments, this was not statistically significant due to large variation in the control (Figure 1). Multivariate analyses (PERMANOVA, ANOSIM, CAP and DistLM) showed no significant differences between any of the treatments and no significant correlations between the abiotic and biotic data.

Microinvertebrates (eukaryotes)

The re-sequenced samples appeared different to the original analysis. The issue was identified as a difference in the bioinformatics processing, and these samples will be reprocessed with the corrected bioinformatic scripts to determine if the two outlying samples can be incorporated into the dataset. In the interim, the outliers were removed from the statistical analyses as it was thought that they were likely to be a result of poor amplification of Polymerase Chain Reaction (PCR) products.

Compared with controls, the community assemblages of the microinvertebrates were not significantly different at 500 mg kg^{-1} U (PERMANOVA, $P = 0.12$) but were significantly different at 1000 and 2000 mg kg^{-1} U (PERMANOVA, $P = 0.01$). A CAP analysis also showed similarities between the control and 500 mg kg^{-1} U group, especially when using treatment group as an *a priori* predictor (Figure 2a). A CAP analysis using all TOC, pH, AVS and Acid Extractable Metal (AEM) concentrations as predictors also showed good separation of the groups with U concentration showing the strongest correlation with the differences in biotic data (Figure 2b). Uranium was commonly the best predictor of the observed differences in a number of different analyses but Al, Total Organic Carbon (TOC) and S were also significant predictors.

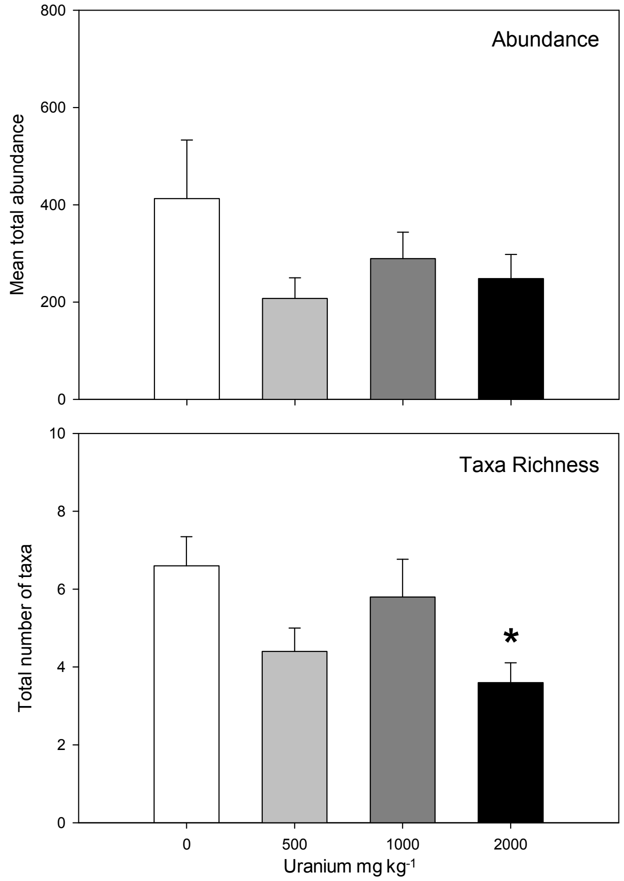


Figure 1 Macroinvertebrate abundance and taxa richness in the U treatments of Pilot study 2. An asterisk denotes a significant difference from the control (P < 0.05).

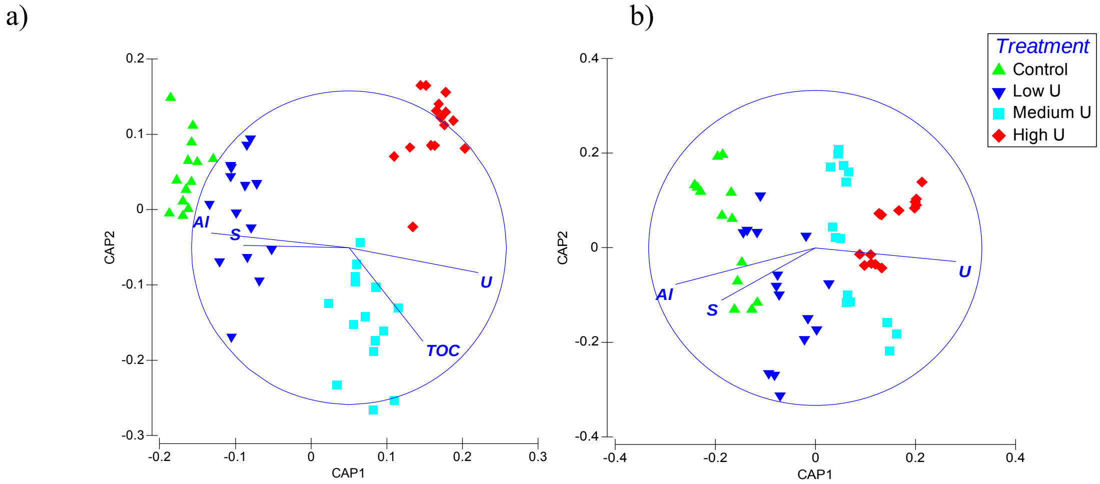


Figure 2 CAP analysis using a) treatment group as the predictor of differences b) environmental parameters as the predictor of differences, of the eukaryote communities exposed to the different U treatments (control – 0 mg kg^{-1}, low – 500 mg kg^{-1}, medium – 1000 mg kg^{-1}, high – 2000 mg kg^{-1}). Overlayed vectors show correlations between abiotic and biotic data of >0.6.

Bacteria (prokaryotes)

The re-sequenced bacteria samples were not >97% different as found in the original analysis, and the samples from the same treatments grouped closer to each other, as would be expected. Cluster analysis of the dataset indicated that, at the most, the re-sequenced samples were 60 to 70% different. This confirms that there was an error in the original dataset that occurred either during sequencing or during bioinformatic processing. The sequences from both runs will now be combined and undergo bioinformatic processing at CSIRO using different scripts. This will indicate if there error was in the bioinformatics or, if due to a sequencing error, whether it can be determined which sequences may be inaccurate so that they can be removed from statistical analyses.

Assessment of appropriate statistical analysis methods

The ultimate objective of this project is to determine a sediment quality guideline for U. In order to achieve this, a 'No Effect Concentration' (NEC – ie a threshold point along a concentration gradient above which a marked or significant shift in the measured response is observed) needs to be estimated from the community assemblage data. The characteristics of the statistical method(s) that best define this threshold may influence and dictate the design of the final experiment.

A number of multivariate hypothesis testing methods are able to compare significant differences between groups (eg PERMANOVA and ANOSIM), but are not designed to model concentration-response relationships or detect thresholds. Toxicity estimates from such tests are limited to one of the concentrations of the test treatments and cannot be interpolated between a NEC and observed effect. Such methods also require sufficient replication within a treatment in order to detect significant differences, which generally limits the maximum number of treatments that can be managed. In the field of ecotoxicology, limitations of hypothesis tests to determine toxicity estimates, and true NECs, have been well documented, and there has been a stated preference towards the use of concentration-response modelling (Landis & Chapman 2011, van Dam et al 2012). We have identified and investigated a number of assemblage analysis methods that are capable of modelling concentration-response relationships for community datasets.

Distance-based linear models (DistLM), redundancy analyses (dbRDA) and CAP are tools that can be used to predict effects on biota along environmental gradients. Analyses such as these might be used to determine a sediment quality guideline for U but the techniques lack the ability to determine a threshold due to the linear nature of the models. Consequently, a true NEC cannot be determined by these methods. Other multivariate techniques are capable of detecting thresholds of community change and show promise in the ability to determine a NEC that can be used to derive a sediment quality guideline. These tests include TITAN, Gradient Forests and Non-linear CAP.

The TITAN method (Baker & King 2010) is based on the indicator value scores developed by Dufrêne and Legendre (1997). The method integrates occurrence, abundance and directionality of taxa responses, to produce change points (thresholds) for individual species and the community as a whole. It does this for both species disappearing and species that are appearing along an environmental gradient. The method uses bootstrap sampling to assess the uncertainty of the change point and thereby estimates confidence limits around the value. TITAN analyses of the microinvertebrate and macroinvertebrate datasets reported change points of 560 and 340 mg kg^{-1}, respectively, for species that were disappearing. The Caenidae family was identified as a good macroinvertebrate indicator, while there were numerous indicator species identified in the larger microinvertebrate dataset. The TITAN method shows promise in that it can identify community level change at environmental thresholds, and it can also identify which species are appreciably influencing the change (see Figure 3). The method is limited to using only one key environmental predictor (U in this case) and other influencing environmental predictors (eg TOC) may be overlooked. This issue may be solved using a Principal Coordinate score that incorporates all the key environmental parameters, which may be used in place of a single environmental measurement.

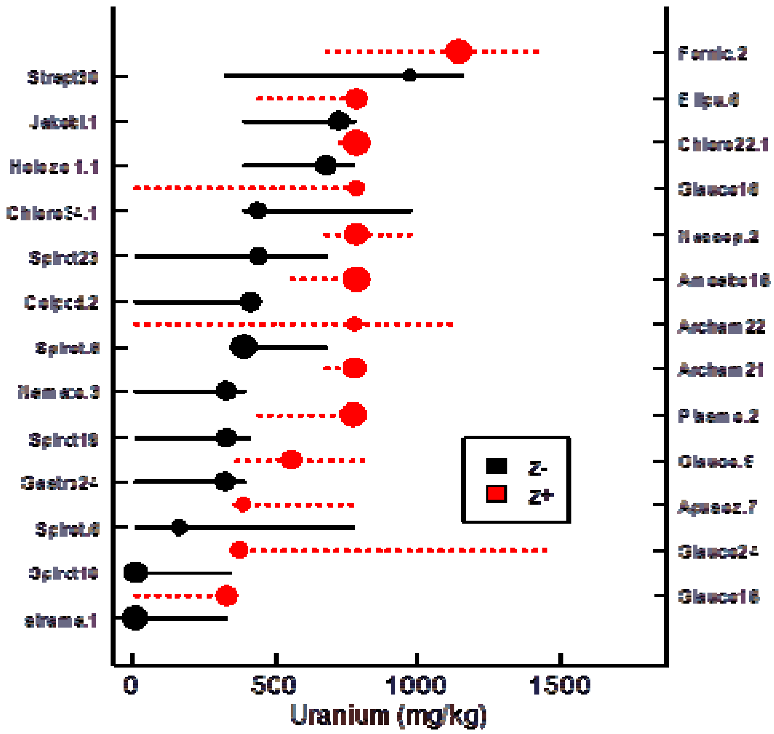


Figure 3 A 'taxaz' plot produced from the analysis of the microinvertebrate dataset using TITAN. The plot shows the best indicator species with high significance, purity and reliability scores. The species names are reduced to a 7 character representation. The terms z- and z+ are species that are disappearing from, and appearing in, samples, respectively. Dots are the calculated change point for the species and a larger dot indicates a higher indicator value. The lines indicate 95% confidence limits.

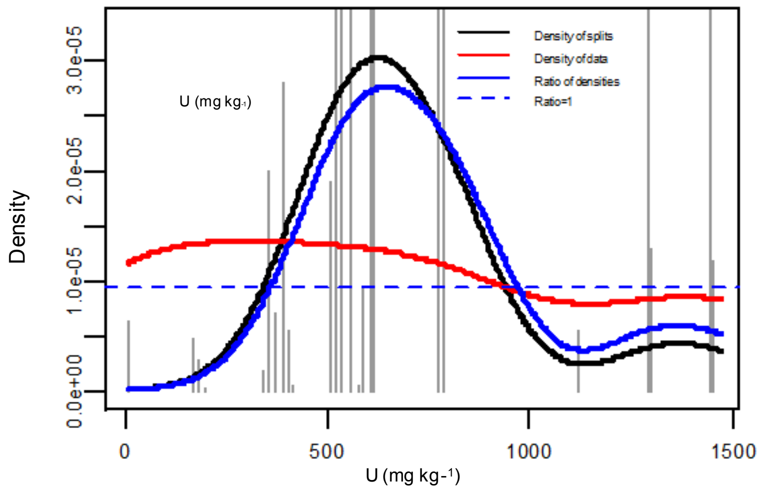
The Gradient Forests method (Ellis et al 2011) uses random (regression tree) forests, which are grown for each individual species. Each random forest consists of a group of regression trees that repeatedly split the observations into partitions. The splitting occurs at certain values of an environmental predictor and produces functions that represent the compositional turnover along each environmental gradient. Gradient Forest analysis of the microinvertebrate and macroinvertebrate datasets identified U as the most important environmental predictor.

The analysis indicates a compositional change for the macroinvertebrates that peaked at ~600 mg kg^{-1} U and it appeared that this may have been derived from just one key family, the Caenidae (Figure 4a & b). Interestingly, the caenids were also the only species identified by TITAN as a good indicator species.

Compositional changes of the microinvertebrate community appeared to occur at multiple points along the U concentration gradient but the most notable peak was at 300-400 mg kg^{-1} U (Figure 5a). The multiple compositional changes resulted in an overall community change that appeared not to have a notable threshold (Figure 5b). Like the TITAN method, Gradient

Forests shows promise as a tool for detecting a NEC because it can identify changes along an environmental gradient and can identify which species are changing at certain concentrations. It also has the advantage of being able to do this for any number of environmental parameters. However, the method does not produce a single change point value, although it is clear that one may be inferred from the outputs. The method also does not provide confidence limits around the estimates.

a)



b)

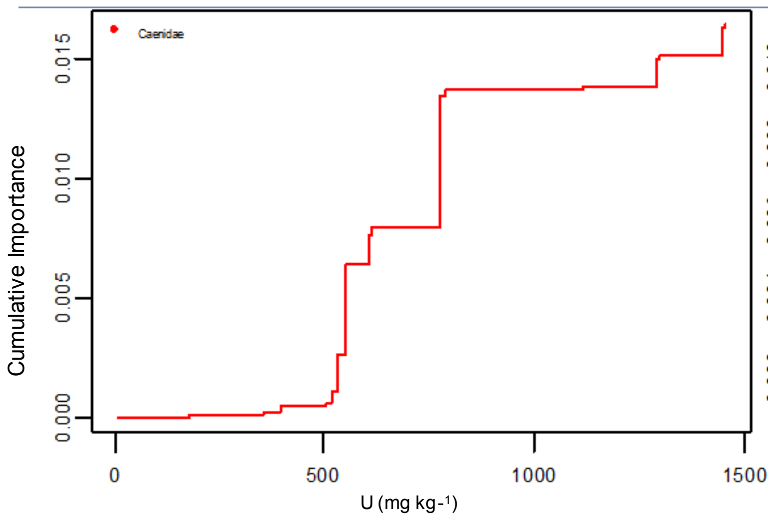


Figure 4 Macroinvertebrate dataset analysed by Gradient Forests a) the density of splits along the U concentration gradient shows where compositional changes are occurring and b) the cumulative change in composition of caenids across the U concentration gradient

The NCAP method (Millar et al 2005) is an extension of the CAP analysis and uses a link function that can be a model of any particular form, eg exponential decay, log-logistic. The method uses distance-based dissimilarity measures and conducts Principle Coordinate Analysis. A nonlinear gradient that maximises the canonical correlation with the principal coordinates is then defined. The significance of the fit of the nonlinear gradient is determined by a randomisation procedure and bootstrapping is used to determine confidence limits of parameters for the nonlinear model. The method has not yet been trialled using the macroinvertebrate dataset, while analysis of the microinvertebrate dataset was unable to determine a nonlinear relationship between the environmental gradients and community structures but this may be due to the lack of a true gradient in this dataset. Nevertheless, this method is attractive because log-logistic models, which are traditionally used to model concentration-response relationships, can be fitted to community datasets to derive 'toxicity estimates'. Consequently, the investigation of this method is ongoing.

a)

b)

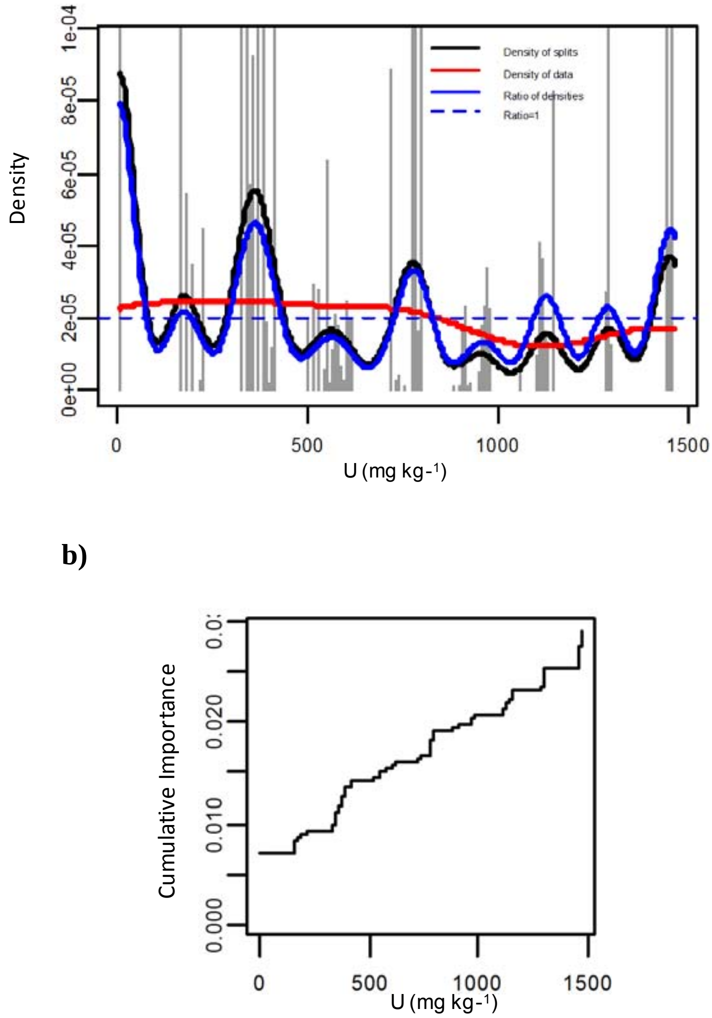


Figure 5 Microinvertebrate dataset analysed by Gradient Forests a) the density of splits indicating where compositional changes are occurring in relation to U concentrations, and b) the cumulative change in composition of the whole community across the U gradient

Conclusions and experimental design

The results from Pilot 2 demonstrated that changes in macroinvertebrate and macroinvertebrate communities should be detected with the current spiking method, physico-chemical characterisation, species enumeration methods and statistical techniques. However, it should also be noted that the analysis of bacterial communities will not occur in the first instance because of questions surrounding the ecological significance of a change in community structure. Specifically, sediment bacteria are now being primarily viewed as factors capable of modifying U toxicity due to their effect on the geochemistry of the sediments and consequent changes in metal speciation. An unknown functional redundancy is likely to exist in bacterial communities and it is impossible to determine how a community change may affect the sediment geochemistry. This issue is also compounded because a high proportion of abundant bacteria appear to be unclassified. Therefore, extracted DNA will be archived for possible future amplification of functional genes (eg N and C cycling) or species identification gene targets. Additional geochemical tests are also being considered.

The investigation of statistical methods found at least two methods that may be suitable for determining a NEC from the field data. However, discussions concerning the merit of the methods are ongoing and will be further informed after the data from the full-concentration response experiment has been collected. The most significant conclusion from the current statistical investigation is that an experimental design that maximises the number of treatments is preferable. This was also the advice provided by three external statisticians, two of whom are authors of TITAN and NCAP (pers comms: Matthew Baker, University of Maryland; Marti Anderson, Massey University; Jim Thomson, Monash University). Consequently, the concentration range for the final experiment will be: control (~5 mg kg^{-1} U), 50, 100, 200, 400, 600, 800, 1000, 1500, 2000, 3000, 4000 mg kg^{-1} U. Each treatment will consist of 4 replicates. The passive absorption method of U spiking described in Harford et al (2012) will be used to spike the sediments with U, and deployment, retrieval and processing will be the same as performed for Pilot 2.

Steps for completion

The following tasks are planned over the next 12–18 months to complete this project:

1 Determine the accuracy of the bacterial ecogenomic dataset (2012–13);

2 Complete statistical analyses of Pilot study components (2012–13);

3 Write an internal report and journal paper for Pilot studies (2012–13);

4 Complete the full concentration-response experiment (2012–13);

5 Analyse the 2013 and earlier data and derive a NEC and subsequent sediment quality guideline (2013–14);

6 Write an internal report and journal paper for the main experiment (2013–14).

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Reference toxicity testing for routine toxicity test species

KL Cheng, AC Hogan, AJ Harford & RA van Dam

Background

Reference toxicity testing has been a routine part of the ***eriss*** Ecotoxicology Program's QA/QC system since 2004–05. The methods were developed in accordance with formal guidance on reference toxicant testing (Environment Canada 1990). The aim for 2011–12 was to continue with the established reference toxicity testing program, using uranium (U) as the reference toxicant, for *Moinodaphnia macleayi*, *Chlorella* sp, *Hydra viridissima, Lemna aequinoctialis* and *Mogurnda mogurnda*.

Methods

Descriptions of the testing procedures are provided in Riethmuller et al (2003). There were variations to the *Lemna aequinoctialis* protocol in Riethmuller et al (2003). Specifically, the diluent used has been modified to use 1% CAAC (Centre for Advanced Analytical Chemistry) and growth rate calculated from surface area increase has been included as an endpoint. Details of these changes will be documented in a Supervising Scientist Internal Report.

Progress

In total, 18 reference toxicants tests (*Chlorella* – 6; *Hydra* – 2; *Moinodaphnia* – 5; *Mogurnda* – 2; and *Lemna aequinoctialis* – 3) were completed during 2011–12. Of these tests, 13 provided valid results, as summarised in Table 1. In order to generate a reference control chart, a sufficient number of properly conducted tests must be included in order to capture a representative range of variability. Environment Canada (1990) recommend the inclusion of effect concentrations (ECs) from at least 15–20 reference toxicity tests in order to calculate reliable warning limits (± 2 Standard Deviations, SD) and 99% confidence limits (± 3 SDs). The associated control charts are presented in Figures 1 to 5. The Ecotoxicology laboratory aims to complete four tests per species per annum. However, due to issues with the *Chlorella* sp. and *M. macleayi* reference toxicity testing (see below), this was not achieved for three species (*H. viridissima, M. mogurnda and L. aequinoctialis*) over the current reporting period. Every attempt will be made in future years to ensure the minimum numbers of tests are completed.

A summary of the issues identified during 2011–12 for each component of the reference toxicity test program is provided below.

Chlorella sp (green alga)

Of the six *Chlorella* sp tests, four were valid for this reporting period. For all tests, control growth rates were within the acceptability criterion of 1.4 ± 0.3 doublings/day. Two tests (1260G and 1262G) did not achieve a full effect and the EC_{50} could not be calculated. This may have been due residual nutrients from the culture medium, which would ameliorate toxicity.

The running mean EC_{50} is 39 µg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 5 and 72 µg L^{-1} U, respectively (Figure 1).

Table 1 Summary of the results from reference toxicity tests

Species & endpoint	Test Code	EC_{50} (µg L^{-1})	Valid?	Comments
Chlorella sp	1238G	19 (17, 22)	Yes	
(72-h cell division growth rate)	1260G	NC[b]	No	Did not get full effect[a]
	1262G	NC[b]	No	Did not get full effect[a]
	1269G	44 (10, 58)	Yes	
	1286G	51 (35, 65)	Yes	
	1296G	47 (35, 60)	Yes	
Hydra viridissima	1258B	82 (75, 97)	Yes	
(96-h population growth)	1288B	91 (85, 96)	Yes	
Moinodaphnia macleayi	1249I	17 (7, 19)	Yes	
(48-h immobilisation)	1250I	NC[b]	No	High control mortality
	1263I	234 (218, 256)	No	Exceeds upper warning limit[a]
	1267I	133 (94, 263)	Yes	
	1289I	204 (184,221)	No	Exceeds upper warning limit[a]
Mogurnda mogurnda	1255E	1242 (1150, 1342)	Yes	
(96-h sac fry survival)	1283E	1359 (1282, 1440)	Yes	
Lemna aquinoctialis	1248L	8360 (6411, 12040)	Yes	
(96-h growth rate)	1287L	9822 (8683, 10950)	Yes	
	1301L	11780 (10010, 13930)	Yes	

Values in parentheses represent 95% confidence limits

[a] See text for discussion

[b] Not calculable

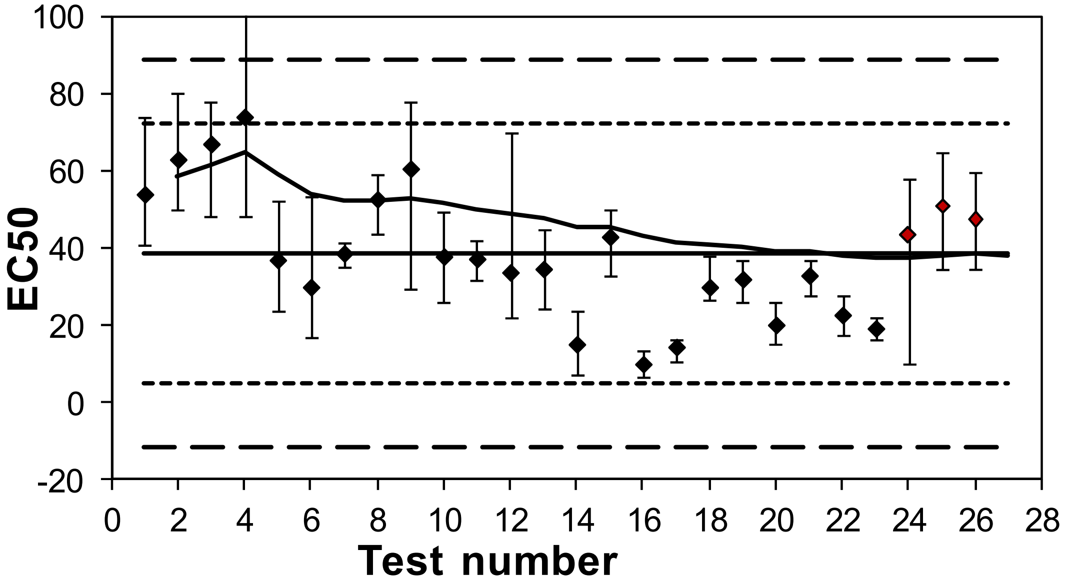

Figure 1 Reference toxicant control charts for *Chlorella* sp, as of Oct 2012. Data points represent EC_{50} µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

H. viridissima (green hydra)

Both reference toxicity tests for *H. viridissima* were valid. There were no issues associated with this protocol. The running mean EC_{50} is 83 µg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 42 and 124 µg L^{-1} U, respectively (Figure 2).

M. macleayi (cladoceran)

Of the five reference toxicity tests for *M. macleayi*, two were valid. Test 1249I and 1267I resulted in an EC_{50} of 17 and 133 µg L^{-1} U, respectively. Tests 1263I and1289I both had EC_{50}s that exceeded the upper warning limit of 200 µg L^{-1} U. Test 1250I was invalid due to unacceptably high control mortality. The current running mean EC_{50} (lower, upper warning limits) is 72 (<0, 200) µg L^{-1} U (Figure 3). Investigations into the apparent reduced sensitivity of *M. macleayi* to U are ongoing and some of this work is described in the current ARRTC report 'Influence of dissolved organic carbon on the toxicity of uranium to the cladoceran, *Moinodaphnia macleayi*'.

A reduction in *M. macleayi* sensitivity to U in late 2011 to early 2012 coincided with a change in an ingredient (alfalfa powder) of the fermented food with vitamins (FFV). As previously reported at ARRTC 27, differences in the sensitivity of *M. macleayi* to U have been attributed to different batches of FFV. Consequently, it was hypothesized that the organic components of FFV had altered due to a change in FFV ingredients, which increased the binding of U to the organic components and reduced the bioavailability of the U. A test was conducted to compare the effects of different FFV batches, which were produced before and after the ingredient change, to determine if this was contributing to the reduced sensitivity to U. However, the hypothesis could not be appropriately tested due to high control mortality in the treatments fed the older FFV. A semi-quantitative chemical characterisation of the FFV, through Nuclear Magnetic Resonance (NMR), showed that there were no notable differences in the functional groups of batches made before and after the ingredient change. Broader investigation of this issue found that *M. macleayi* exhibited similar sensitivity to that previously published, for magnesium (Mg) and copper (Cu). Specifically, the EC_{50} for Mg was 77 (67–87; 95% CL) mg/L compared with 63 (56–70) mg/L reported by van Dam et al (2010). For Cu, a 54% effect was observed following exposure of *M. macleayi* to the previously reported EC_{50} of 23 µg/L (Orchard et al 2002). Thus, the issue appears to be specific to U, and may not be due to the FFV. Investigations are continuing in 2012–13.

M. mogurnda

Both reference toxicity tests for *M. mogurnda* were valid with the EC_{50} values within the lower and upper warning limits of 944 and 2068 µg L^{-1} U, respectively. There were no problems associated with this protocol. The current running mean EC_{50} is 1506 µg L^{-1} U (Figure 4).

L. aequinoctialis

The reference toxicant test method for *L. aequinoctialis* was finalised, and the reference toxicity control chart currently includes ten valid tests with data based on growth rate based on frond number. The current running mean EC_{50} is 10,491 µg L^{-1} U and all results within the lower and upper warning limits of 7764 and 13219 µg L^{-1} U, respectively (Figure 5).

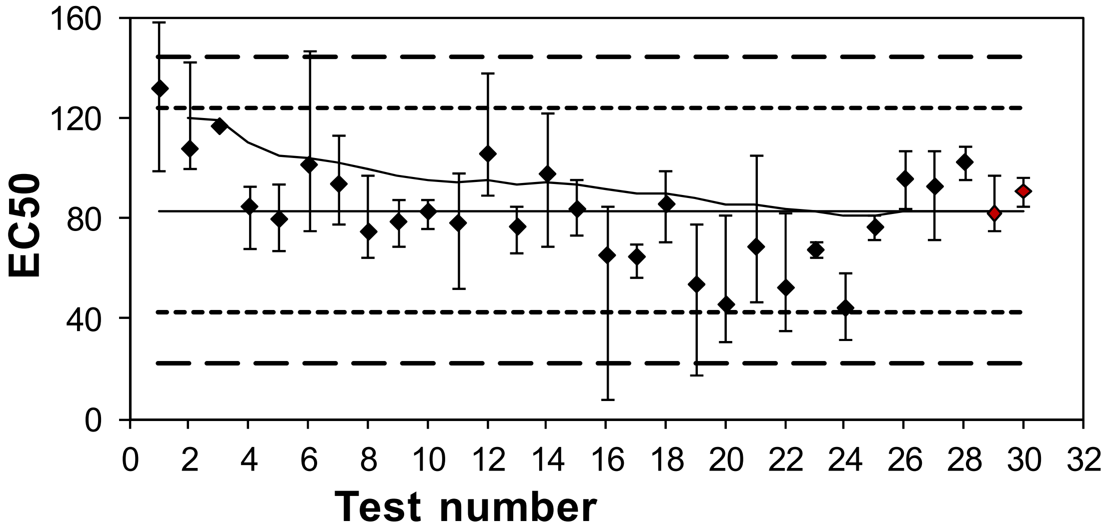

Figure 2 Reference toxicant control charts for *H. viridissima* as of Oct 2012. Data points represent EC_{50} µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. The last 2 data points represent tests conducted within this reporting period.

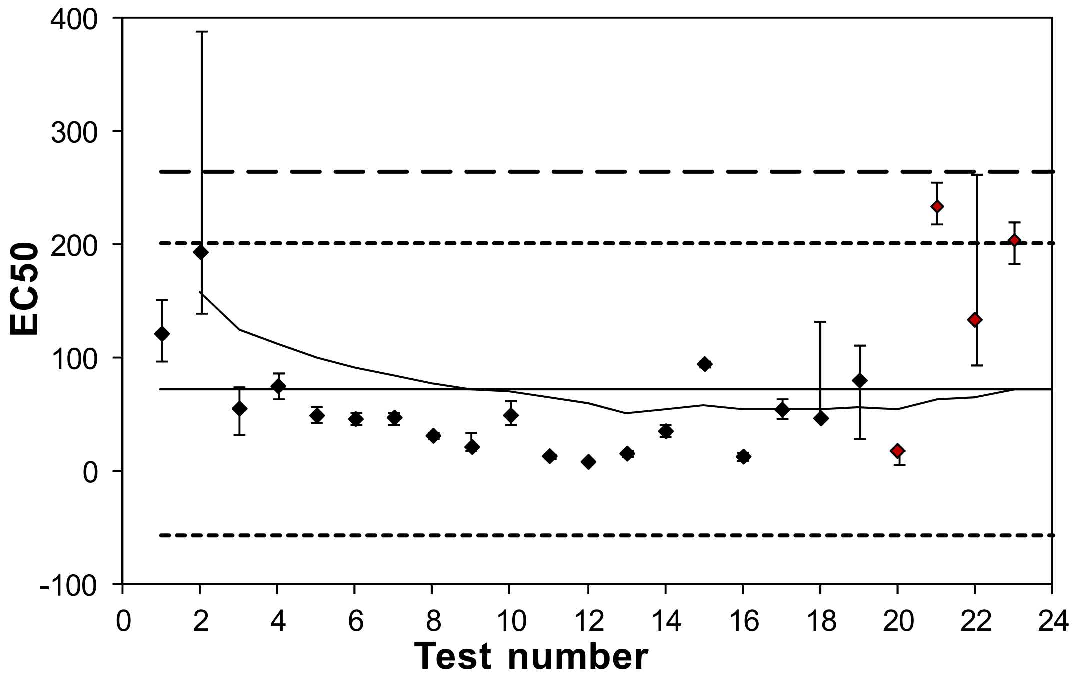

Figure 3 Reference toxicant control charts for *M. macleayi* as of Oct 2012. Data points represent EC_{50} µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. The last 4 data points represent tests conducted within this reporting period.

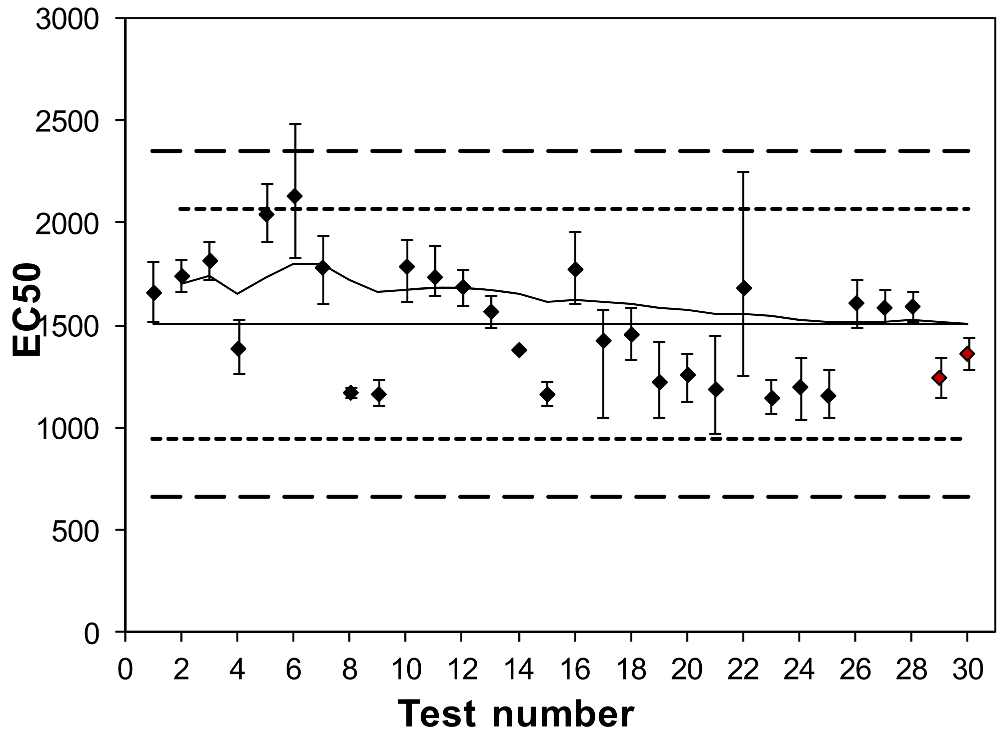

Figure 4 Reference toxicant control charts for *M. mogurnda* as of Oct 2012. Data points represent EC_{50} µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. The last 2 data points represent tests conducted within this reporting period.

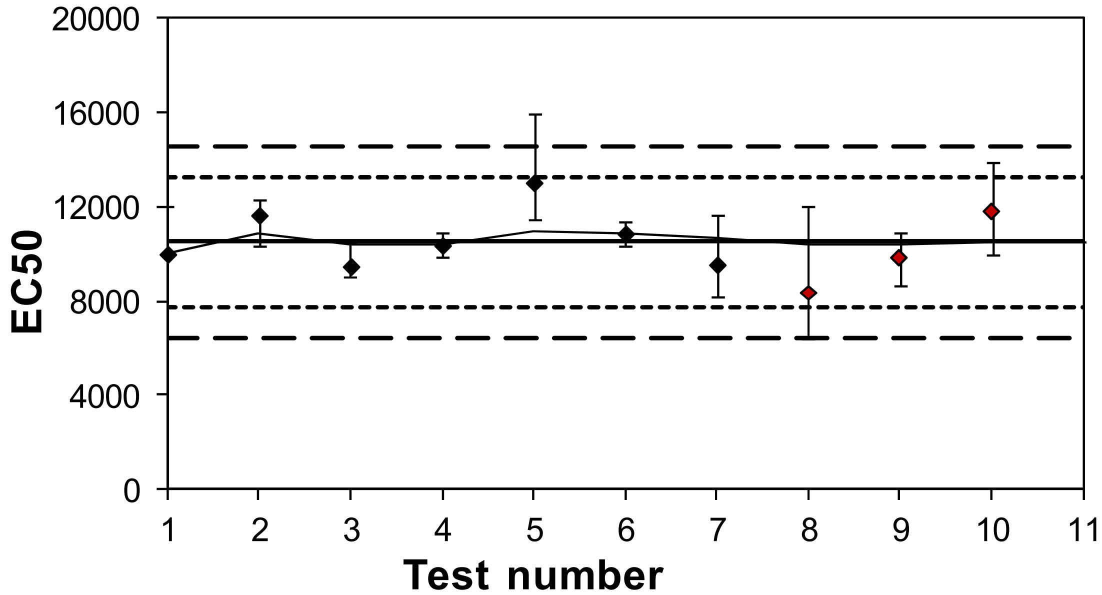

Figure 5 Reference toxicant control chart for *Lemna aequinoctialis*. Data points represent EC_{50} µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. The last 3 data points represent tests conducted within this reporting period.

Planned testing in 2012–13

The reference toxicity testing programs for all five species will continue in 2012–13, with the aim of completing at least four tests per species. Investigations will continue into the reduced sensitivity of *M. macleayi* to U. An Internal Report for the method development of the *L. aequinoctialis* reference toxicity protocol is currently in progress.

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Derivation of a trigger value versus exposure duration model for EC and Mg using pulse exposure toxicity data

AC Hogan, MA Trenfield, AJ Harford & RA van Dam

Background

Concentrations of mine-derived solutes in Magela Creek from the Ranger uranium mine do not occur at constant levels, but vary widely during the wet season due to changes in creek discharge, mine water discharge and mine water source. Continuous monitoring of electrical conductivity (EC) in Magela Creek since the 2005–06 wet season has confirmed the marked variability in creek water quality associated with mine water discharges (Supervising Scientist 2012).

Electrical conductivity is the key signature variable for the effect of discharges of Ranger mine water into Magela Creek. The EC is dominated by magnesium sulfate ($MgSO_4$) with a strong linear relationship having been established between EC and Mg using over five wet seasons of data (Supervising Scientist 2011). Consequently, concentrations of Mg can be confidently predicted from EC measurements. A large body of research has been undertaken on the toxicity of $MgSO_4$ to local aquatic species, resulting in the derivation by SSD of a site-specific water quality trigger value (TV) for magnesium (Mg) in Magela Creek of 2.5 mg/L (Mg being the predominant toxic ion in $MgSO_4$; van Dam et al 2010). However, the Mg toxicity data and associated TV were based on a chronic exposure regime over several days (3 to 6 days depending on the species) and, as such, are not representative of the majority of the much more transitory environmental exposures in Magela Creek.

Comparison of the proposed Mg TV with the continuous monitoring EC data from the 2005–06 to 2008–09 wet seasons revealed 43 exceedances of the TV. The median Mg concentration and exceedance duration was 3.4 mg L^{-1} and 6 h, respectively, and all except one of the exceedances were of durations much shorter than the published chronic toxicity test exposure regimes (Hogan et al 2012). Therefore, they were considered unlikely to be causing detrimental effects downstream of the mine, but at that time there were no quantitative data to support this assumption. Given the high conservation value of the Magela Creek catchment, it was considered necessary to better understand the potential effects of short-term exceedances of the Mg TV (ie pulse exposures of Mg).

The aims of the study were to (i) assess the toxicity to local freshwater species of Mg pulse exposures relevant to those measured in Magela Creek, and (ii) use the data to develop a model from which Mg or EC TVs can be derived for any given exposure duration. Most aspects of this project have been previously reported to ARRTC. This summary presents the final toxicity results and subsequent development of the Mg/EC exposure duration – TV model.

Methods

Six local freshwater species (green alga, *Chlorella* sp; duckweed, *Lemna aequinoctialis*; snail, *Amerianna cumingi*; cladoceran, *Moinodaphnia macleayi*; green hydra, *Hydra viridissima*; and northern-trout gudgeon, *Mogurnda mogurnda*) were exposed to single Mg pulse exposures of

4 h, 8 h and 24 h (at a constant Mg:Ca ratio of 9:1, as per van Dam et al 2010), before being transferred to clean water and their responses (eg reproduction, growth, etc) monitored for the remainder of the standard toxicity test durations (3 to 6 days depending on species). Two to four toxicity tests were undertaken for each species/pulse duration combination. A limited number of continuous Mg exposure toxicity tests were completed to confirm the responses and toxicity values previously derived and reported in van Dam et al (2010).

Results and discussion

For all species, Mg toxicity increased as exposure duration increased. However, the extent to which toxicity increased differed between species, from 2-fold to 40-fold. Moreover, the nature of the positive relationship between toxicity and exposure duration differed between species, from linear to exponential. The concentrations of Mg resulting in 10% inhibition of response (IC10) for each species are summarised in Table 1. The IC10 data are considered to be reliable measures of a low/acceptable effect on species, and are presently the preferred toxicity measure for deriving TVs.

Table 1 Magnesium IC10[a] values for each species and Mg pulse exposure duration

	IC10 value (mg L^{-1})			
Species	**4-h pulse**	**8-h pulse**	**24-h pulse**	**Continuous exposure**[b]
Chlorella sp	5950	5620	3880	818
Lemna aequinoctialis	4030	1500	80	36
Amerianna cumingi	3030	387	301	5.6
Moinodaphnia macleayi [c]	212	62	128	39
Hydra viridissima	1210	1000	709	246
Mogurnda mogurnda [d]	>4100	>4100	>4100	4010

a IC10: concentration at which there was 10% inhibition in response of the organism.

b Continuous exposure toxicity data from van Dam et al (2010).

c *M. macleayi* data for exposure at onset of reproductive maturity shown only.

d *M. mogurnda* data represent concentrations at which there is mortality of 5% of larvae (ie LC05; due to this test being an acute test).

For one species, the cladoceran *M. macleayi*, increased sensitivity to Mg was observed following pulse exposures at the onset of reproductive maturity (ie at ~27-h old) compared with exposure of neonates (<6-h old). This increased sensitivity may be related to the coincidence of exposure with the physiological processes of moulting and/or reproductive development.

The data shown in Table 1 were used to derive 99% species protection TVs for each exposure duration, based on log-logistic species sensitivity distributions consistent with the procedure documented in the Australian Water Quality Guidelines (ANZECC/ARMCANZ 2000). The resultant TVs for each pulse exposure duration are shown in Table 2.

The EC, and corresponding Mg, TVs presented in Table 2 were plotted against exposure duration (Figure 1). Polynomial interpolation was used to enable the prediction of EC TVs for any given exposure duration within the range of durations that have been assessed (ie from 4-h to 72-h). Polynomial interpolation involves fitting a polynomial model of $n - 1$ degrees, where n is the sample size, to the data. Such a model, in this case an inverse third order polynomial, will pass exactly through the data points to provide a smooth approximate interpolation of the data.

Table 2 Trigger values for Mg and EC for different pulse exposure durations

Pulse duration	99% species protection trigger value	
	Mg (mg L^{-1})	EC (µS cm^{-1}) [a]
4 hours	94	1140
8 hours	14	174
24 hours	8	102
Continuous (3–6 days) [b]	3	42

[a] EC calculated based on the Mg trigger value, using an established EC versus Mg relationship.

[b] Continuous exposure trigger values taken from van Dam et al (2010), noting that the value of 3 mg L^{-1} has been rounded up from 2.5 mg L^{-1}.

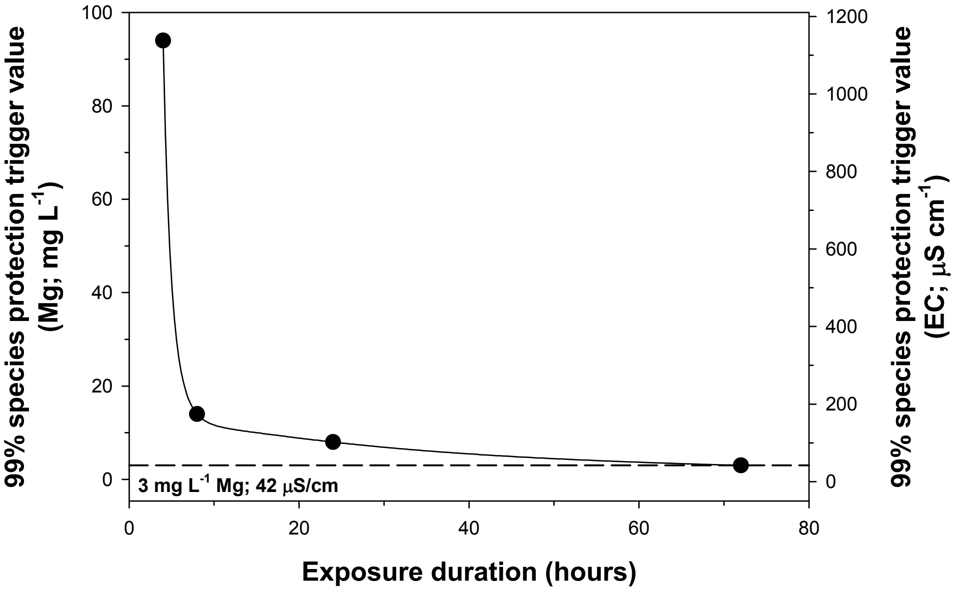

Figure 1 Relationship between trigger value, expressed as Mg (mg L^{-1}) and EC (µS cm^{-1}; after being converted from Mg concentration using an established relationship), and exposure duration. The fitted line is an inverse third order polynomial. The chronic exposure TV of 3 mg L^{-1} Mg or 42 µS cm^{-1} (horizontal broken line) is shown for comparison.

Tayler and Frostick (in prep) undertook a detailed analysis of the historical EC continuous monitoring dataset using the above Mg/EC TV – pulse duration relationship in order to assess the model's functionality and usefulness within a trigger framework. They found that the peak ECs of all the short-term events in Magela Creek were well below the duration-based TVs interpolated from the model. For example, the highest magnitude pulse of 124 µS cm^{-1} that lasted for 5 h 30 min was only 29% of the duration-based TV of 433 µS cm^{-1}. The longest duration EC pulse of 17 h 20 min, with a peak EC of 60 µS cm^{-1} was 52% of the duration-based TV of 115 µS cm^{-1}. Moreover, an average event, with a duration and magnitude around the mean of all events (4 h 20 min, 50 µS cm^{-1}), was only 5.5% of the duration-based TV.

With the treatment of pond and process water becoming an increasingly significant component of the water management regime at Ranger Mine, the quality of mine water entering Magela Creek should improve such that the frequency of exceedances of the EC Guideline/chronic Limit will decrease in the future, at least until the mine decommissioning phase commences. As such, there is little justification to embark on a large-scale study of multiple pulses at this point in time. However, it is recommended that additional research be pursued to investigate organism recovery time and the potential for carry-over toxicity. This would provide further guidance in applying the trigger framework to multiple pulse scenarios.

The results of the Mg pulse exposure toxicity study have been detailed in an Internal Report (Hogan et al 2012), and submitted to *Environmental Toxicology and Chemistry* for peer-reviewed publication.

Steps for completion

During the 2012–13 wet season, this model will form the basis of a magnesium/EC trigger value framework that will provide improved interpretation of the potential for environmental effect of transient pulses of EC (and magnesium) in Magela Creek downstream of the Ranger mine. This framework may replace the current weekly grab sample-based approach that is the basis of the current regulatory compliance regime.

Further work assessing organism recovery time and the potential for carry-over toxicity may be necessary to provide guidance on the application of the Mg/EC TV framework to situations where multiple pulses occur within short timeframes.

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Toxicity of manganese to tropical freshwater species

RA van Dam, AJ Harford & AC Hogan

Background

Manganese (Mn) was recognised as a contaminant of potential ecotoxicological concern at Ranger mine in the early 2000s following observations of increasing concentrations in shallow groundwater adjacent to Magela Creek (MC20; up to 50 000 µg L^{-1}) and concentration spikes in Coonjimba Billabong and Corridor Creek (800–1600 µg L^{-1}; van Dam et al 2004). Notwithstanding these high concentrations, Mn concentrations in Magela Creek itself have remained between 2 and 15 µg L^{-1}. The current site-specific guideline for Mn in Magela Creek of 26 µg L^{-1} (based on upstream reference site data; Iles 2004) has been exceeded in less than 2% of the Magela Creek water samples collected since 1980 (Harford et al 2009). Consequently, the risks of Mn toxicity to aquatic biota have been considered low to date. However, the likelihood of higher concentrations of Mn being released to Magela Creek in Ranger mine waters may increase with the commissioning of the brine concentrator plant in mid-2013. A pilot-scale brine concentrator plant tested in 2011 produced distillate waters containing Mn at concentrations ranging from 130–240 µg L^{-1} (Harford et al 2009). These Mn concentrations are higher than those currently measured in mine waters discharged from Ranger (RP1 had 0.2 to 63 µg L^{-1} during 2011–2012), and the addition of distillate to such waters may eventually result in Mn concentrations in Magela Creek higher than have previously been measured.

Historically, the acute and chronic toxicity of Mn to freshwater biota has been considered to be low (ie in the mg L^{-1} range), as reflected in the relatively high 99% species protection trigger value (TV) reported in ANZECC/ARMCANZ (2000) of 1200 µg L^{-1}. However, a recent review of Mn toxicity in freshwaters by the Environment Agency (UK) has recommended a Predicted No Effect Concentration (PNEC) of 123 µg L^{-1} (Peters et al 2010). Furthermore, three previous preliminary studies investigating Mn toxicity to local ARR species (***eriss*** unpublished data 1993; Harford et al 2009; Harford et al 2012) have indicated that the green hydra, *H. viridissima*, is more sensitive to Mn than any other species reported in the literature, with no-observed-effect concentrations and 10% inhibition concentrations (IC10) between 20 and 180 µg L^{-1}. These concentrations are within the range of residual Mn concentrations measured in process water distillate and, thus, highlighted the need for a more comprehensive assessment of Mn toxicity under environmentally relevant conditions.

Currently, insufficient Mn toxicity data exist for local species in natural Magela Creek water (NMCW) to be able to (i) conclude with high confidence that no adverse effects would be expected given the current water quality, and (ii) predict at what Mn concentrations adverse effects would be expected to occur. This is particularly important given that the low water hardness and relatively low pH of NMCW represent favourable conditions for Mn bioavailability (Peters et al 2010, 2011).

Consequently, a study was initiated, with the following aims:

1 Assess the toxicity of manganese (Mn) in NMCW (pH ~6–6.5) to six tropical freshwater species;

2 Derive a site-specific water quality trigger value for Mn based on the toxicity data; and

3 Determine the ability of water hardness to ameliorate Mn toxicity.

Methods

Standard ecotoxicological protocols (Riethmuller et al 2003, Houston et al 2007) for six local freshwater species are being used to determine the toxicity of Mn in NMCW. For each toxicity test, filtered (0.1 µm) Mn concentrations were measured at test commencement and end. For some toxicity tests, additional Mn chemistry is being assessed (as described below). Average water chemistry of the NMCW samples used for the toxicity testing to date has been as follows: pH – 6.4; conductivity – 16 µS cm^{-1}; alkalinity – 6 mg L^{-1} as $CaCO_3$; dissolved oxygen – 85% saturation; and dissolved organic carbon – 2.5 mg/L^{-1}.

At least two toxicity tests are being undertaken for each species. The data from each of the tests are pooled and analysed using concentration-response regression modelling (3-parameter logistic or sigmoidal models). In addition to describing the full toxicity response, the concentration-response models are used to estimate the Mn concentration resulting in a 10% (IC10) and 50% (IC50) reduction in the measured organism response (eg reproduction, growth) relative to the control response.

Progress to date

Chemistry

With the exception of the *H. viridissima* tests (see *Toxicity* results and discussion below), there was very little difference between the 0.1 µm filtered Mn concentrations measured before and after the tests, indicating negligible loss (including precipitation) of Mn from the test systems.

Toxicity

To the end of October 2012, Mn toxicity had been characterised for *Chlorella* sp (algae) and *Lemna aequinoctialis* (duckweed, Figure 1 and Table 1). Manganese toxicity to both species was very low, with *L. aequinoctialis* more sensitive than *Chlorella* sp. Comparison with data from the literature is very difficult, as the present study appears to be the first that has assessed Mn toxicity under conditions of low pH (ie pH <6.5) and very low water hardness (ie ~5 mg L^{-1} as $CaCO_3$). Nevertheless, some comparison can be made. Based on IC10s, *Chlorella* sp. was 5× less sensitive than the green alga, *Pseudokirchneriella subcapitata* (pH ~8, hardness ~75 mg L^{-1} as $CaCO_3$), while *Lemna aequinoctialis* was 15× more sensitive than *Lemna minor* (pH ~7, hardness ~50 mg L^{-1} as $CaCO_3$) (Peters et al 2010). These comparisons do not further inform the role that pH and hardness play in determining Mn toxicity, and further studies would need to be initiated if the influence of toxicity modifying factors needs to be understood.

Characterisation of Mn toxicity to the other species is underway, but not yet completed. Approximate toxicity estimates based on the test work completed to date are shown in Table 2. *Hydra viridissima* (green hydra) and *Moinodaphnia macleayi* (cladoceran) are markedly more sensitive than the other species assessed in the present study. The high sensitivity of *H. viridissima* relative to the other species is consistent with the previous preliminary Mn toxicity testing undertaken by Harford et al (2009) (Table 2). Only one other species has been reported to be as sensitive to Mn exposure as *H. viridissima*. Peters et al (2010) reported an EC10 of 0.096 mg L^{-1} for the amphipod, *Hyalella azteca* (pH ~7.7, hardness ~100 mg L^{-1} as $CaCO_3$). Typically, Mn no/low effect toxicity estimates (eg EC/IC10s, no-observed-effect-concentrations) for freshwater species are > 1 mg L^{-1} (ANZECC/ARMCANZ 2000, Peters et al 2010). It is noteworthy that two of the species tested in the present study had IC10s < 1 mg L^{-1}.

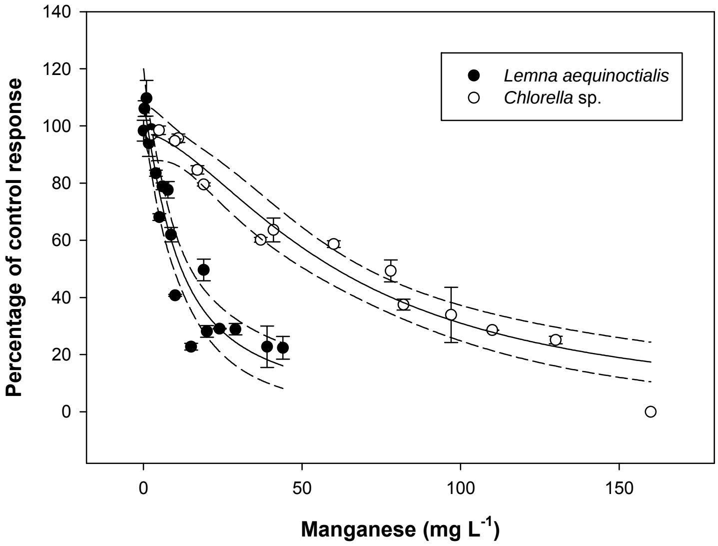

Figure 1 Manganese concentration-response curves for the green alga, *Chlorella* sp., and the duckweed, *Lemna aequinoctialis*. Data points represent the mean (± standard error) of 2-3 replicates. A 3-parameter logistic model was used for both datasets (*Chlorella* sp.: n = 15, r^2 = 0.95, P < 0.0001; *L. aequinoctialis*: n = 18, r^2 = 0.90, P < 0.0001). Broken lines represent the 95% confidence limits for the curve fits.

Table 1 Final manganese toxicity estimates for the green alga, *Chlorella* sp., and the duckweed, *Lemna aequinoctialis*

Species	Mn toxicity (mg L^{-1})	
	IC10[a] (95% CLs)[b]	**IC50[a] (95% CLs)**
Chlorella sp.	14 (nc[c]–21)	60 (50–70)
Lemna aequinoctialis	2.4 (nc–4.1)	12 (10–15)

[a] IC10 and IC50: concentrations resulting in a 10% and 50% effect relative to the control response, respectively.

[b] 95% CLs: 95% confidence limits.

[c] nc: not calculable.

Table 2 Preliminary manganese toxicity estimates for the green hydra, *Hydra viridissima*, the cladoceran, *Moinodaphnia macleayi*, the snail, *Amerianna cumingi* and the fish, *Mogurnda mogurnda*

Species	No. tests	Mn toxicity (mg L^{-1})			
		IC10[a]		**IC50[a]**	
		This study	Harford et al (2009)	This study	Harford et al (2009)
Hydra viridissima	2	0.09	0.06	1.7	0.77
Moinodaphnia macleayi	2	0.48	0.65	1.1	1.3
Amerianna cumingi	2	0.60	NT[b]	6.0	NT
		LC05[c]		**LC50[c]**	
Mogurnda mogurnda	3	74	NT	260	NT

[a] IC10 and IC50: concentrations resulting in a 10% and 50% effect relative to the control response, respectively.

[b] NT: Not tested.

[c] LC05 and LC50: concentrations resulting in 5% and 50% mortality relative to the control response, respectively.

An unusual observation has been the loss of a significant proportion of Mn from the test solutions during the *H. viridissima* tests, especially below 230 µg L^{-1} (Figure 2). This loss of Mn from the test waters has not been observed for any of the other test species (although water chemistry had not yet been received for the *A. cumingi* tests), and also did not occur in the hydra toxicity test reported by Harford et al (2009). Potential sources of Mn loss included adsorption to the test solution bottles and/or the test containers, precipitation and/or adsorption/absorption by the test animals. The preliminary toxicity estimates reported in Table 2 for *H. viridissima* were based on Mn concentrations calculated by averaging the before and after test 0.1 µm filtered Mn concentrations in the test solutions.

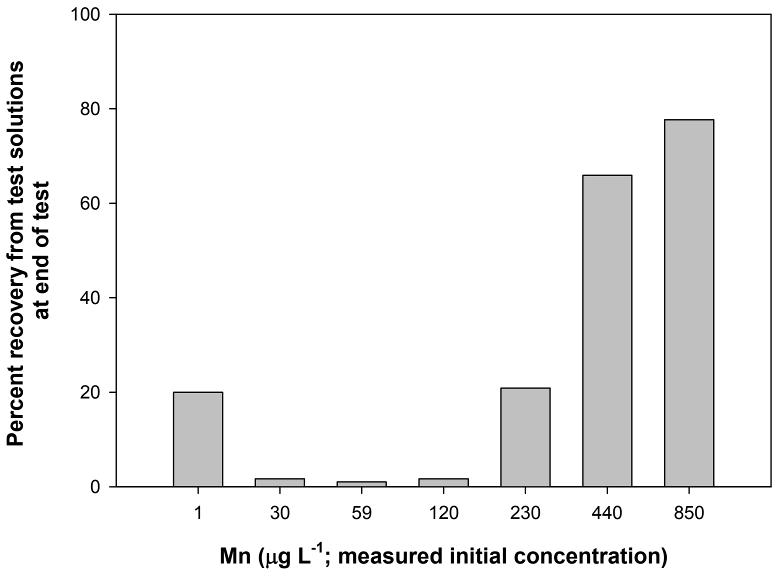

Figure 2 Percentage recovery of Mn in test solutions at the end of a *Hydra viridissima* Mn toxicity test. Samples from each replicate were pooled for chemical analysis.

An experiment was conducted to assess the fate of Mn in the hydra test system. Three Mn concentrations in NMCW (background, 250 and 600 µg L^{-1}) were assessed. An additional treatment was included for each Mn concentration, whereby the test petri dishes were pre-inoculated with a solution of 250 µg Mn L^{-1} for 24 h prior to the test commencement. This treatment was incorporated to see if Mn binding sites on the petri dishes could be saturated prior to the experiment, thereby reducing this source of Mn loss during the test. Measurements of Mn were made on the following components of the test system:

- Test solutions from the test petri dishes at test commencement and every 24 h just prior to test solution renewal, until the end of the test (96 h) (total and 0.1 µm filtered Mn);
- Test solutions from the 5 L test solution storage bottles at the commencement and end of the test (total and 0.1 µm filtered Mn);
- Hydra tissue at the end of the test (total Mn in all hydra);
- The surface of the test petri dishes, following rinsing with 5% HNO_3 (total Mn).

The results indicated that total Mn loss (from beginning of test to end of test) was similar to that described above. Interestingly, on a day by day basis, Mn loss appeared to be greatest on day 4, which is counter to the hypothesis that Mn is adsorbing to the test dishes (where a decrease in daily loss over the test period is normally observed, eg Hogan et al 2010). The higher loss of Mn on day 4 coincided with the appearance of a floating precipitate on the last day of the test (presumably a form of Mn-oxyhydroxide, although this was not characterised), particularly in the 600 µg L^{-1} treatment. Currently, it is unclear why this precipitation occurred on the last day of the tests (NB: test solution pH on day 4 was not higher than on the previous 3 days, suggesting a speciation change due to an increase in pH was not responsible for the precipitation). Pre-inoculating the test dishes with Mn appeared to only slightly reduce Mn loss in the 600 µg L^{-1} treatment (by ~20%), but not the 250 µg L^{-1} treatment, compared with no Mn pre-inoculation. These data require further analysis and consideration, and a further trial may be undertaken to assess the reproducibility of some of the results.

The reasons for the difference in Mn fate in the current testing with *H. viridissima* compared to that reported by Harford et al (2009) are unclear. Although there was some difference in test diluents pH between the two studies (pH 6.2 for Harford et al compared to pH 6.5 for the current study) and even though Mn speciation is pH-dependent, the kinetics of Mn speciation are extremely slow, and such pH differences are considered unlikely to result in significant speciation changes over the 96-h time course of a hydra experiment (Barry Chiswell, University of Queensland, pers comm). This issue requires further investigation.

Steps for completion

The remaining Mn toxicity testing will be completed over the next 3 months (from November), and a site-specific TV will be available prior the commissioning of the brine concentrator. Following the initial toxicity assessment, a targeted assessment of the effect of water hardness (Ca and Mg concentrations) on Mn toxicity may be undertaken for the two most sensitive species (likely to be *H. viridissima* and *M. macleayi*). Moreover, the effect of pH on Mn toxicity may also be examined at a later stage, over the relevant range of pHs observed in Magela Creek.

To complement the toxicity data, Mn speciation will be examined using both geochemical speciation modelling and ultra-filtration (0.003 µm; using tangential flow filtration). The full study will be published as one or two papers in an international peer-reviewed journal.

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Influence of dissolved organic carbon on the toxicity of uranium to the cladoceran, *Moinodaphnia macleayi*

MA Trenfield, CE Costello, KL Cheng, AJ Harford & RA van Dam

Background

Since 2007, the ***eriss*** Ecotoxicology Program has been researching the influence of dissolved organic carbon (DOC) on the toxicity of uranium (U) to tropical freshwater biota (Trenfield et al 2011a,b, 2012, van Dam et al 2012). However, this research did not include the cladoceran, *Moinodaphnia macleayi*, which has been found to be the most sensitive of those species studied to date, to U. The key constraint to using this species for such studies was the fact that, during chronic tests, test animals are fed with an organic-rich suspension (fermented food with vitamins – FFV), containing both particulate and dissolved organic matter. This makes quantitative assessment of U toxicity across varying DOC concentrations difficult to achieve. In particular, the presence of particulate organic matter would prevent a proper assessment of the effect of dissolved organic matter on U toxicity. This study set out to attempt to overcome this constraint and, if possible, quantify the influence of DOC on U chronic toxicity to *M. macleayi*. A previous preliminary study had found that DOC did indeed reduce the acute toxicity of U to *M. macleayi* in short-term tests where animals were not fed (Hogan et al 2009).

Methods

Culturing

Moinodaphnia macleayi cultures were kept in natural Magela Creek water (MCW; pH 5.5–6.5, EC < 20 µS/cm, DOC 2–6 mg/L) at 27 ± 1°C on a 12 h day:12 h night photoperiod, and fed daily with 1 µL/mL fermented cladoceran food (FFV) and 2×10^5 cells/mL algal (*Chlorella* sp) cells (Riethmuller et al 2003).

Optimising diet

The *M. macleayi* diet was modified so as to remove the potential confounding effect of the particulate organic matter (POM) present in the FFV on the assessment of the effect of DOC on U toxicity. The POM was removed via filtration. Trials were conducted under culturing conditions to determine the effect of filtering FFV (0.45 or 0.1 µm filtered) on *M. macleayi* survival and reproduction. Also, the background DOC concentration in the test vials after adding the filtered FFV (at 1 µL FFV/mL) was characterised.

Filtration resulted in ~20% reduction in offspring numbers relative to unfiltered FFV, but survival was not affected (Table 1). There was no difference between 0.45 or 0.1 µm filtered FFV (Table 1). The reduction in reproduction was considered an acceptable consequence of removing the potential confounding effect of POM, and the 0.45 µm filtered FFV was used for the U toxicity/DOC tests. The background DOC concentration in the test vials after adding the filtered FFV was 1.7 mg/L.

Table 1 Effect of filtration of FFV food source on reproduction and survival of *M. macleayi* over 3 reproductive broods

Endpoint	FFV type		
	Unfiltered	**0.45 µm filtered**	**0.1 µm filtered**
Mean offspring per adult (SEM)[1]	34.9 (±0.35)	26.7 (±0.60)	26.8 (±1.3)
Percent adult survival	100%	100%	100%

[1] SEM: Standard error of the mean.

Toxicity testing

The 3-brood *M. macleayi* toxicity test method is detailed in Riethmuller et al (2003). Healthy *M. macleayi* neonates less than 6-h old were exposed to a range of U concentrations (0–750 µg/L) at a range of DOC concentrations (1.7 [background], 6.0, 9.3 and 19.5 mg/L) under 'static renewal' conditions as per the culturing environment, but using the modified feeding regime detailed above. DOC was added in the form of Suwannee River Fulvic Acid (SRFA) Standard I. Background DOC comprised that already present in MCW and the filtered FFV food source. Daily observations were made on test organism survival, the number of neonates produced and neonate survival. The test was terminated when 80% of surviving control animals had released their third brood offspring (8–9 days).

Speciation modelling

The speciation of U in the test solutions was calculated using the WHAM 6.0 geochemical speciation code, with input parameters based on measured physico-chemical MCW data.

Statistics

Data from the U toxicity / DOC tests were pooled and presented as a function of the control response. Non-linear regression (3-parameter logistic or sigmoidal) was used to generate uranium concentration-response curves for each DOC concentration. Toxicity estimates, specifically, IC10s and IC50s, were obtained from the curve fits.

Progress to date

Two tests were completed before the study was postponed due to the apparent reduction in sensitivity of *M. macleayi* to U (see below for discussion). The results from the two tests clearly showed that DOC reduced the toxicity of U to *M. macleayi* (Figure 1; Table 2). Based on IC50s, the reduction in U toxicity with increasing DOC concentration displayed a strong linear relationship (Figure 2). Across the DOC range tested (1.7–19.5 mg/L), DOC reduced the toxicity of U 9-fold (based on IC50 concentrations; Table 2). Speciation modelling confirmed the reduction in U toxicity was due to a decrease in toxic bioavailable U species, including the free uranyl ion, UO_2^{2+}, through its complexation with SRFA (Figure 3).

However, the toxicity of U at the background DOC of 1.7 mg/L (IC50 = 138 µg/L) was markedly lower than had previously been reported for this species under similar testing conditions (ie same test method, using Magela Creek water). In *M. macleayi* chronic U toxicity tests from the early 1990s to 2006, the IC50 ranged from 21 to 57 µg/L (median 36 µg/L; *n* = 9; van Dam et al 2008, 2012). This reduction in sensitivity of *M. macleayi* to chronic U exposure is consistent with the reduction in acute U toxicity that has been observed in the reference toxicity testing program. Efforts to identify the cause of this change in sensitivity are underway, and have been summarised in the paper 'Reference toxicity testing program for routine toxicity test species', this volume.

A key reason for undertaking this study was that DOC concentrations were not measured for any of the historical chronic U toxicity tests using *M. macleayi*. If the DOC correction algorithm for U toxicity developed by van Dam et al (2012) is going to be used to adjust the site-specific U Limit for Magela Creek, then DOC data associated with U toxicity data are required for all species in the dataset, including *M. macleayi*. However, the recent U toxicity data for which DOC concentrations are available do not reflect the historical high sensitivity displayed by this species. Consequently, a revised U Limit based on a dataset that included the most recent *M. macleayi* data would not account for the historical high sensitivity. This highlights the importance of identifying the cause/s of the reduced sensitivity of *M. macleayi* to U.

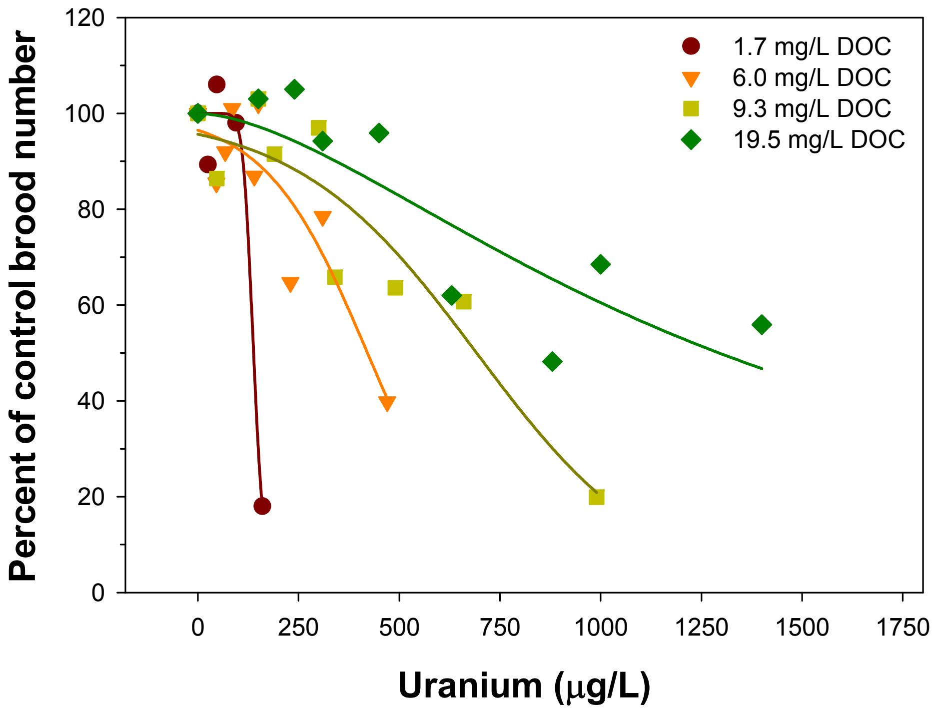

Figure 1 Concentration-response curves for uranium to *M. macleayi* at different DOC concentrations. Each value represents a mean of 10 replicates. Curve fits are based on 3-parameter sigmoidal or logistic models.

Table 2 Uranium IC50s for *M. macleayi* at different DOC concentrations

DOC concentration (mg/L)	Uranium IC50 (95% CLs) (µg/L)	Model r^2 and P value
1.7	138 (98–165)	0.94, 0.028
6.0	421 (330–589)	0.75, 0.004
9.3	692 (551–899)	0.84, 0.001
19.5	1290 (912 – > 2000)	0.72, 0.005

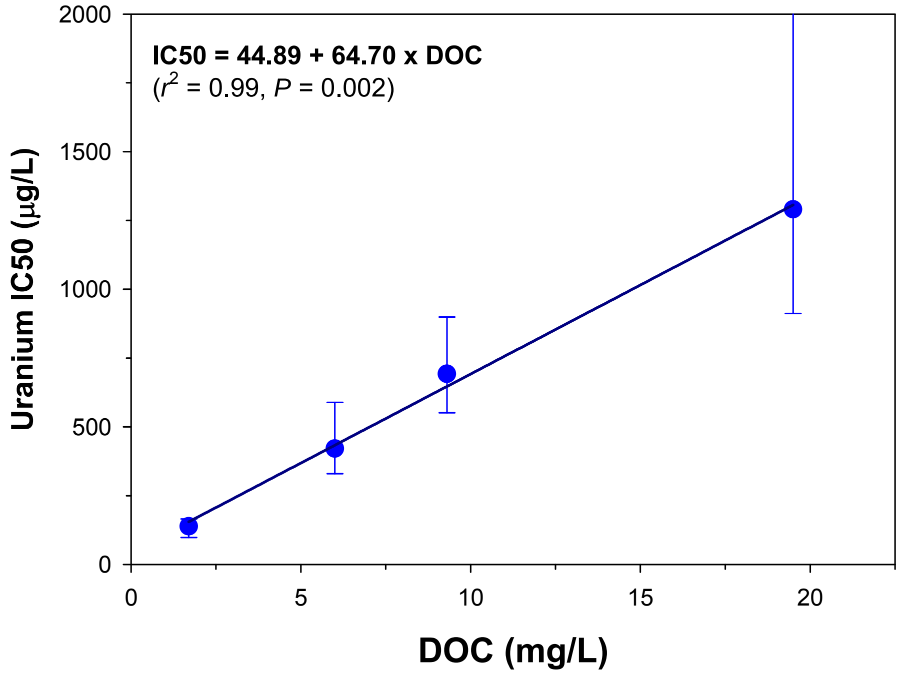

Figure 2 Relationship between uranium toxicity, expressed as the IC50, for *M. macleayi*, with DOC concentration

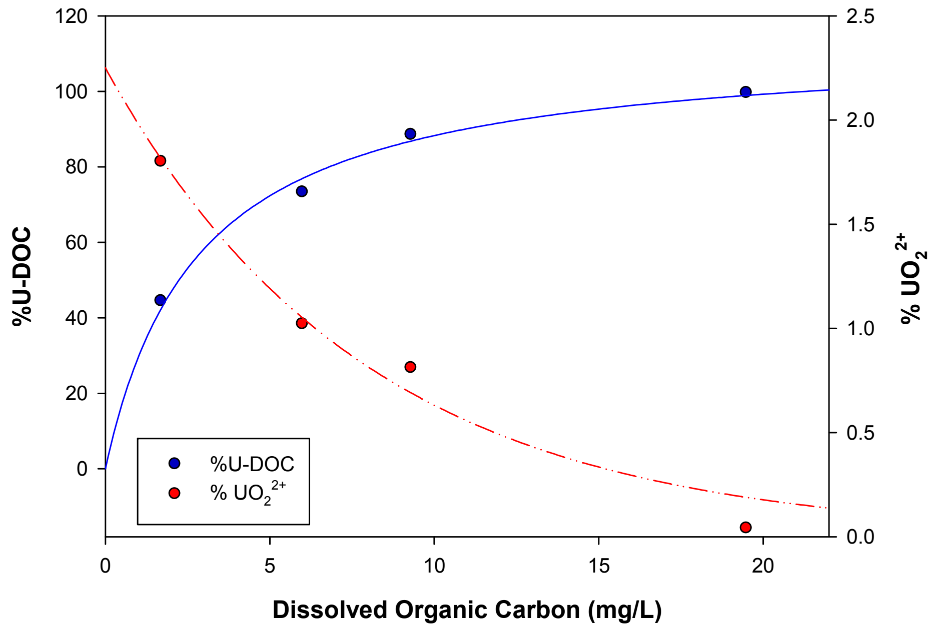

Figure 3 Effect of DOC on the proportion of uranium complexed with DOC and present as the free uranyl ion (UO_2^{2+}), at the uranium IC50 concentrations (see Table 2), as predicted by the WHAM 6.0 speciation model

Steps for completion

Until the causes/s of the apparent reduced sensitivity of *M. macleayi* to U can be identified, it will be difficult to complete this study and, subsequently, incorporate the DOC correction algorithm into any update of the current site-specific limit for U in Magela Creek.

Additional work to try to progress this issue is likely to include the following:

- assessment of the effect of various organic food sources on U toxicity to *M. macleayi*;
- assessment of the effect of various organic food sources on U speciation, primarily through the use of ultra-filtration using tangential flow filtration;
- collection and assessment of the sensitivity of a population of *M. macleayi* from the field.

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Incorporating the influence of dissolved organic carbon on uranium toxicity into the site-specific trigger value for uranium

RA van Dam, MA Trenfield, SJ Markich,[1] AJ Harford, CL Humphrey & JL Stauber[2]

Background

The bioavailability and toxicity of uranium (U) to aquatic biota are known to be influenced by a number of environmental variables, including pH, water hardness and dissolved organic carbon (DOC). However, historically there have been insufficient data to develop predictive relationships. The data deficiency for one of the variables, DOC, has been addressed by a recently completed research project (Trenfield 2012). The results showed that DOC reduces U toxicity largely by complexing with U, thus reducing its bioavailability to aquatic organisms. At present, the current site-specific water quality trigger value (TV) for U in Magela Creek (6 µg/L) does not account for this effect of DOC on U toxicity. With concentrations of DOC in the Magela Creek stream channel typically in the range 2–8 mg/L, and exceeding 20 mg/L in backflow billabongs, it is likely that U bioavailability and toxicity will vary considerably depending on location in the Magela Creek catchment. Consequently, an analysis was undertaken to determine whether the DOC–U toxicity relationship could be incorporated into the site-specific U TV, such that the TV could be adjusted to account for the aquatic DOC concentration.

Approach

The relationship between DOC and U toxicity was characterised using data for the five freshwater species for which this dependence has previously been investigated, based on studies by ***eriss*** (Hogan et al 2005, Trenfield et al 2011, 2012) and the Australian Nuclear Science and Technology Organisation (Markich et al 2000). These species were the green alga, *Chlorella* sp, green hydra, *Hydra viridissima*, mussel, *Velesunio angasi*, northern-trout gudgeon, *M. mogurnda* and the dinoflagellate, *Euglena gracilis*. The studies using *V. angasi* and *M. mogurnda* were based on acute U toxicity, and were included to increase species representation and to improve understanding of the DOC-toxicity response.

Results and discussion

The relationships between DOC and U toxicity based on IC50 concentrations for the above species are shown in Figure 1 and Table 1. It is evident that the relationship varies depending on the species, DOC source and exposure duration (acute versus chronic). Statistically, the slopes of these relationships (see Table 1) were significantly different (based on Analysis of Covariance). Thus, the data could not be pooled to create a single DOC versus U toxicity

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relationship (as per the method of the US Environmental Protection Agency for water hardness-based correction of trigger values for various metals.

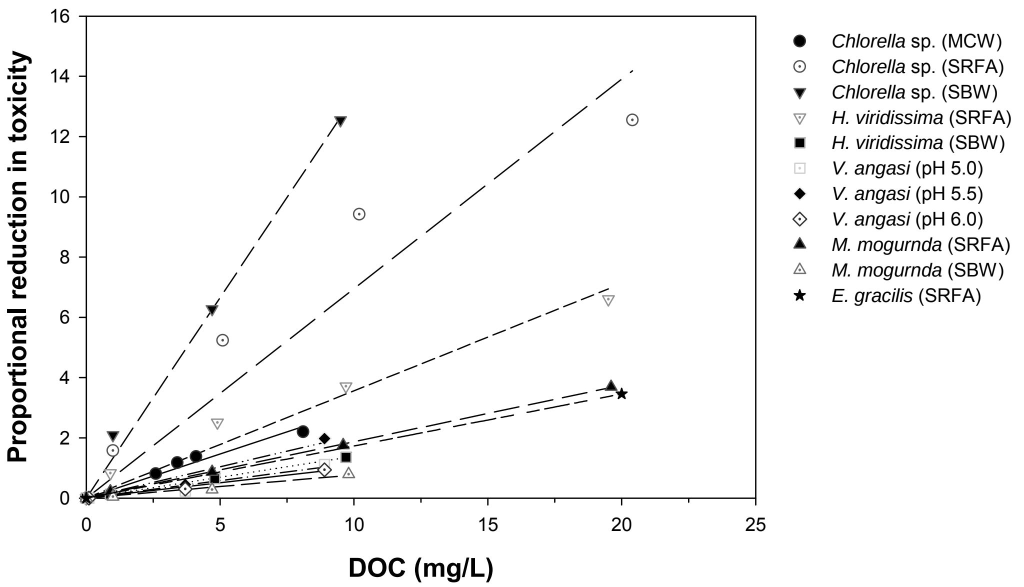

Figure 1 Linear relationships of U toxicity (expressed as the proportion reduction in IC/LC50 relative to the IC/LC50 at the background DOC concentration) versus dissolved organic carbon (DOC; from various sources) concentration for *Chlorella* sp, *Hydra viridissima*, *Velesunio angasi*, *Mogurnda mogurnda* and *Euglena gracilis*. DOC sources: MCW – Magela Creek Water; SRFA – Suwannee River Fulvic Acid; SBW – Sandy Billabong water.

Table 1 Regression statistics for linear relationships of dissolved organic carbon (DOC; mg/L) versus the proportion reduction in IC50/LC50 value for uranium

Species	Acute/chronic	DOC source[1]	*n*	Slope	r^2	P value
Chlorella sp	Chronic	MCW	5	0.30	0.95	0.003
	Chronic	SRFA	5	0.69	0.86	0.014
	Chronic	SBW	4	1.3	0.99	0.003
Hydra viridissima	Chronic	SRFA	5	0.35	0.95	0.003
	Chronic	SBW	4	0.14	0.99	<0.001
Velesunio angasi	Acute	SRFA (pH 5.0)	3	0.11	0.87	0.17
	Acute	SRFA (pH 5.5)	3	0.21	0.89	0.15
	Acute	SRFA (pH 6.0)	3	0.10	0.97	0.08
Mogurnda mogurnda	Acute	SRFA	5	0.19	0.99	<0.001
	Acute	SBW	4	0.08	0.97	0.011
Euglena gracilis	Chronic	SRFA	2	0.17	n/a[2]	n/a

[1] DOC sources: MCW – Magela Creek water; SRFA – Suwannee River fulvic acid; SBW – Sandy Billabong water.
[2] n/a: Not applicable, due to the model being based on only 2 values.

Instead, the slopes of the relationships between DOC and (normalised) acute and chronic U toxicity were modelled using cumulative probability distributions (Figure 2). The 5th percentiles of the distributions were defined as correction factors to be applied to acute or chronic U toxicity values or to produce TVs based on the measured aquatic DOC concentration. The 5th percentiles of the slope distributions were chosen as the correction factors to ensure that they did not under-protect an unacceptably large number of species from the effects of DOC on U toxicity. Moreover, this choice is consistent with the use of 5% hazardous concentrations for contaminants in risk assessments.

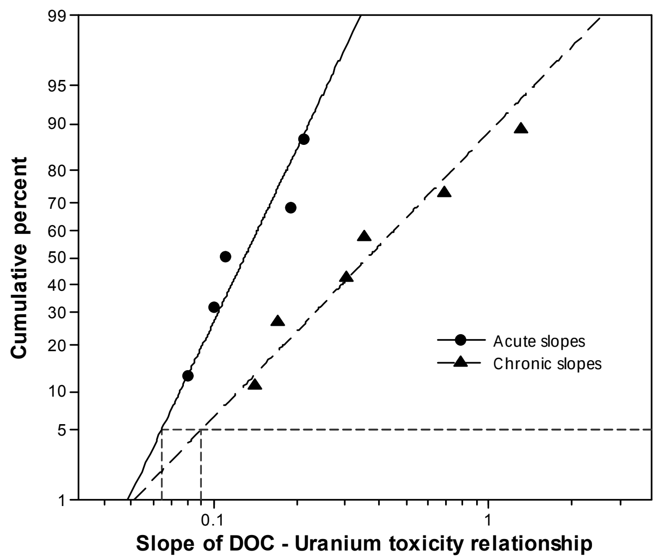

Figure 2 Log-normal cumulative probability distributions of the slopes of the relationships for uranium toxicity versus dissolved organic carbon (based on normalised IC/LC50 values for acute and chronic exposures. The short dashed lines show the intersection of the 5th percentile with the models (acute slope 5th percentile = 0.064; chronic slope 5th percentile = 0.090).

The slope factors were 0.064 for acute toxicity and 0.090 for chronic toxicity. This equates to a 6.4% and 9.0% reduction in acute and chronic U toxicity relative to the toxicity at the base DOC concentration, for every 1 mg/L increase in DOC concentration (over the DOC range 0–30 mg/L).

The chronic slope factor can be used to modify a site-specific U TV to account for changes in DOC concentrations in the receiving environment. Firstly, the relevant site-specific chronic U toxicity values used to derive the TV need to be adjusted to a standard low DOC concentration. In the case of Magela Creek, a DOC concentration of 1 mg/L can be considered an applicable estimate of a low DOC concentration.

Thus, the U toxicity values (eg IC10 or IC50) can be corrected to 1 mg/L DOC using the following equation using the following equation:

$$\text{U tox}_1 = \text{U tox}_i \times (1 + \text{slope}_{chronic}) / (1 + \text{slope}_{chronic} \times \text{DOC}_i)$$

where U tox_1 is the U toxicity value corrected to 1 mg/L DOC, U tox_i is the initial (ie original) toxicity value, DOC_i is the DOC concentration in mg/L at which U tox_i was calculated, and $\text{slope}_{chronic}$ is the slope factor for chronic toxicity (0.090). Once the (base) site-specific TV has been re-derived using the corrected toxicity data, it can then be adjusted based on the DOC concentration in the aquatic environment of interest, using the following equation:

$$\text{DOC modified trigger value (DOCMTV)} = \text{TV}_1 / (1 + \text{slope}_{chronic}) \times (1 + \text{DOC}_f \times \text{slope}_{chronic})$$

where TV_1 is the TV calculated at 1 mg/L DOC, DOC_f is the aquatic DOC concentration of interest, and $\text{slope}_{chronic}$ is the slope factor for chronic toxicity (0.090). As an example, for a U TV_1 of 2 µg/L, the DOCMTV for a surface water with a DOC concentration of 8 mg/L would be 3.2 µg/L (ie 2 / [1 + 0.09] × [1 + 8 × 0.09]).

The above DOC correction method will increase the environmental relevance of the site-specific U TV. However, given the conservative approach of adopting the 5th percentile of the distribution of the slopes as the DOC correction factor, it will not have a large effect on the U TV across the range of DOC concentrations typically measured in Magela Creek (ie ~2–8 mg/L). In contrast, it will have a substantial effect on U TVs that are adopted as post-rehabilitation and closure water quality criteria for billabongs nearer the Ranger mine, where DOC concentrations can reach 20 mg/L.

The results of this study were recently published in *Environmental Toxicology and Chemistry* (van Dam et al 2012).

Steps for completion

The next phase of work will involve updating the current site-specific U TV (which provides the basis for the current U compliance regime in Magela Creek) to incorporate (i) additional U toxicity data acquired over the past 10 years, and (ii) the above DOC correction method.

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Calculating annual solute loads in Gulungul Creek

A Mackay & WD Erskine

Background

The Supervising Scientist Division (SSD) undertakes stream height and water quality monitoring of surface waters around the Ranger uranium mine and the data are used in the assessment of potential impacts of mining. These data (water level, discharge and electrical conductivity [EC]) can also be used to calculate solute loads. Solute load data have been presented for Magela Creek (Turner et al 2012) but this is the first time that annual load data have been reported for Gulungul Creek. In recent years the potential for mine impact in the Gulungul catchment has been increasing as a result of the works associated with several lifts of the wall of the tailings storage facility (TSF) (Map 2).

The solute load data at the upstream gauging station on Gulungul Creek (GCUS) are compared with those at the downstream station (GCDS) near the Arnhem Highway (Map 2). The upstream station is located where there is no mine impact and the downstream station is located immediately below the mine. The aim of this work is to identify whether mining activities have had a measureable impact on the solute load in Gulungul Creek.

The Gulungul Creek catchment covers an area of approximately 100 km^2, originating in the Arnhem Land plateau and terminating in a backflow billabong (Gulungul Billabong) immediately upstream of where it joins Magela Creek. Current infrastructure in the catchment includes part of the TSF, mine access roads, minor tracks and three gauging stations on Gulungul Creek (Map 2). Between the upstream (catchment area 39 km^2) and downstream (catchment area 66 km^2) gauging stations there are four right bank tributaries, two left bank tributaries and numerous small floodplain channels. Of the right bank tributaries, three flow from the area affected by the Ranger mine, and one flows from Jabiru airport. The left bank tributaries flow from non-mine impacted areas. Mine-site sources need to be compared with diffuse sources from the non-mine-impacted part of the catchment located between the two stations so that the contribution from the mine alone can be quantified.

The potential contribution of mine-derived solutes to Gulungul Creek include: i) surface discharge of groundwater, originating from the TSF; ii) land-disturbance by earth works undertaken for lifts of the TSF; and/or iii) overland flow from the waste rock (primarily schist) used in the construction of the walls of the TSF. During the wet season, ERA monitors water quality from the TSF area, which, depending on quality, is either captured and redirected to the on-site water management system or is released into right bank tributaries of Gulungul Creek.

Areas of black soils, salt deposits and surface sediments with elevated EC and water soluble magnesium (Mg) and sulfate (SO_4), have also been identified along drainage lines to the west of the TSF (Hollingsworth & Klessa 2004). The nature and occurrence of these deposits may be significant because soluble or secondary minerals might store solutes, such as Mg or SO_4, that can be readily released and mobilised into waterways following rainfall (Hollingsworth & Klessa 2004).

Methods

Continuous in situ EC and stage height (m) data have been measured in Gulungul Creek at either 5- or 6-minute intervals at GCUS and GCDS since the 2005–06 wet season. Surface water grab samples have also been collected routinely at these stations since the 2001–02 wet season. These samples were analysed for a range of solutes including uranium (U), manganese (Mn), Mg and SO_4.

The method used to calculate the solute load in Gulungul Creek involved four steps. The first step comprised a QA/QC process to assure that there was close agreement between EC values measured in situ (field) or in grab samples (laboratory) and the instantaneous values measured (at the time of water sample collection) by the continuous monitoring datasonde. This was done to determine if the EC data collected during the different water sampling campaigns are directly comparable.

The second step identified the most appropriate solute that: 1) had a good correlation with the time-series EC data from the continuous monitoring program; and 2) could be used as an indicator of a mine source. In this case, as for Magela Creek, Mg was identified as the most appropriate indicator. The grab sampling data were used to establish a quantitative relationship between EC and Mg, so that the continuous time-series EC record could be used to infer an effectively continuous concentration record for Mg concentrations in Gulungul Creek.

The third step used the results from the previous two steps to calculate the annual solute loads in Gulungul Creek by combining the concentration data with discharge.

The fourth step consisted of estimating the solute contribution from the catchment (diffuse sources) in comparison with the mine-site (point source) at the downstream gauging station on Gulungul Creek.

Results

Step 1 – EC relationships

For assessment and validation purposes, a comparison was made between the EC values measured in situ (field) or in grab samples (laboratory) with the instantaneous values measured by the continuous monitoring equipment (Figure 1).

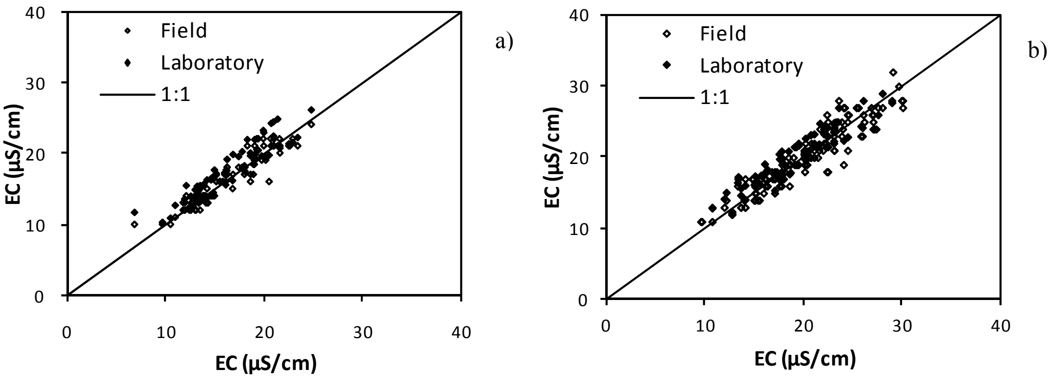

Figure 1 Relationships between electrical conductivity (EC) collected as part of the continuous monitoring program (x-axis) and EC for the field and laboratory results (y-axis) at a) GCUS and b) GCDS gauging stations on Gulungul Creek

The straight line on the plot passing through the origin represents a 1:1 relationship between results. There is a close linear relationship between both the field and laboratory values, and the continuous EC data, from GCUS (field: $R^2 = 0.88$, $\rho<0.001$; lab: $R^2 = 0.85$, $\rho<0.001$) and GCDS (field: $R^2 = 0.89$, $\rho<0.001$; lab: $R^2 = 0.86$, $\rho<0.001$) gauging stations in Gulungul Creek, indicating that the data are directly comparable, and hence confirming the well-maintained calibration of the in situ EC probes.

Step 2 – EC-Mg relationship

Magnesium sulfate ($MgSO_4$) is a major constituent of runoff and leachate produced from the weathering of waste rock and low-grade stockpiles at the mine. Magnesium sulfate has become the key indicator of mine-derived waters due to its relative contribution to the total EC measured in on-site water bodies and in Magela Creek (Turner et al 2012). Therefore, the relationship between EC and Mg concentrations was assessed for grab water samples collected through time from Gulungul Creek.

A statistically significant relationship was found between Mg and EC for GCUS ($R^2 = 0.74$, $\rho<0.001$) and GCDS ($R^2 = 0.67$, $\rho<0.01$). Therefore, the primary EC data can be used as a reliable surrogate for predicting Mg concentrations in Gulungul Creek.

Step 3 – Mg load

The predicted Mg concentration data were used to derive loads of Mg transported by Gulungul Creek during each water year between 2005–06 and 2011–12. Magnesium load was calculated using Equation 1, where t is time, i is a defined period of time, $[Mg]$ is instantaneous predicted Mg concentration (mg/L) and Q is instantaneous discharge (L/s).

$$\text{total load} = \int_{t=0}^{t=i} [Mg]\, Q\, dt \qquad (1)$$

By multiplying the Mg concentration by the corresponding discharge for each time increment and then summing over time, the total mass of Mg over a water year can be calculated. The water year is defined as the time from September in one year to August in the next year and is used so that all data for the same wet season (late October to April) are grouped in the same water year.

The estimated Mg loads and runoff for Gulungul Creek are presented in Tables 1 and 2, respectively. The results show that between 2005–06 and 2011–12, between 37 and 148% of the Mg load transported by Gulungul Creek was contributed from sources between the upstream and downstream gauging stations.

The load estimate for the 2006–07 water year has a potentially higher uncertainty than for the other years reported in Table 1 because GCDS was inundated by floodwater and the monitoring equipment was not functioning between 1and 22 March 2007. Missing stream flow data at GCDS were infilled using adjusted data from GCUS and G8210012. The missing continuous EC data were estimated using measurements from grab samples collected at the gauging stations during the period when the continuous monitoring equipment was offline. Assessment of the data interpolation and extrapolation procedures used to infill the period of missing data found that they do not introduce large errors in the derived solute load because the solute concentrations varied over a relatively small range during the post flood period (Walling 1984).

The load calculations show that the mean difference in annual Mg load between GCUS and GCDS was 21 ± 5 (SE) t/yr (Table 1). The range was 7 to 43 t/yr (Table 1), with the smallest difference recorded for the driest year (2008–09) and the greatest difference for the wettest year (2006–07) (Table 2). A close linear relationship exists between the upstream-downstream difference in Mg load and runoff (Figure 2) demonstrating that there is a strong hydrological control on Mg load in Gulungul Creek.

Table 1 Estimated Mg loads (t/yr) in Gulungul Creek

Water year	Gauging stations		Difference^	Percent difference#	leGras[1] Difference^*	leGras right-bank tribs
	GCUS	GCDS				
2005–06	32	53	21	66%	17	-
2006–07	29	72	43	148%	27	-
2007–08	19	35	16	84%	10	-
2008–09	10	17	7	70%	6	-
2009–10	17	36	19	112%	18	4.1
2010–11	37	67	30	81%	31	2.4
2011–12	30	41	11	37%	-	-
Mean	25 ± 4 (SE)	46 ± 7 (SE)	21 ± 5 (SE)	85 ± 13 % (SE)	18 ± 4 (SE)	-

1 LeGras CA (2011). *A review of research and monitoring in the Gulungul Creek catchment 1978-2011.* A report to Energy Resources of Australia.

^ Difference is calculated by subtracting the GCUS load from the GCDS load. # Percent difference is calculated by dividing the difference by the GCUS load. * Estimated mine-related loads for the calendar year not the water year.

Table 2 Measured runoff (GL/yr) in Gulungul Creek upstream and downstream of the Ranger mine

Water year	GCUS	GCDS
2005–06	49	72
2006–07	52	120
2007–08	28	47
2008–09	11	19
2009–10	21	36
2010–11	52	86
2011–12	41	50
Mean	36 ± 6 (SE)	61 ± 13 (SE)

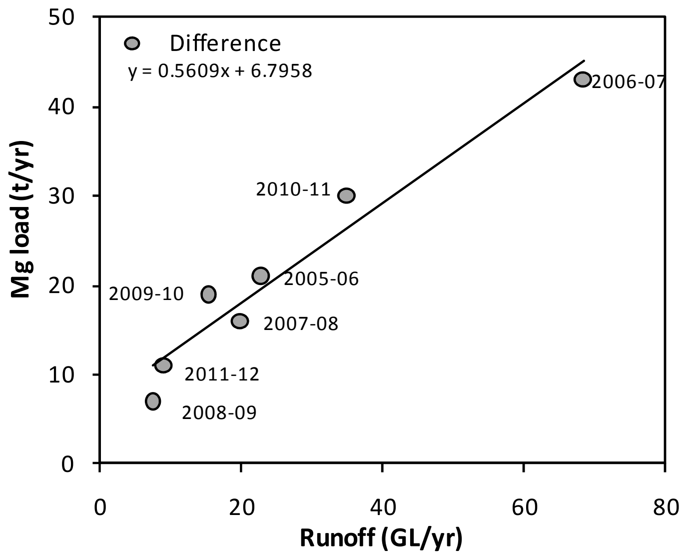

Figure 2 Relationship between difference in annual Mg load and difference in annual runoff between the upstream (GCUS) and downstream (GCDS) gauging stations on Gulungul Creek ($R^2 = 0.94$, $\rho< 0.001$)

Step 4 – Significance of the Ranger mine as a potential solute source to Gulungul Creek

If mining activities were not present in the Gulungul Creek catchment then it could reasonably be assumed that there would be very similar annual Mg load-runoff relationships for the upstream and downstream gauging stations, given the similarities between the (non-mine) contributing catchments. In fact, the slopes of the annual Mg load-runoff relationship for the upstream ($R^2 = 0.92$, $\rho < 0.001$, slope = 0.567) and downstream ($R^2 = 0.93$, $\rho < 0.001$, slope = 0.548) gauging stations (Figure 3b) are essentially the same, which supports the assumption. However, the line for the downstream station is vertically displaced from the upstream one, the difference being indicative of the mine-specific contribution.

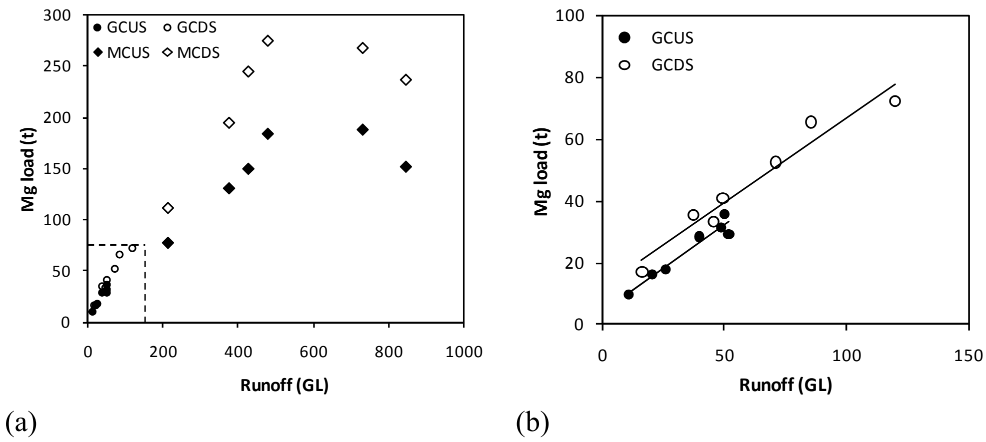

Figure 3 Estimated magnesium loads (t) and annual runoff (GL) for the a) upstream (GCUS; MCUS) and downstream (GCDS; MCDS) monitoring stations on Gulungul Creek and Magela Creek, respectively, and b) expansion of panel (a) insert for GCUS and GCDS

Therefore, the approximate annual mine contribution of Mg load to Gulungul Creek between 2005–06 and 2011–12 is the difference between the two regression lines in Figure 3b. This annual mine contribution equals, on average, 7.8 t/yr of the 21 t/yr difference between the two Gulungul stations. Therefore, approximately 37% of the Mg load between the two stations is derived from the mine and 63% from diffuse sources. In the context of the Ranger minesite (Figure 3a), between 2005–06 and 2010–11 an average of 74 t of Mg was contributed annually by the minesite to Magela Creek (Turner et al 2012). Therefore, the contribution of Mg from the minesite to Gulungul Creek equates to around one-tenth of that contributed by the mine to the Magela Creek system.

Discussion

Previous estimates of Mg loads made by LeGras (2011), including loads released from the TSF area into the right-bank tributaries, are included for comparison in Table 1. LeGras (2011) estimated load by calculating mean Mg concentrations from weekly grab samples taken over a small portion of the period of annual flow, and did not use the continuous time-series data for EC and discharge, as used here. Instead, LeGras (2011) estimated discharge for each calendar year between 2004 and 2011 by: i) predicting discharge at G8210012 (Map 2) from a regression relationship between rainfall at Jabiru airport and discharge; and then ii) using catchment area ratios to estimate the discharge at the upstream and downstream monitoring sites on Gulungul Creek. These data were then multiplied by the mean (based on weekly grab samples) Mg concentration to derive the annual Mg load in Gulungul Creek.

Similarly, LeGras (2011) multiplied the mean Mg concentration for each swale drain along the west and south west base of the TSF by the total estimated discharge, to determine Mg load transported from the TSF area into the right-bank, mine-side tributaries of Gulungul Creek. Despite using a less rigorous method than applied here, LeGras (2011) found that the difference in mean annual Mg load between the upstream and downstream stations on Gulungul Creek was essentially the same as reported here (Table 1). However, the present work greatly extends LeGras (2011) by being able to more definitively estimate the extent of Mg contribution from the mine into the creek.

Conclusions

The above results suggest that over the past seven water years between 37% and 148% of the Mg load transported by Gulungul Creek has been contributed by sources between the upstream and downstream monitoring stations. It has only been since the commencement of the 2009–10 wet season that the Mg loads entering Gulungul Creek from the TSF area have been able to be estimated directly (Table 1) using the water quality data and discharges measured by ERA in the runoff collection system installed along the base of the TSF. These values are one order of magnitude less than the difference in mean annual Mg load between the upstream and downstream stations, and are consistent with the values derived here from detailed analysis of the continuous monitoring data.

The results presented here indicate that mining-related activities between 2005 and 2012 have increased the Mg load in Gulungul Creek by an average of 7.8 t/yr. This is one order of magnitude less than the average annual load contributed to Magela Creek between 2005 and 2011. However, diffuse sources between the upstream and downstream monitoring stations on Gulungul Creek also contribute on average 13.2 t/yr of Mg. The analysis to date suggests that the mine supplies 37% and diffuse sources 63% of the difference in Mg load between the upstream and downstream stations on Gulungul Creek.

However, further work is needed to provide complete verification and to more clearly distinguish the mining and non-mining sources and contributions, as outlined in Section 1. The issue of the contribution of Mg from diffuse sources will be investigated in more detail by collating and analysing the available concentration data for Mg in the sub-catchments of Gulungul Creek between the upstream and downstream gauging stations with further assessment using the continuous monitoring data. The propagated level of uncertainty for the Mg loads for Gulungul Creek will also be quantified as part of this subsequent work.

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Atmospheric radioactivity monitoring in the vicinity of Ranger and Jabiluka

C Doering, R Cahill, L Da Costa, J Pfitzner & A Bollhöfer

Introduction

Uranium mining has the potential to release radon (a radioactive gas) and particulate-bound radionuclides to the atmosphere at levels above the natural background through ground disturbance and other activities. The inhalation of radon progeny in air and long-lived alpha activity (LLAA) radionuclides contained in or on dust can contribute to the radiation dose received by the public in the vicinity of a uranium mine.

The Ranger uranium mine (RUM) in northern Australia is a planned exposure situation in the context of current recommendations of the International Commission on Radiological Protection (ICRP) (ICRP 2007). The ICRP recommended dose limit for the public for planned exposure situations is 1 mSv in a year. This limit applies to the sum of the above background doses received by the public from all sources and exposure pathways.

In addition to dose limitation, the ICRP recommends that the level of protection should be optimised so that the likelihood of incurring exposures, the number of people exposed and the magnitude of individual doses are kept as low as reasonably achievable, taking into account economic and societal factors (ICRP 2007). The concept of dose constraint is used in the optimisation process to provide an upper bound on the annual doses that people should receive from an individual source or practice. For planned situations involving public exposure, the ICRP recommends that the dose constraint should be less than 1 mSv in a year and a value of no more than about 0.3 mSv in a year would be appropriate (ICRP 2007).

The main areas of permanent habitation in the vicinity of RUM are Jabiru town and the Mudginberri community. ***eriss*** maintains atmospheric monitoring stations to measure radon progeny and dust-bound LLAA radionuclide concentrations at the Jabiru Water Tower (Jabiru town) and Four Gates Road (Mudginberri) (Map 3). The purpose of these measurements is twofold: (i) to provide an independent check of the values measured and reported by the RUM operator; and (ii) to provide assurance that any dose to the public associated with mine-related radioactivity in air is low and does not pose any unacceptable health risk.

Methods

Environmental radon daughter monitors from Radiation Detection Systems in Adelaide were used for continuous monitoring of the potential alpha energy concentration (PAEC) of radon progeny in air. The monitors operated at a flow rate of 0.30–0.35 l/min drawing air through a Whatman GF/C filter positioned above an alpha counter. Hourly PAEC data was logged in the internal memory of the monitors, which were downloaded at approximately fortnightly intervals.

EcoTech MicroVol-1100 low flow-rate (~3 l/min) air samplers fitted with Whatman GF/C filters were used for dust sampling. Filters were changed at approximately fortnightly intervals and analysed in ***eriss*** laboratories for total alpha activity using Daybreak 582 alpha counters. Count times were typically three to four days to ensure reasonable counting statistics were achieved. Measurement of the background alpha activity of the counting system was made prior to analysis of each filter. The background count rate was subtracted from the filter count rate to determine the net count rate. A correction factor for counter efficiency was then applied to determine the alpha activity on the filter.

Results

Radon progeny

Figures 1 and 2 show hourly radon progeny PAEC data for the period January 2011 to September 2012 from the ***eriss*** atmospheric monitoring stations at Jabiru Water Tower and Four Gates Roads, respectively. Gaps in the data are due to instrument maintenance and data quality issues. Quarterly average PAEC values have been calculated from the hourly data and are also shown in the figures.

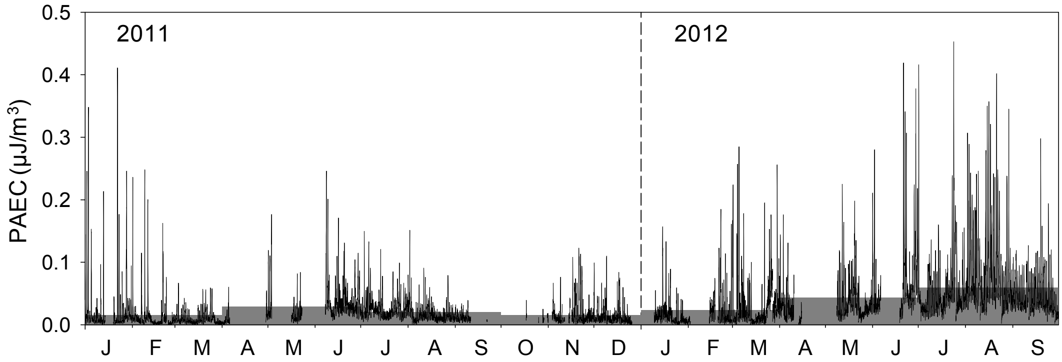

Figure 1 Hourly (black line) and quarterly average (grey columns) radon progeny PAEC in air at Jabiru Water Tower (Jabiru town)

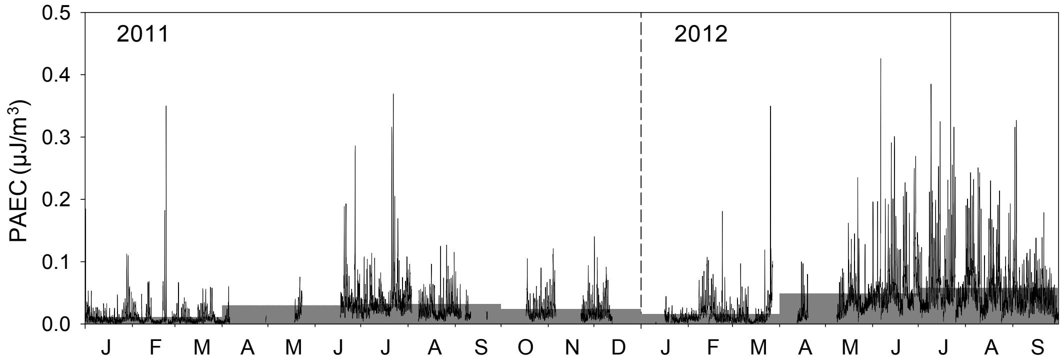

Figure 2 Hourly (black line) and quarterly average (grey columns) radon progeny PAEC in air at Four Gates Road (Mudginberri)

The spikiness in the hourly PAEC data reflects the normal diurnal pattern in radon progeny in surface air. Higher concentrations typically occur in the early morning around sunrise when atmospheric conditions tend to be most stable, causing a build up of radon progeny in the surface air. Thereafter the surface air becomes mixed by convection (solar heating) and advection (wind), which disperses the radon progeny into a larger atmospheric volume.

The quarterly average PAEC results shown in Figures 1 and 2 show the typical wet-dry seasonal trend, with higher concentrations occurring in the second and third quarter of the year (dry season) and lower concentrations occurring in the first and fourth quarter of the year (wet season). The effect of rainfall is to suppress radon exhalation from the soil surface and thus decrease the radon progeny PAEC in air.

Table 1 gives the annual average radon progeny PAEC in air and the estimated total and mine-related public doses for 2011 for both Jabiru town and Mudginberri. Annual average radon progeny PAEC and public dose estimates for 2012 have not yet been determined, as measurements for this year are still ongoing.

Table 1 Annual average radon progeny PAEC in air and public dose estimates for 2011*

	Jabiru town	Mudginberri
Annual average PAEC (µJ/m^3)	0.019 (0.045)	0.022
Total dose (mSv)	0.179 (0.434)	0.210
Mine-related dose (mSv)	0.021 (0.065)	0.003

* Values in parentheses are those reported by the mine operator in its *Radiation Protection and Atmospheric Monitoring Program Report for the Year Ending 31 December 2011*.

The total dose from radon progeny in air includes contributions from both natural background and the mine and was calculated as:

$$E_{RP\text{-}TOTAL} = PAEC_{RP\text{-}TOTAL} \times DCC_{RP\text{-}PUBLIC} \times t_{OCC}$$

where:

$E_{RP\text{-}TOTAL}$ is the total annual effective dose from the inhalation of radon progeny in air;

$PAEC_{RP\text{-}TOTAL}$ is the annual average radon progeny PAEC;

$DCC_{RP\text{-}PUBLIC}$ is the current ICRP recommended dose conversion coefficient for radon progeny for the public of 0.0011 mSv per µJh/m^3 (ICRP 1993); and

t_{OCC} is the occupancy time, which is assumed to be 8760 hours.

The mine-related dose from radon progeny in air was determined using a simple wind correlation model. It was assumed that the measured radon progeny PAEC at Jabiru town included both a mine-related and natural background component when the wind was from the 90°–110° sector. When the wind was from other directions or when there was no wind blowing (ie still conditions), it was assumed that the measured radon progeny in air was due to natural background only. The same assumptions were also made for Mudginberri when the wind was from the 140°–160° sector. Hourly wind direction data for 2011 was acquired from the Bureau of Meteorology weather station at Jabiru airport. Analysis of this data (Figure 3) indicated that the wind was from the direction of the mine for 2090 hours at Jabiru town and 410 hours at Mudginberri. By correlating the hourly wind direction data with the hourly radon progeny data, the average PAEC when the wind was from the direction of the mine and when the wind was not from the direction of the mine was determined. The mine-related dose to the public from radon progeny was then calculated as:

$$E_{RP\text{-}MINE} = (PAEC_{RP\text{-}MINE} - PAEC_{RP\text{-}OTHER}) \times DCC_{RP\text{-}PUBLIC} \times t_{MINE}$$

where:

$E_{RP\text{-}MINE}$ is the annual effective dose from the inhalation of mine-related radon progeny in air;

$PAEC_{RP\text{-}MINE}$ is the annual average radon progeny PAEC when the wind was from the direction of the mine;

$PAEC_{RP\text{-}OTHER}$ is the annual average radon progeny PAEC when the wind was from other directions;

$DCC_{RP\text{-}PUBLIC}$ is the current ICRP recommended dose conversion coefficient for radon progeny for the public of 0.0011 mSv per µJh/m³ (ICRP 1993); and

t_{MINE} is the number of hours that the wind was from the direction of the mine.

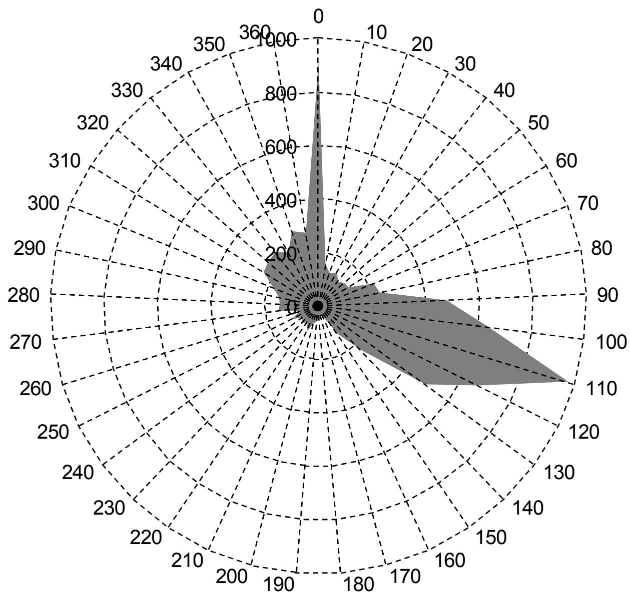

Figure 3 Histogram of hourly wind direction data at Jabiru airport for 2011. A wind direction of 0 indicates still conditions (ie no wind).

Dust-bound LLAA radionuclides

Figures 4 and 5 show measured concentrations of dust-bound LLAA radionuclides in air for the period January 2011 to September 2012 from the ***eriss*** atmospheric monitoring stations at Jabiru Water Tower and Four Gates Road, respectively. Gaps in the data are due to instrument maintenance and data quality issues. Time weighted quarterly average concentrations have been calculated from the individual measurements and are also shown in the figures.

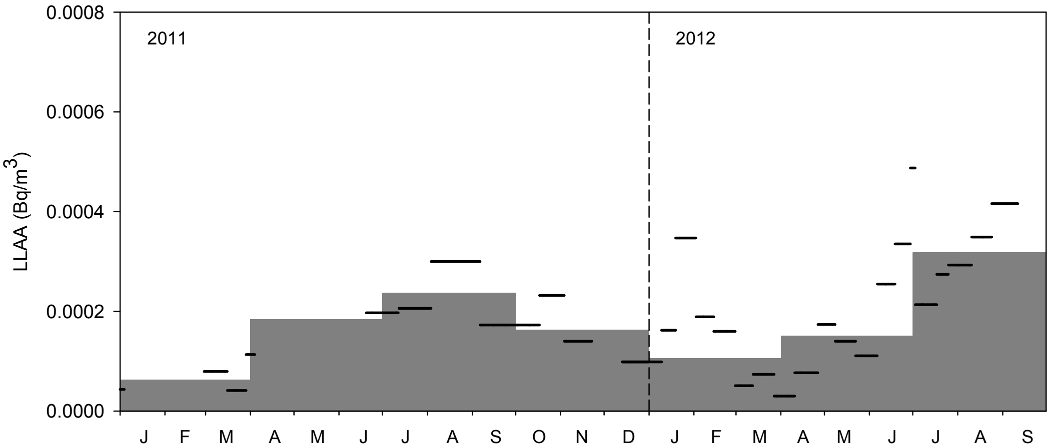

Figure 4 Measured (black lines) and time weighted quarterly average (grey columns) LLAA radionuclide concentrations at Jabiru Water Tower (Jabiru town)

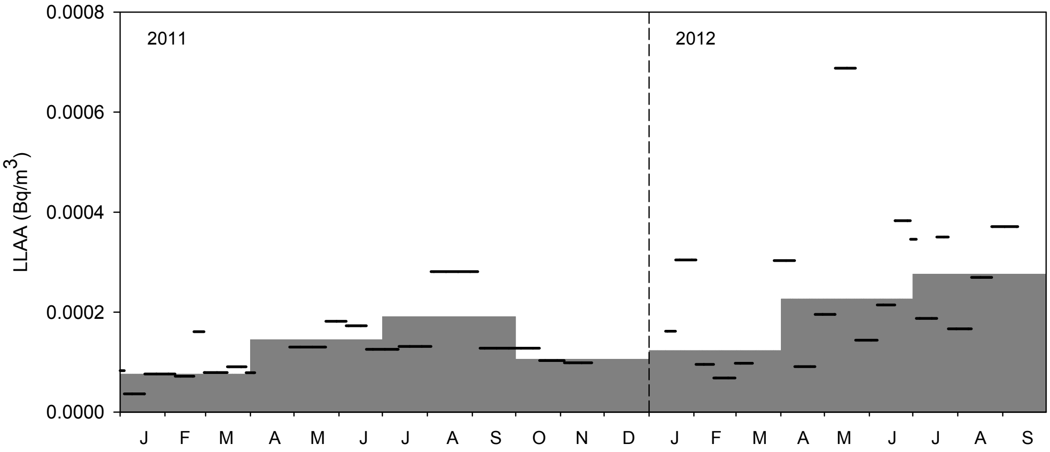

Figure 5 Measured (black lines) and time weighted quarterly average (grey columns) LLAA radionuclide concentrations at Four Gates Road (Mudginberri)

The general trend is for LLAA radionuclide concentrations to be higher in the second and third quarter of the year (dry season) and lower in the first and fourth quarter of the year (wet season). This is due to rainfall suppression of dust generation during the wet season.

Table 2 gives the annual average LLAA radionuclide concentration and the estimated total and mine-related public doses from these radionuclides in dust for 2011 for both Jabiru town and Mudginberri. Annual average LLAA radionuclide concentration and public dose estimates for 2012 have not yet been determined, as measurements for this year are still ongoing.

Table 2 Annual average LLAA radionuclide concentrations and public dust dose estimates for 2011

	Jabiru town	Mudginberri
Annual average LLAA (mBq/m^3)	0.184	0.130
Total dose (mSv)	0.007	0.005
Mine-related dose* (mSv)	8×10^{-4}	7×10^{-5}

*Calculated from the assumption that the ratio of mine-related to total annual dose from dust is the same as that for radon progeny.

The total dose from dust-bound LLAA radionuclides includes contributions from both natural background and the mine and was calculated on a time and volume weighted basis as:

$$E_{LLAA\text{-}TOTAL} = [\Sigma(V_{ACT}/V_{EXP} \times t_S \times LLAA) / \Sigma(V_{ACT}/V_{EXP} \times t_S)] \times DCC_{LLAA\text{-}PUBLIC} \times BR \times t_{OCC}$$

where:

$E_{LLAA\text{-}TOTAL}$ is the annual effective dose from the inhalation of dust-bound LLAA radionuclides;

V_{ACT} is the actual sample volume for an individual dust sample;

V_{EXP} is the expected sample volume for the individual dust sample based on a sampler flow rate of 3 l/min and the sampling time;

t_S is the sampling time;

LLAA is the measured LLAA radionuclide concentration in the individual dust sample;

$DCC_{LLAA\text{-}PUBLIC}$ is the dose conversion coefficient for dust-bound LLAA radionuclides in air for the public of 0.0061 mSv/Bq_α, which was derived from individual radionuclide dose conversion coefficients given in ICRP (1996);

BR is breathing rate, assumed to be 0.75 m^3/h; and

t_{OCC} is the occupancy time, assumed to be 8760 hours.

The mine-related dose from dust-bound LLAA radionuclides has been estimated by assuming that the ratio of mine-related to total annual dose from dust is the same as that for radon progeny and was calculated as:

$$E_{LLAA\text{-}MINE} = (E_{RP\text{-}MINE}/E_{RP\text{-}TOTAL}) \times E_{LLAA\text{-}TOTAL}$$

where:

$E_{LLAA\text{-}MINE}$ is the annual effective dose from the inhalation of mine-related dust-bound LLAA radionuclides;

$E_{RP\text{-}MINE}$ is the annual effective dose from the inhalation of mine-related radon progeny in air;

$E_{RP\text{-}TOTAL}$ is the total annual effective dose from the inhalation of radon progeny in air; and

$E_{LLAA\text{-}TOTAL}$ is the annual effective dose from the inhalation of dust-bound LLAA radionuclides.

The assumption of equivalent mine-related to total dose ratios for both radon progeny and dust-bound LLAA radionuclides is likely to result in an overestimate of the mine-related dose from dust. This is because dust in air should settle out much quicker as a function of distance from the mine compared with gaseous radon, meaning that the mine-related to total dose ratio for dust should be less than that for radon progeny.

Conclusions

The ***eriss*** atmospheric radioactivity monitoring results indicate that in 2011 the total effective dose to the public from the inhalation of radon progeny in air was 0.179 mSv at Jabiru town and 0.210 mSv at Mudginberri. The mine-related component of this dose, however, was only 0.021 mSv at Jabiru town and 0.003 mSv at Mudginberri. The results also indicate that the total effective dose to the public from the inhalation of dust-bound LLAA radionuclides was 0.007 mSv at Jabiru town and 0.005 mSv at Mudginberri, with a mine-related component of less than 0.001 mSv at both locations. In the context of ICRP recommended dose limits and dose constraints for public exposure in planned situations, these mine-related inhalation doses are low and do not present any unacceptable health risk to persons living in the vicinity of RUM.

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Monitoring of radionuclides in groundwater at Ranger

A Bollhöfer & P Medley

Introduction

Groundwater quality at Ranger uranium mine has been monitored by ERA, by the Northern Territory Government's Department of Mines and Energy (DME) and by ***eriss*** for the past three decades (Martin & Akber 1996, 1999, Klessa 2001). Bores were routinely sampled and analysed by ***eriss*** between 1996 and 2003, when the decision was made that radionuclides will be analysed in aliquots of bore water routinely sampled by DME and sampling by ***eriss*** was discontinued. Groundwater quality parameters measured include major ions (ERA/DME), heavy metals (ERA/DME/***eriss***) and radionuclides (ERA/***eriss***). These data are currently stored in various individual databases and formats. A meeting led by ERA was held in November 2011 to discuss a joint organisational approach, as part of an effort to improve the Ranger groundwater knowledge base, facilitate a more coordinated approach to the acquisition and storage of groundwater data and to progress the development of closure criteria for Ranger. As an outcome of the meeting, ***eriss*** radionuclide data from bores sampled by DME up to 2009 were sent to ERA for inclusion in their comprehensive groundwater GIS.

The focus of ***eriss*** groundwater monitoring and research projects has been the measurement of groundwater radionuclide activity concentrations in various bores around the Ranger mine with some time records going back to the early 1980s (Martin & Akber 1996, 1999). All groundwater data available to ***eriss***, which include ^{226}Ra and uranium isotopes and metals, have been quality checked and verified and imported into a single EnRad groundwater database. The data have also been migrated into the new ***eriss*** *EnviroSys* database.

Method

Groundwater samples are collected by DME on an annual basis, and aliquots are sent to ***eriss*** for radionuclide analyses. All samples received in 2010 and 2011 have now been analysed. 2012 samples collected by DME in both March and September have been received for analysis. It is anticipated that this analysis will be completed by Q1 2013.

The target radionuclides (^{226}Ra and ^{234,238}U) are radiochemically separated from the bulk water samples, and measured via alpha spectrometry. Radiochemical methods for the separation of uranium and radium isotopes are published in Martin and Hancock (2004) and Medley et al (2005). As thorium and lead are particle reactive and are 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. The ^{234}U/^{238}U activity ratio in particular may be a useful tracer to discriminate natural from mine-derived sources of uranium in groundwater (Ryan & Bollhöfer 2007).

Metal concentrations in aliquots of the groundwater samples are determined via a combination of ICP-MS and ICP-OES methods.

Results

Figure 1 shows a vertical aerial photograph of the Ranger mine and the locations of bores sampled by DME since 2002 (note: not all of the bores are sampled every year). Highlighted are RN22937 (OB23), RN22902 (OB1A), RN9329, RN22934 (OB20) and RN22935 (OB21A) (OB numbers are the historic observation bore numbers). The bores are located around the tailings storage facility (TSF) with screen depths of 36–51 m (RN22937), 16–31 m (RN22902), 17–19 m (RN9329), 21–36 m (RN22934) and 31–43m (RN22935).

Figure 1 Aerial photograph of Ranger mine with monitoring bore locations sampled by DME

Figure 2 shows ^{226}Ra and ^{238}U activity concentrations measured in filtered water from these bores.

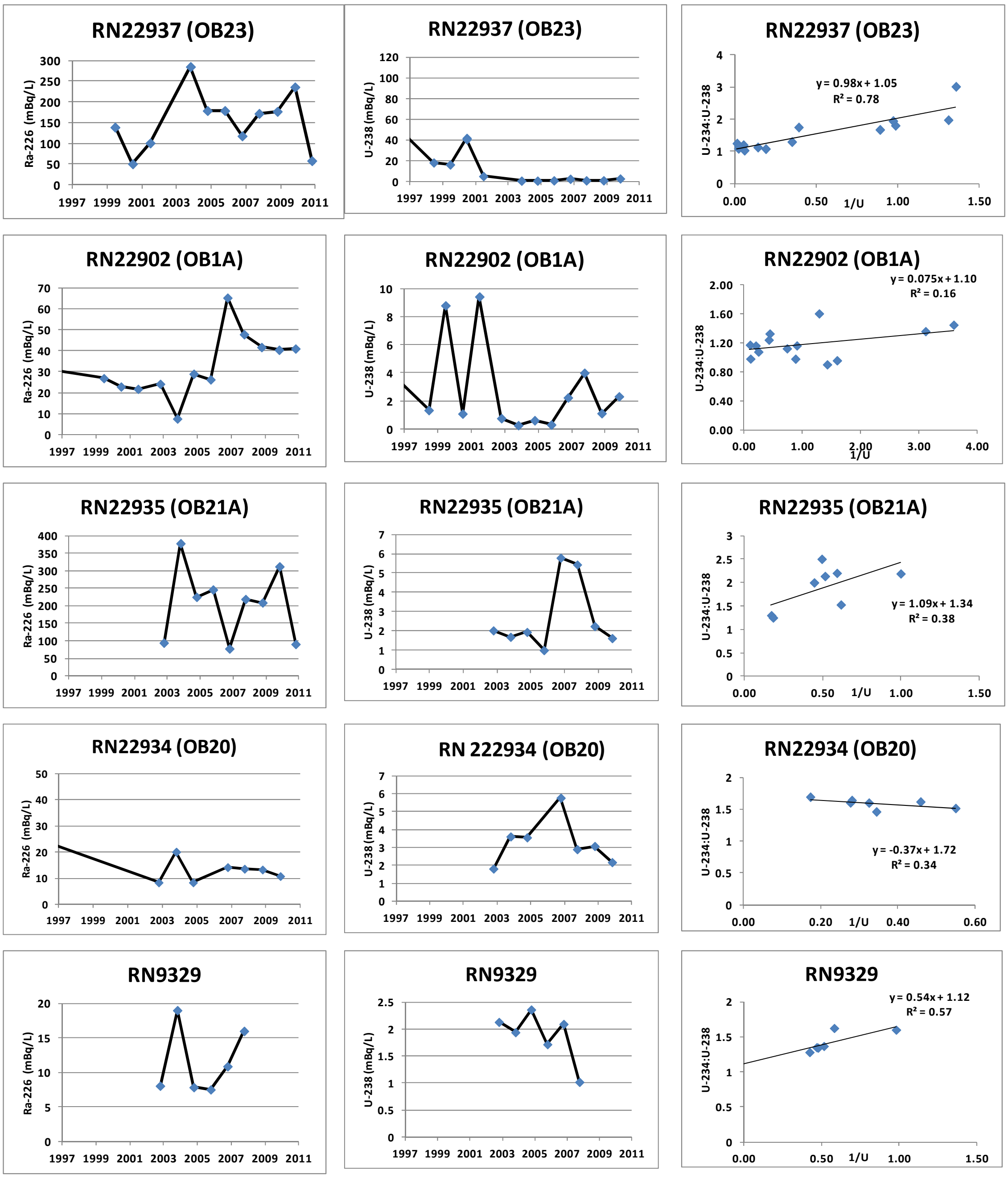

Figure 2 ^{226}Ra and ^{238}U activity concentrations measured in filtered water from bores located around the TSF and inverse concentration plots for the $^{234}U/^{238}U$ activity ratio

A plot of the $^{234}U/^{238}U$ activity ratios against the inverse of the ^{238}U activity concentration is shown in Figure 2 as well. In this type of plot, the intercept of the regression line with the y-axis gives an indication of the $^{234}U/^{238}U$ activity ratio of a single contaminating uranium source (the 'contaminating end-member') in a system that has a constant background ^{238}U activity concentration. Due to the recoil effect of ^{234}U and its preferential leaching over the ^{238}U isotope in natural groundwaters, most natural groundwaters have a $^{234}U/^{238}U$ activity ratio greater than 1 (Ivanovich & Harmon 1982). In contrast, it is generally assumed that uranium in liquid waste generated by the processing of uranium ore shows $^{234}U/^{238}U$ activity ratios of ~1, as the strong acid used in the processing of ores will effectively dissolve all of the uranium without preferentially leaching one of the isotopes (Zielinski et al 1997).

Generally, the five bores selected show activity concentrations of ^{226}Ra larger than ^{238}U. RN22937 to the north of the TSF shows a statistically significant ($p<<0.05$) decrease of the $^{234}U/^{238}U$ activity ratio when plotted against the inverse of the ^{238}U activity, with a contaminating end-member $^{234}U/^{238}U$ activity ratio of 1.07±0.10. Whereas the ^{238}U activity concentration has been decreasing over the past decade, the ^{226}Ra activity concentration is variable with activity concentrations between 50 and 285 mBq/L.

RN22902 to the northwest of the TSF wall showed essentially no trend in ^{226}Ra or ^{238}U activity concentrations until 2005 but there has been an increase in both radionuclide activity concentrations in 2007. The contaminating end-member $^{234}U/^{238}U$ activity ratio is close to 1 (1.10±0.07), but the correlation between the $^{234}U/^{238}U$ activity ratio and the inverse of the ^{238}U activity concentration is not significant ($p=0.15$).

RN22934 and RN22935 are located to the south of the TSF and exhibited maximum ^{226}Ra activity concentrations in September 2004, whereas ^{238}U peaked around 2007. ^{226}Ra activity concentrations have varied from 8-20 mBq/L in RN22934 and from 78-380 mBq/L in RN22935 over the past decade. There is no statistically significant correlation between the $^{234}U/^{238}U$ activity ratio and the inverse of the ^{238}U activity concentration for RN22934 ($p=0.17$) or RN22935 ($p=0.11$), and both intercepts with the y-axis are significantly different from 1.

RN9329 to the west of the TSF in the Gulungul Creek catchment has shown an increase in ^{226}Ra but a decrease in ^{238}U activity concentrations between 2004 and 2008. The inverse concentration plot shows a decrease of the $^{234}U/^{238}U$ activity ratio with increasing uranium activity concentrations ($p=0.08$). The contaminating end-member $^{234}U/^{238}U$ activity ratio is 1.12±0.14.

Discussion

There are two distinctively different sources of groundwater solutes: primary and secondary sources. Primary sources for example are tailings seepage, land irrigation or runoff from on-site structures. Secondary sources are solutes remobilised due to groundwater–rock interaction caused by pH and redox reactions, desorption and ion-exchange processes and dissolution of minerals. Secondary processes can have a significant effect on solute concentrations in groundwater and Martin and Akber (1999) have shown that desorption of ^{226}Ra rather than direct transport is the most important factor controlling ^{226}Ra activity concentration in groundwaters around the TSF. Consequently, ^{226}Ra is not a good indicator of seepage from the TSF. Uranium has a retardation factor 5-10 times lower than radium (Kalf & Dudgeon 1999) and if seepage of uranium from the TSF was occurring one would expect a $^{234}U/^{238}U$ end-member activity ratio of ~1 in the seepage affected bores. For example OB13A immediately north of the TSF has shown an increase in ^{238}U activity concentrations from about 45 mBq/L in the 1980s to above 10 Bq/L in the 1990s coupled with a decrease in $^{234}U/^{238}U$ activity ratios to values close to 1 (Iles et al 2002).

Contaminant plumes exist to the north of the TSF, along the Coonjimba fault line, and south-west of the TSF, as shown by solute transport modelling and analysis of available ERA data (Puhalovich et al 2012a,b). The elevated ^{226}Ra activity concentrations measured in RN22937 could be a result of ^{226}Ra competing for cation adsorption sites in the host rock caused by an increase in groundwater salinity. Construction of stockpiles and the trial landform may have been additional factors influencing the ^{226}Ra and ^{238}U activity concentrations in this bore.

However, uranium and sulfate (ERA 2011) concentrations in this bore are low at present and direct seepage of uranium from the TSF is unlikely, despite the inverse concentration plot pointing to a contaminating end-member with a $^{234}U/^{238}U$ activity ratio of ~1 in the 1990s to early 2000s.

Bore RN22902 has shown low but variable ^{238}U activity concentrations over the past decade. Increases observed from 2007 (^{226}Ra and ^{238}U) may be the result of several lifts of the TSF walls during that time, and runoff and seepage from fresh waste rock used as a building material for the lift. There is no correlation of the $^{234}U/^{238}U$ activity ratio with the inverse ^{238}U activity concentration indicating that there is not one single source responsible for the variability of uranium concentration observed in this bore.

RN22934 and RN22935 are located to the south of the tailings dam. ^{226}Ra activity concentrations in RN22935 are on average 20 times higher than in RN22934, most likely a result of different groundwater chemistry in the two bores, with higher values of EC, calcium, magnesium and sulfate concentrations observed in RN22935 in 2010 (ERA 2011). The low ^{238}U activity concentrations and sulfate levels in these bores (ERA 2011) and y-intercepts that are significantly different from 1 suggest that there has been little, if any, seepage from the tailings dam. This is also the case for RN9329 located in the Gulungul catchment. ^{226}Ra and ^{238}U activity concentrations and sulfate levels (ERA 2011) in this bore are both low. Uranium with a $^{234}U/^{238}U$ activity ratio of ~1 may originate from leaching of uranium from acid sulfate soils in the area and subsequent lateral transport of this seepage.

Steps for completion

The routine measurement of ^{226}Ra, ^{238}U and ^{234}U activity concentrations in monitoring bores sampled by DME will continue. The data will be added to the SSD databases and an internal report summarising the groundwater data is planned for Q2 2013.The radionuclide data will also be added to the overarching spatial groundwater GIS for the site that has been developed by ERA.

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Results of the stream monitoring program in Magela Creek and Gulungul Creek catchments, 2011–12

CL Humphrey, A Bollhöfer & DR Jones

Progress under this KKN for the stream monitoring program in the Magela Creek and Gulungul Creek catchments is reported by way of (i) results of the routine monitoring program conducted for the 2011–12 period, and (ii) monitoring support tasks for the same period, including research and development, reviews and reporting. The latter tasks are reported separately (ARRTC paper KKN 1.3.1 Ranger stream monitoring research).

Since 2001, routine water quality monitoring and ecotoxicity programs have been deployed by the 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 of 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 the SSD that meet these requirements are:

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

- *Water physico-chemistry*:
  - Continuous monitoring: through the use of multi-probe data sondes and data loggers, continuous measurement of pH, electrical conductivity (EC), turbidity and temperature in Magela Creek, and EC, turbidity and temperature in Gulungul Creek;
  - Event-based automatic sampling: The upstream and downstream monitoring sites in both Magela and Gulungul Creeks are equipped with auto-samplers, programmed to collect a 1 L water sample in response to the occurrence of pre-specified EC or turbidity conditions. The samples are analysed for total concentrations of uranium, magnesium, calcium, manganese and sulphate.
  - Ongoing quality control sampling: Routine site visits for spot *in situ* measurement of pH, EC, turbidity and temperature (fortnightly), periodic grab sampling for measurement of uranium, magnesium, calcium, manganese and sulfate (monthly) and radium (samples collected fortnightly but combined to make monthly composites).
- *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 data sondes 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 2011–12 are summarised below.

Chemical and physical monitoring of Magela Creek

A Frostick, K Turner, S Fagan & WD Erskine

Background

From 2010–11, SSD's weekly grab sampling program was replaced by continuous monitoring of stream height, electrical conductivity (EC), turbidity and water temperature coupled with event-triggered automatic pump sampling as the primary water quality monitoring method (Turner & Jones 2010). Due to the time taken to retrieve samples from the field (typical between 2 and 12 hours), automatically collected samples are analysed for total metals rather than filtered, ensuring a conservative measurement of key analytes. The change in method has substantially enhanced SSD's ability to detect early changes in water quality and to carry out any subsequent management or investigative activities as early as possible. A comprehensive quality control program is carried out to ensure high quality of the continuous data and the results of chemical analyses. Every two weeks in situ spot checks measure physicochemical parameters (pH, EC, turbidity) and every four weeks grab samples are collected for chemical analysis. In addition to continuous monitoring, manual grab samples are taken every two weeks from Magela Creek for radium analysis. Map 2 shows the location of the upstream (MGUGT) and downstream (MCDW) monitoring stations and key Ranger mine features.

Results

Flow commenced at the Magela Creek upstream and downstream monitoring stations on 23 November 2011. Flow remained low until increased rainfall in December, which resulted in several peaks in turbidity at both stations, which is typical for first flush conditions (Figure 1).

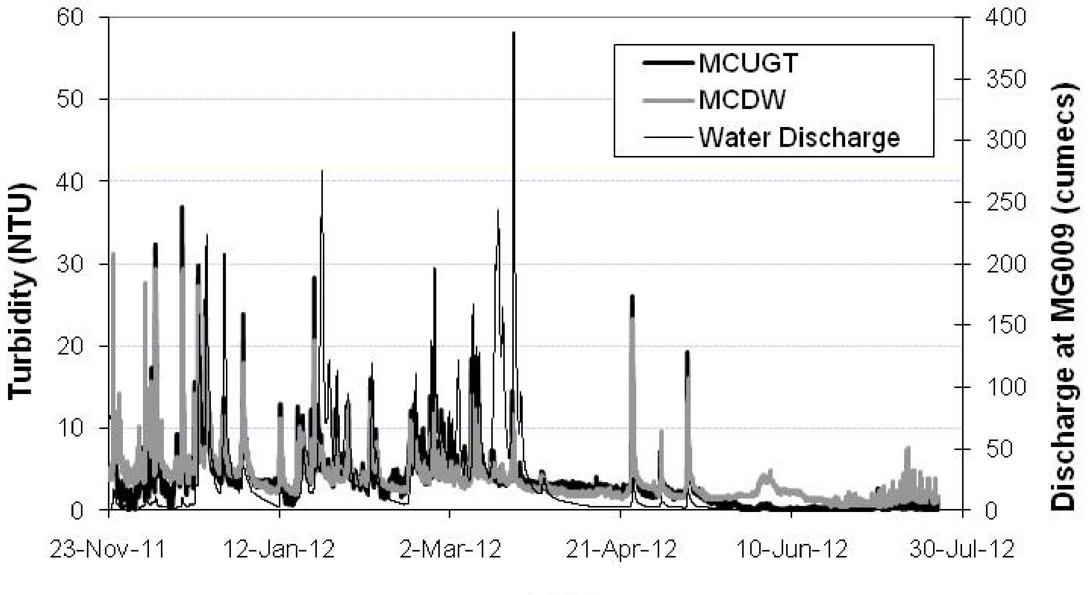

Figure 1 Continuous turbidity and discharge in Magela Creek between November 2011 and July 2012

A peak in EC occurred at the downstream monitoring site on 4 December 2011 (Figure 2). The peak duration was 7 hours and 10 minutes and it reached a maximum EC of 54 µS/cm. The source of the contamination was identified as solute-laden groundwater expressing from a creek line within the Djalkmara Land Application Area. This was exacerbated by low flow conditions, and hence low dilution capacity, within Magela Creek.

As is typical in Magela Creek, EC was intermittently elevated to levels as high as 42 µS/cm and decreased during periods of increased flow from mid to late December 2011. Low rainfall during early January 2012 resulted in low flow conditions, with the exception of a rainfall event on 12 January 2012 (26.6 mm of rain was recorded at Jabiru Airport). During this rainfall event the EC at MCDW reached 23 µS/cm prior to and 22 µS/cm following the event peak. A 38.2 mm rainfall event (recorded at Jabiru Airport) on 18 January 2012 resulted in a peak in EC of 35.8 µS/cm at MCDW associated with the rising limb of the hydrograph.

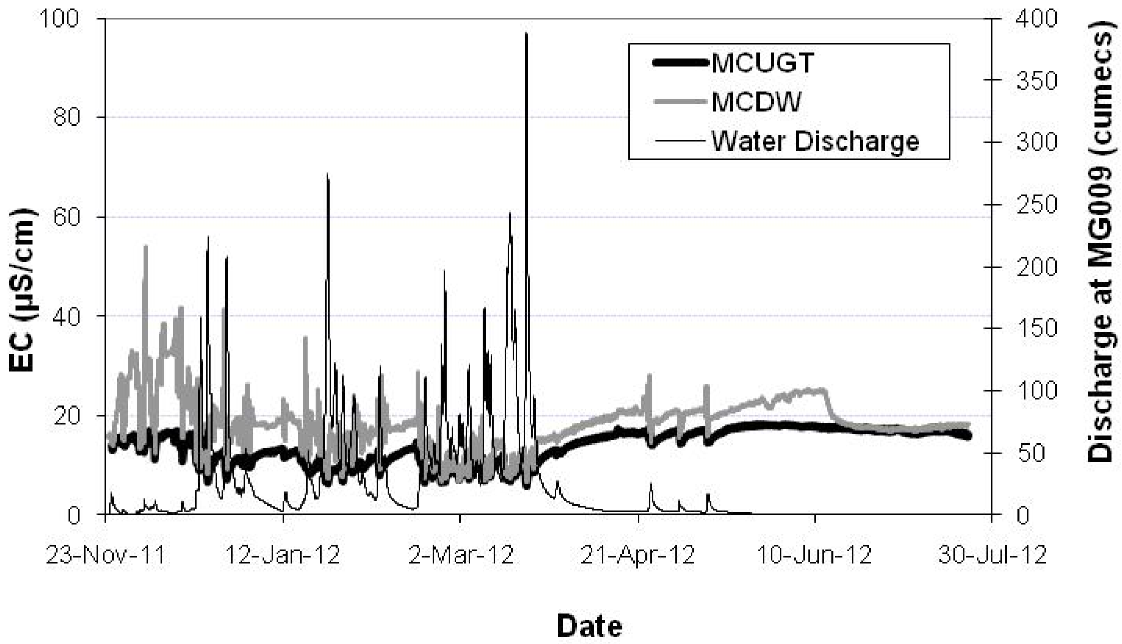

Figure 2 Continuous electrical conductivity and discharge in Magela Creek between November 2011 and July 2012

During the 2011–12 wet season, the maximum total uranium concentration measured at MCDW (downstream from the Ranger mine) was 0.4 µg/L, a value similar to the maximum filtered uranium concentration from previous years. This value is approximately 7% of the local ecotoxicologically-derived limit of 6 µg/L for protection of aquatic ecosystems, and approximately 2% of the 20 µg/L guideline for potable water (Figure 3).

The maximum uranium concentration occurred during the EC event on 4 December 2011, discussed previously. Manganese concentrations were also elevated, reaching 24 µg/L during the beginning of the event (Figure 4). This value is below the guideline trigger value of 26 µg/L for manganese. An earlier routine quality assurance/quality control (QA/QC) water sample taken on 30 November 2011 recorded a manganese concentration of 28 µg/L. However, surface water flows on this date were less than 0.5 cumecs, noting that the manganese guideline trigger value only applies when surface water flows are greater than 5 cumecs.

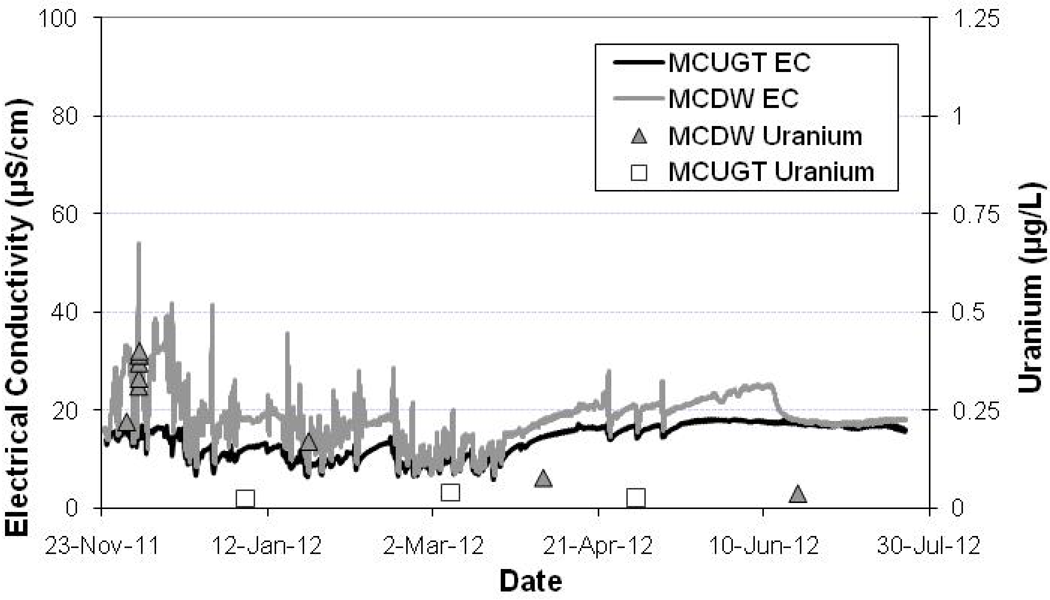

Figure 3 Total uranium concentrations in event-triggered samples and continuous electrical conductivity in Magela Creek between November 2011 and July 2012

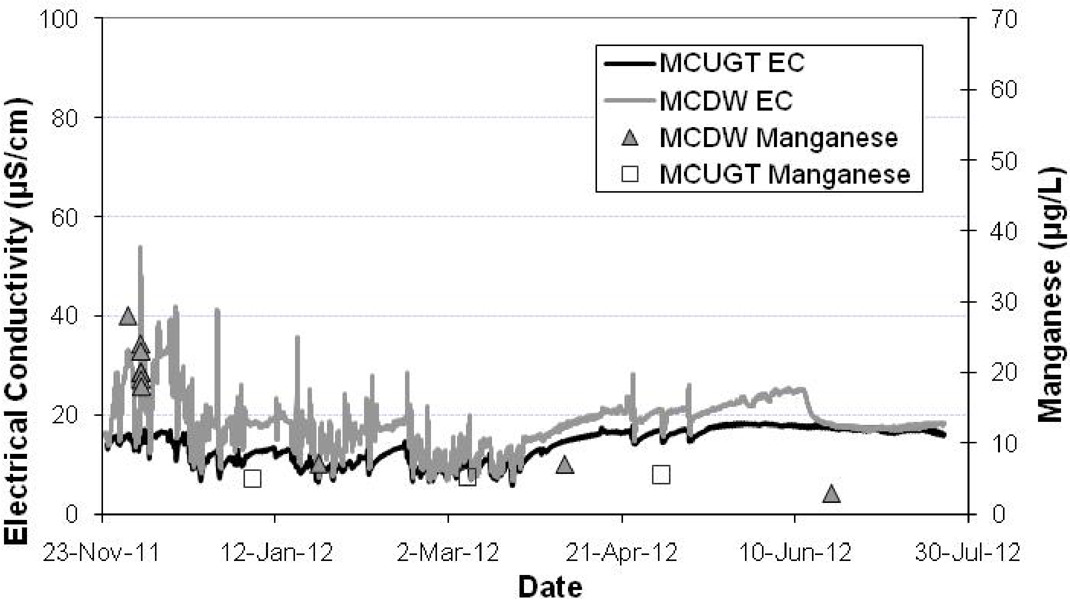

Figure 4 Total manganese concentrations in event-triggered samples and continuous electrical conductivity in Magela Creek between November 2011 and July 2012

Trends in magnesium and sulfate concentrations measured in the automatically collected water samples resembled the EC data with concentrations peaking during the EC event on 4 December 2012 at 3.8 mg/L and 13 mg/L, respectively (Figures 5 & 6). Automatic samples were not triggered for any other EC peaks during the 2011–12 wet season as these did not exceed the 43 µS/cm (corresponding to 3 mg/L magnesium) guideline.

Recessional flow conditions in Magela Creek are typified by stable EC before gradually increasing due to increased groundwater input. Recessional flow became established in Magela Creek during April 2012.

Continuous monitoring continues until cease to flow is agreed by stakeholders or until the water level is so low that the measuring equipment cannot be submerged. Monitoring ceased in late July in 2012.

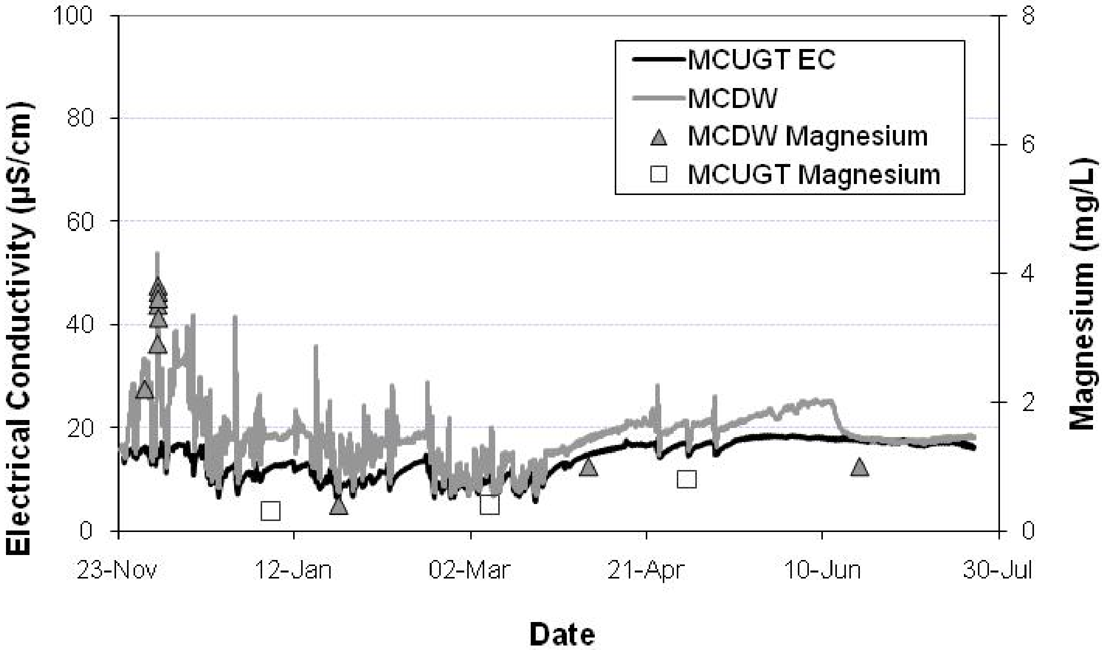

Figure 5 Total magnesium concentrations in event-triggered samples and continuous electrical conductivity in Magela Creek between November 2011 and July 2012

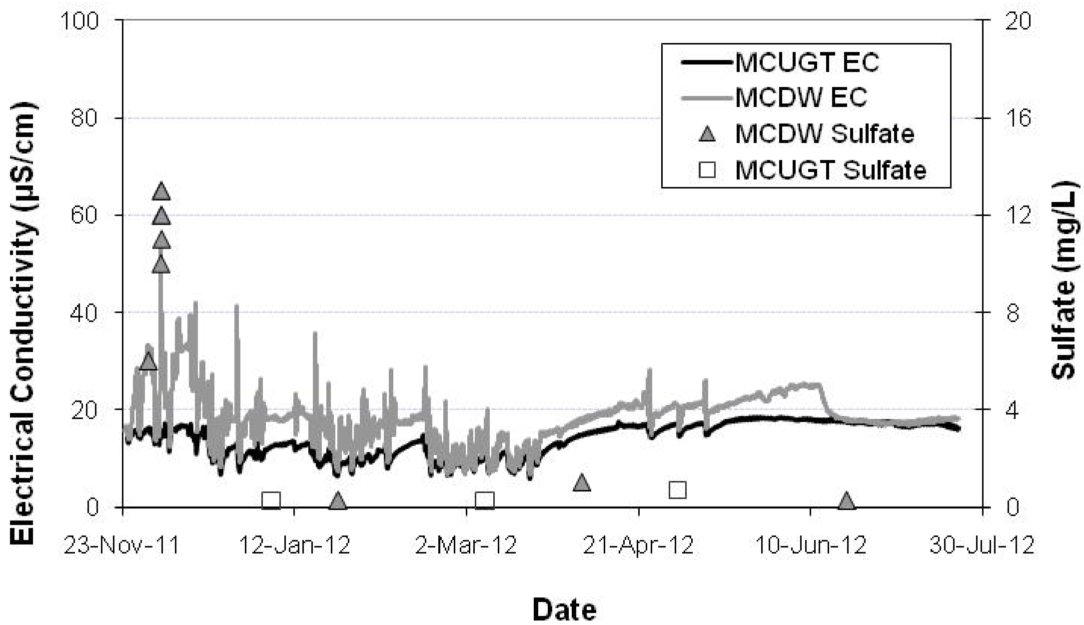

Figure 6 Total sulfate concentrations in event-triggered samples and continuous electrical conductivity in Magela Creek between November 2011 and July 2012

Conclusions

Overall, the water quality measured in Magela Creek for the 2011–12 wet season was comparable with previous wet seasons (Frostick et al 2012), with the results indicating that the aquatic environment in the creek has remained protected from mining activities (Figure 7).

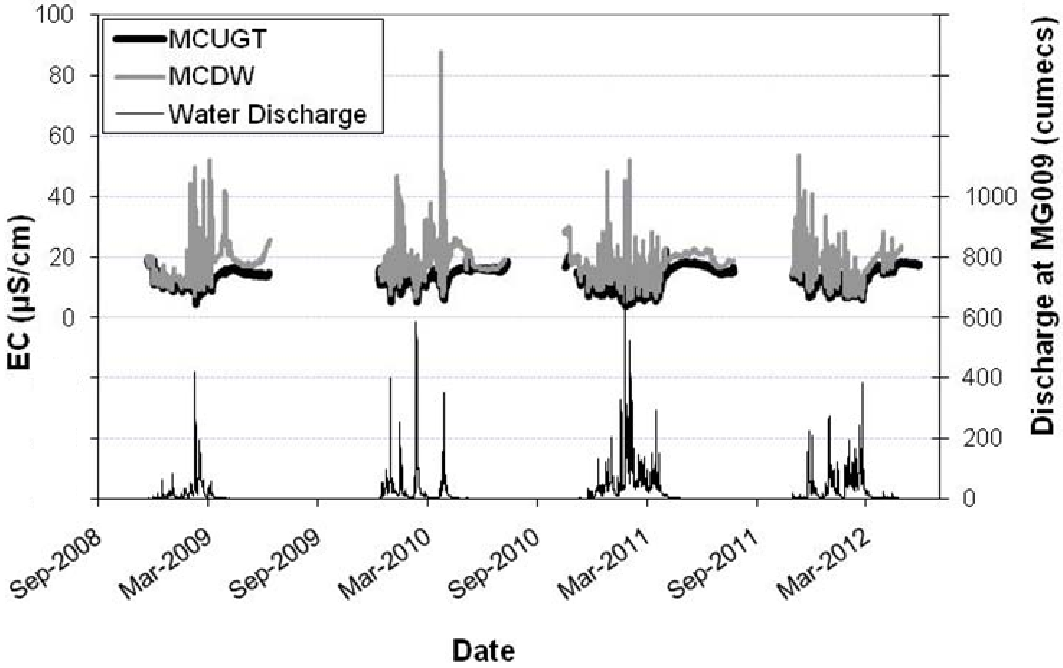

Figure 7 Continuous electrical conductivity and discharge (lower trace) in Magela Creek for each wet season between December 2007 and July 2012 (values averaged over a 1 hour period of measurement)

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Chemical and physical monitoring of Gulungul Creek

A Frostick, L Curtis & WD Erskine

Background

Flow commenced at the Gulungul Creek upstream and downstream monitoring stations on 24 November 2011. The location of the monitoring stations is shown in Map 2. Water quality upstream of Ranger mine was compared with that downstream of the mine using data obtained by continuous monitoring and pumped water samples (Walling 1978, 1984, Turner & Jones 2010).

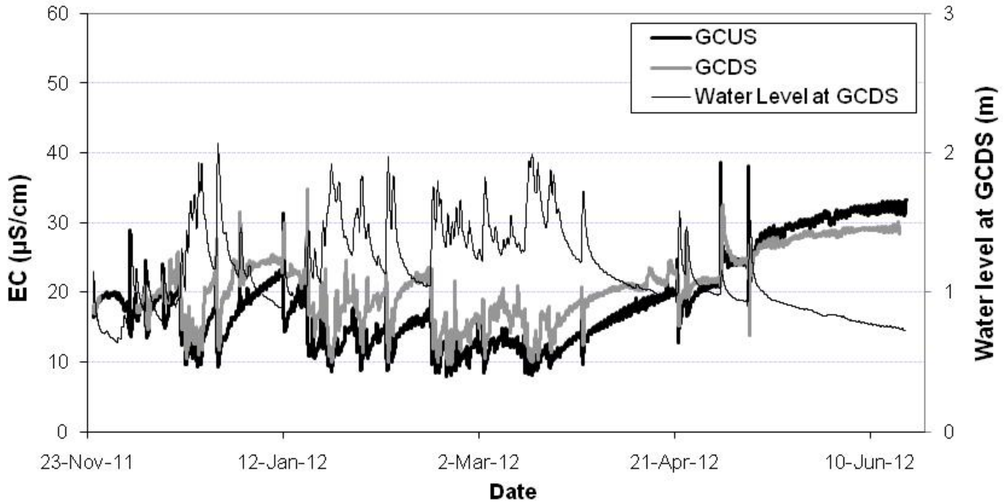

Figure 1 Electrical conductivity and water level in Gulungul Creek between November 2011 and June 2012

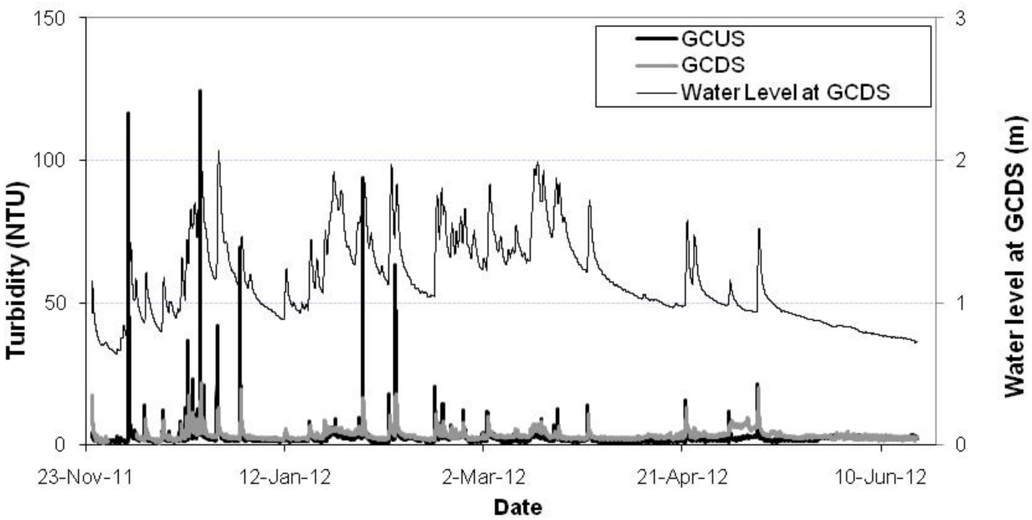

Figure 2 Turbidity and water level in Gulungul Creek between November 2011 and June 2012

Results

There was a peak in electrical conductivity (EC) of 29 µS/cm on 3 December 2011 at the upstream station (GCUS), which was associated with first flush conditions (Figure 1). The peak in EC was followed by a turbidity event which peaked at 117 NTU (Figure 2). These elevated turbidity levels are typical during the early wet season and result from inputs of highly turbid surface runoff from smaller drainage lines. Increased rainfall during mid-late December 2011 resulted in water levels remaining above 0.8 m at the downstream monitoring site (GCDS).

During late December 2011 and early January 2012 there was a steady decline in water level and a corresponding increase in EC. On 12 January 2012, 27 mm of rainfall was recorded at Jabiru Airport, 26 mm at GCDS and 10.2 mm at GCUS. Prior to the peak in water flow the EC reached 31 µS/cm at GCUS and 29 µS/cm at GCDS. A 38 mm rainfall event recorded at Jabiru Airport on 18 January 2012 (28 mm at GCDS and 20.5 mm GCUS) also resulted in peaks in EC associated with the rising hydrograph. EC reached 33 µS/cm at GCUS and 35 µS/cm at GCDS. Results from the EC-triggered automatic samples from GCUS and GCDS show conformance between uranium and, to a lesser extent, magnesium concentrations between the two stations. Differences between upstream and downstream EC, magnesium and sulfate concentrations suggest a localised source of runoff between the upstream and downstream stations. However, further work is required to determine the mine-related contribution to solute loads in Gulungul Creek (see Mackay & Erskine, 'Calculating annual solute loads in Gulungul Creek', this volume).

Water levels increased and EC decreased during subsequent rainfall events throughout late January, February and early March 2012. An increase in discharge on 18 and 19 March 2012 related to monsoonal activity resulted in a decrease in EC at both stations. Recessional flow conditions became established in April with low water levels and rising, converging EC levels at both stations. Late wet season rainfall events during late April and early May 2012 resulted in short-term peaks and troughs in EC and turbidity. Continuous monitoring continued until 17 June 2012 when water levels fell below the level of the sensors.

During the 2011–12 wet season, the maximum total uranium concentration of 0.52 µg/L was measured at GCUS (Figure 3). This value is approximately 9% of the local ecotoxicologically-derived limit of 6 µg/L, and approximately 2.6% of the 20 µg/L guideline for potable water.

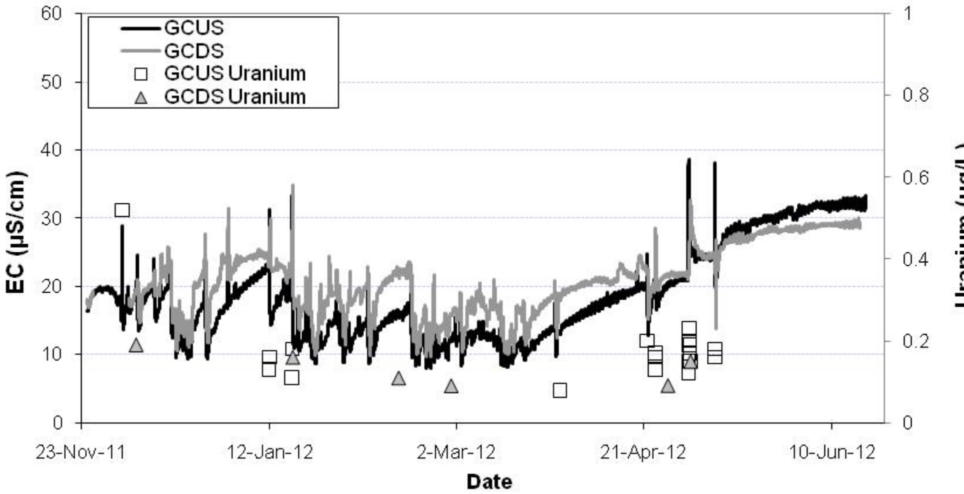

Figure 3 Electrical conductivity and total uranium concentrations in Gulungul Creek between November 2011 and June 2012

Manganese concentration peaks of 14 µg/L were recorded at both stations (Figure 4). Magnesium and sulfate concentrations closely followed the EC trace (Figures 5 & 6).

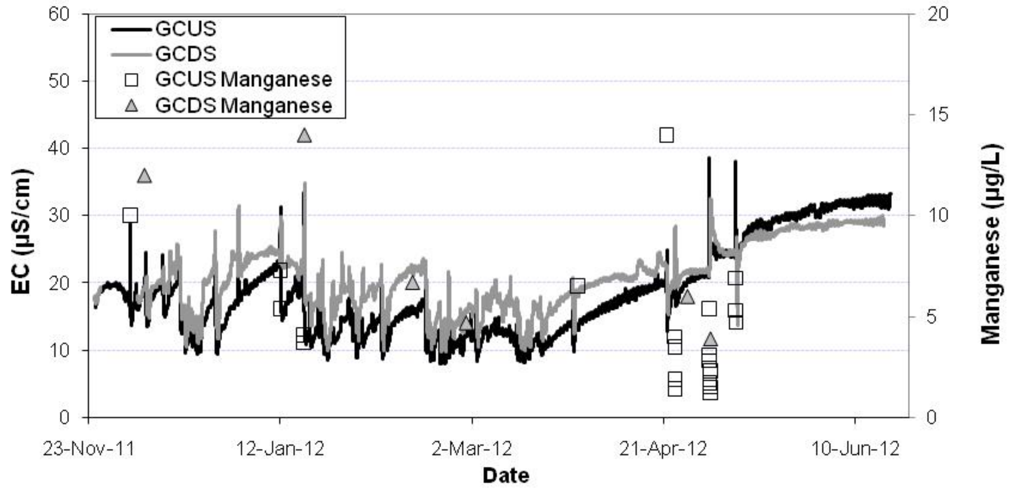

Figure 4 Electrical conductivity and total manganese concentrations in Gulungul Creek between November 2011 and June 2012

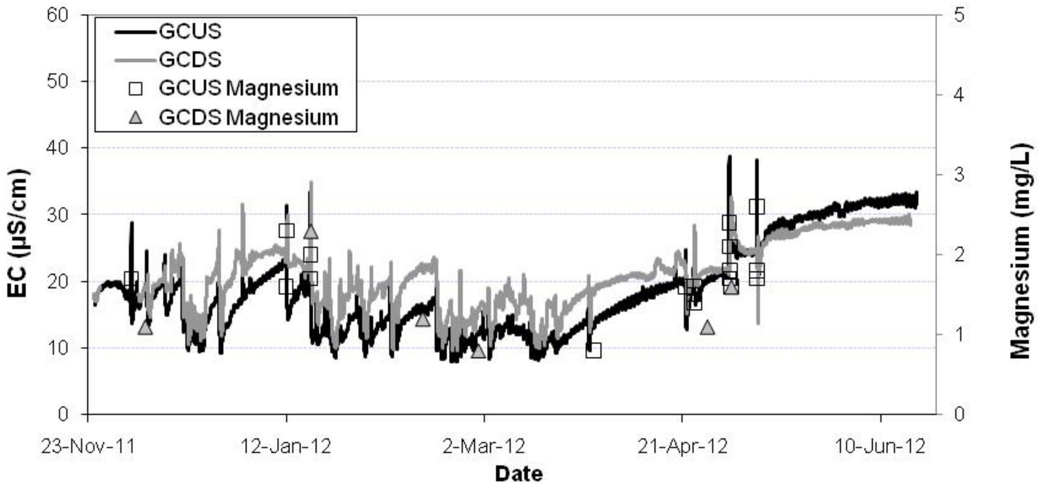

Figure 5 Electrical conductivity and total magnesium concentrations in Gulungul Creek between November 2011 and June 2012

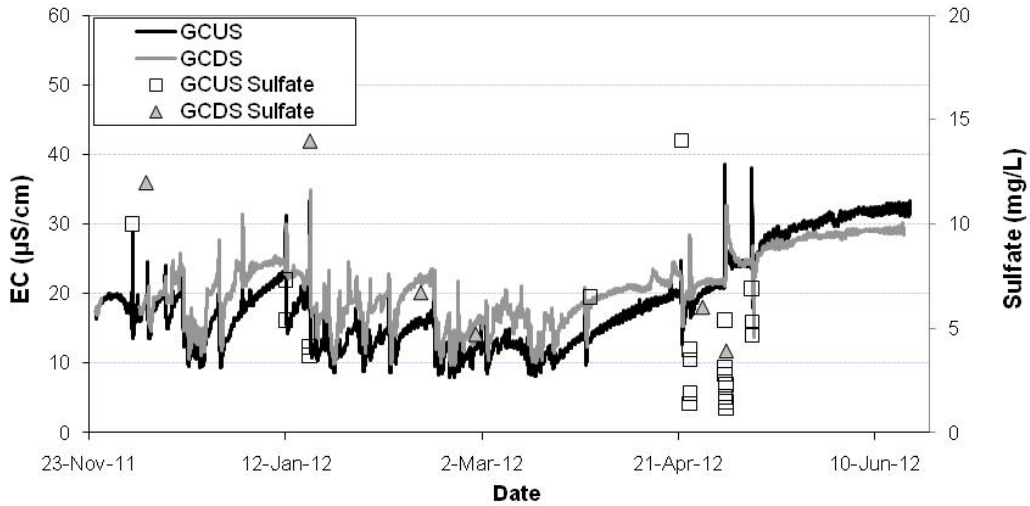

Figure 6 Electrical conductivity and total sulfate concentrations in Gulungul Creek between November 2011 and June 2012

Conclusions

Overall, the water quality measured in Gulungul Creek for the 2011–12 wet season was comparable with results from previous wet seasons (see Frostick et al 2012) and indicates that the aquatic environment has remained protected from mining activities (Figure 7). See Project 'Calculating annual solute loads in Gulungul Creek' (KKN 1.2.5 above) for a more detailed analysis of solute loads in Gulungul Creek (Mackay & Erskine, 'Calculating annual solute loads in Gulungul Creek', this volume).

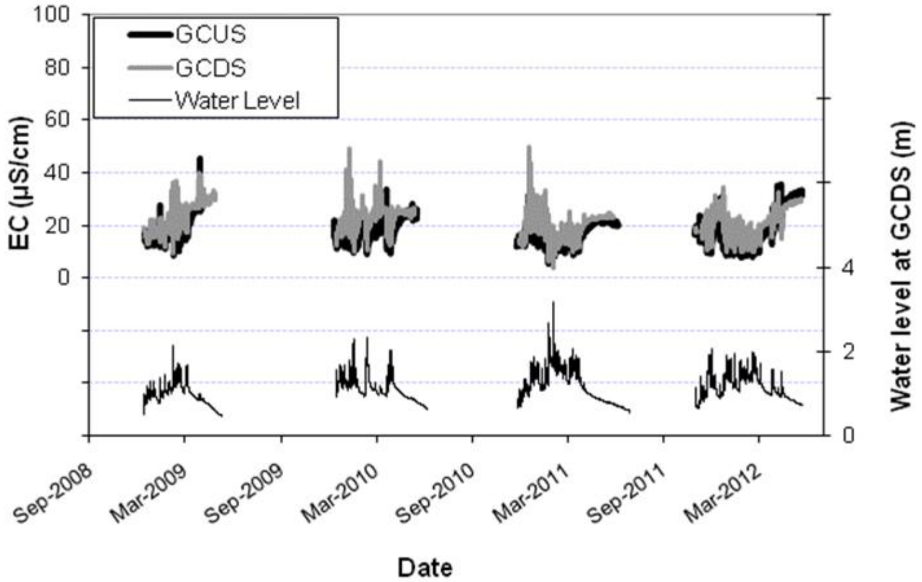

Figure 7 Electrical conductivity measurements and discharge (lower trace) in Gulungul Creek between December 2007 and July 2012 (values averaged over a 1 hour period of measurement)

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Surface water radiological monitoring in the vicinity of Ranger

P Medley & A Bollhöfer

Introduction

The limit value for total Radium-226 (^{226}Ra) activity concentrations in Magela Creek has been defined for human radiological protection purposes and is based on the median difference between upstream and downstream ^{226}Ra activity concentrations measured over one entire wet season (Klessa 2001). The basis of this limit arises from the potential risk of increased exposure to radiation via the biophysical pathway associated with mining activities. Freshwater mussels, in particular, bioaccumulate ^{226}Ra (Martin et al 1998, Bollhöfer et al 2011), which may then be incorporated into the human body upon consumption and deliver an internal radiation dose.

The median of the upstream ^{226}Ra data collected over the current wet season is subtracted from the median of the downstream data in Magela Creek. This difference value, called the 'wet season median difference', quantifies any increase at the downstream site, and should not exceed 10 mBq/L (Sauerland et al 2005). Using known concentration ratios for freshwater mussels in the region (Martin et al 1998) and ingestion dose coefficients from ICRP (1996) a wet season median difference of 10 mBq/L would result in a mine origin ingestion dose from ^{226}Ra bioaccumulated in mussels of about 0.3 mSv, if 2 kg of mussels were ingested by a 10 year old child per year. For Gulungul Creek there is currently no limit value for water ^{226}Ra activity concentrations, as variability is much larger and upstream activity concentrations are typically higher than those measured downstream.

Methods

Surface water samples are collected weekly, alternating between Magela and Gulungul Creeks, upstream and downstream of the Ranger mine. Fortnightly samples from each creek are then combined to give monthly composite samples. Total ^{226}Ra is measured in these samples and results for the 2011–12 wet season can be compared with previous data ranging back to the 2001–02 wet season for Magela Creek. Since 2010, ^{226}Ra analyses of a composite of samples collected by autosampler during individual EC-triggered events have also been performed.

Samples are analysed for total ^{226}Ra (ie dissolved plus particulate phase) via alpha spectrometry in the ***eriss*** environmental radioactivity laboratory using a method described in Medley et al (2005). Alpha spectrometry is also used for ^{228}Ra determination after allowing for ingrowth of the ^{228}Th daughter (Medley 2010).

^{226}Ra activity concentration in Magela Creek

The ^{226}Ra activity concentration data in Magela Creek for the 2011–12 wet season are compared with the previous wet season data in Figure 1. Wet season median differences (shown by the horizontal lines in Figure 1) from 2001 to 2012 are close to zero, indicating that the majority of ^{226}Ra is coming from natural sources of Ra located in the catchment upstream of the mine. The wet season median difference for the entire monitoring period (2001–2012) is 0.1 mBq/L.

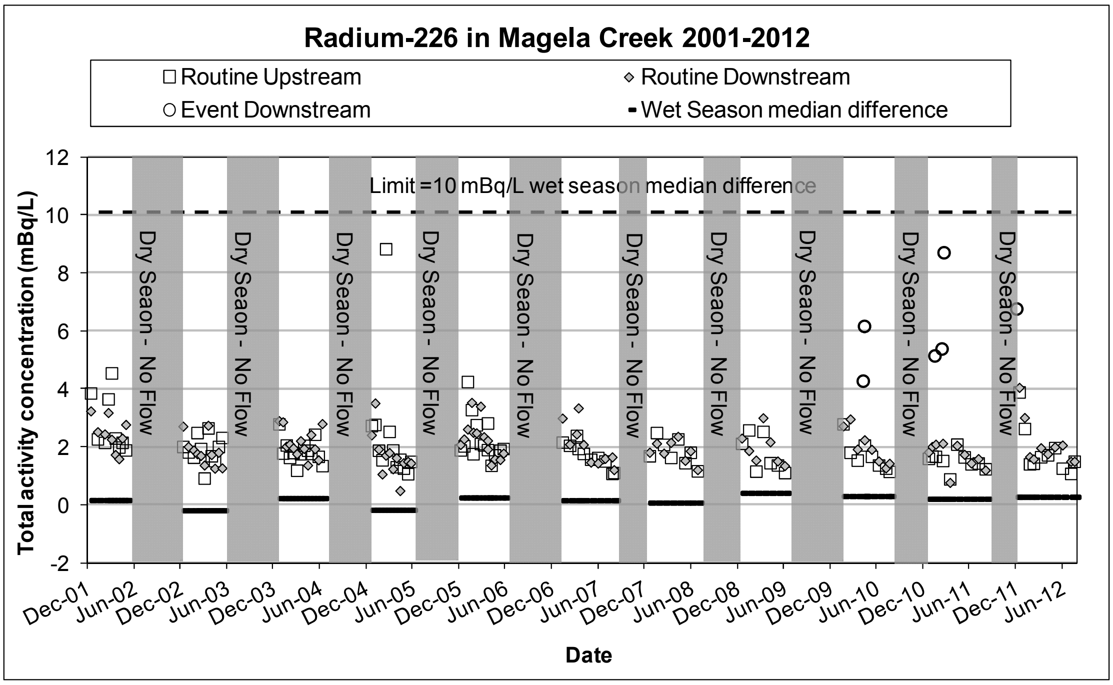

Figure 1 Radium-226 in Magela Creek 2001–2012

The data from monthly sample composites show that the levels of ^{226}Ra are very low in Magela Creek, both upstream and downstream of the Ranger mine. An anomalous ^{226}Ra activity concentration of 8.8 mBq/L measured in a sample collected from the control site upstream of Ranger in 2005 was probably due to a higher contribution of ^{226}Ra-rich soil or finer sediments that are present naturally in Magela Creek (see previous discussion in Sauerland et al (2006).

Since 2010, ^{226}Ra analyses of a composite of samples collected by autosampler during individual EC-triggered events have also been performed. The results are shown in Figure 1, together with the results from the routine radium analyses. The data are not included in the calculation of the wet season median difference shown in Table 1, because these EC events are short-lived and their impact on seasonal ^{226}Ra loads is very small.

The higher radium activity concentrations seen for the event downstream samples are a consequence of the new automated sampling procedure, which is triggered only at times of higher EC values. There was one (4 December 2011) such event-triggered sample obtained for ^{226}Ra analysis during the 2011–12 wet season. The maximum uranium concentration measured in event samples collected for 2011–12 also occurred during this EC event (see Figure 3 in 'Chemical and physical monitoring of Magela Creek', this volume). There was no corresponding upstream (reference) sample collected at the same time since there was no elevated EC pulse upstream to trigger the autosampler at that location. To enable the activity difference of 4.9 mBq/L to be estimated for this December sample, the median of all previous upstream routine ^{226}Ra results (2001–2011) was used as a reference and subtracted from the ^{226}Ra result for this EC triggered sample.

Table 1 Median and standard deviations of the ^{226}Ra activity concentration in Magela Creek (mBq/L) for individual wet seasons (2001–12)

Wet season	Median and standard deviation		Median difference
	Upstream	Downstream	
2001–2002	2.3 ± 1.0	2.5 ± 0.6	0.2
2002–2003	2.0 ± 0.5	1.8 ± 0.5	-0.2
2003–2004	1.8 ± 0.4	2.0 ± 0.5	0.2
2004–2005	1.7 ± 2.1	1.6 ± 0.7	-0.2
2005–2006	2.0 ± 0.8	2.3 ± 0.6	0.3
2006–2007	1.7 ± 0.4	1.9 ± 0.7	0.2
2007–2008	1.8 ± 0.4	1.8 ± 0.4	0.1
2008–2009	1.5 ± 0.6	1.9 ± 0.6	0.4
2009–2010	1.6 ± 0.5	1.9 ± 0.6	0.3
2010–2011	1.6 ± 0.3	1.8 ± 0.4	0.2
2011-2012	1.6 ± 0.8	1.9 ± 0.8	0.3
All years	1.8 ± 0.9	1.9 ± 0.6	0.1

^{226}Ra activity concentration in Gulungul Creek

Since the 2010–2011 wet season, the ^{226}Ra monitoring program has been extended to Gulungul Creek. This is due to the potential for mine impact in the Gulungul catchment in particular as a result of works associated with several lifts of the tailings storage facility (TSF) wall. EC-triggered event sampling is not undertaken for ^{226}Ra in Gulungul Creek and ^{228}Ra analysis has not been undertaken to date.

Activity concentrations for the 2010–11 and 2011–12 wet seasons are shown in Figure 2.

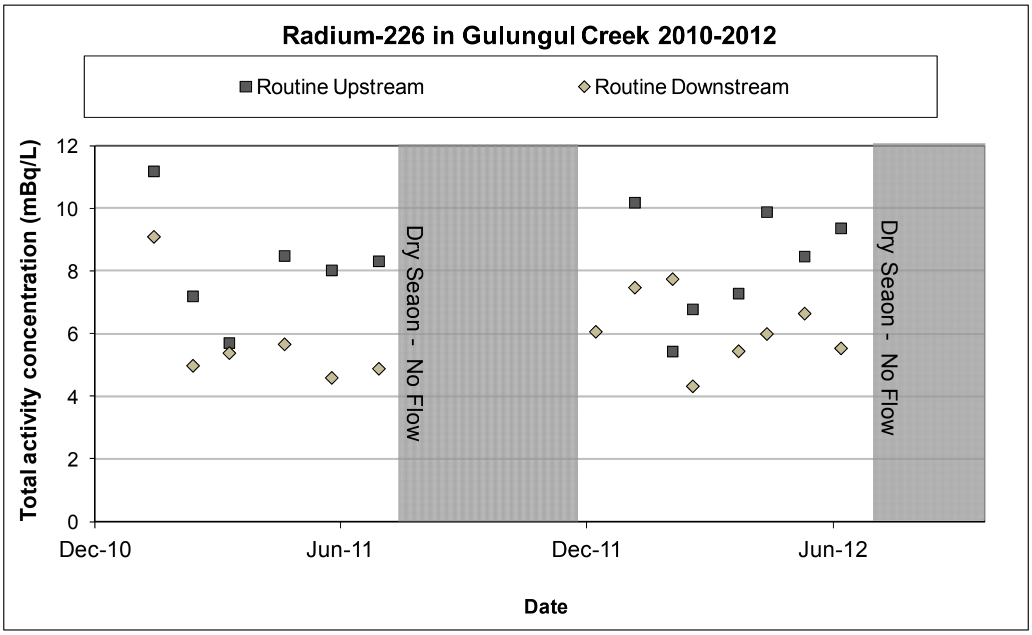

Figure 2 Radium-226 in Gulungul Creek 2010–2012

The wet season median difference for the two year monitoring period (2010–2012) is -2.8 mBq/L (that is, the upstream site has a higher average activity concentration than the downstream site over both years). The median for the upstream site over both wet seasons is 8.0 ± 1.8 mBq/L with upper 80th and 95th percentile values of 10.0 mBq/L and 11.6 mBq/L respectively. The median for the downstream site over both wet seasons is 5.6 ± 1.3 mBq/L with upper 80th and 95th percentile values of 7.0 mBq/L and 9.1 mBq/L, respectively.

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Toxicity monitoring in Magela and Gulungul Creeks

CL Humphrey, M Ellis & D Buckle

Background

In this form of monitoring, effects on receiving waters of water dispersed from the Ranger minesite are evaluated using responses of aquatic animals exposed in situ to creek water. The response measured is reproduction (egg production) by the freshwater snail, *Amerianna cumingi*. Each test runs over a four-day (96-hr) exposure period. In such chronic exposure situations, this species has been shown to be among the most sensitive, to both uranium and magnesium, of SSD's suite of six local species as determined using standardised laboratory toxicity test protocols.

For the 1990–91 to 2007–08 wet seasons, toxicity monitoring was carried out using the 'creekside' methodology. This involved pumping a continuous flow of water from the adjacent Magela Creek through tanks containing test animals located under a shelter on the creek bank. In the 2008–09 wet season, this method was replaced by an in situ testing method in which test animals are placed in floating (flow-through) containers located in the creek itself (see section 3.2 of the 2007–08 Supervising Scientist annual report for details). The most recent refinement to this program has been the extension of toxicity monitoring to Gulungul Creek, with testing commencing in the 2009–10 wet season.

Testing was conducted fortnightly in each creek in the 2011–12 wet season, alternating each creek on a weekly basis (as such, testing was not conducted in both creeks in the same week.)

The first of 18 toxicity monitoring tests (both creeks combined) for the 2011–12 wet season commenced in Magela Creek on the 1 December 2011, once moderate creek flows were established. Tests were then conducted every other week over the 2011–12 wet season with the ninth and final test commencing 5 April 2012. The first of nine Gulungul tests commenced on 8 December 2011 and the final test commenced on 29 March 2012. The second Gulungul test deployed on the 16 December was deemed invalid due to an excess amount of debris that had collected on the equipment, restricting water flow to the snails during the test period (Figure 1B). Consequently, data from this test were not included in subsequent analyses. Upstream and downstream egg production and difference values for both creeks are displayed in Figure 1B.

Analysis of Magela Creek results

After each wet season, toxicity monitoring results for the tests are analysed, with differences in egg numbers (the 'response' variable) between the upstream (control) and downstream (exposed) sites tested for statistical change between the wet season just completed and previous wet seasons.

The mean number of eggs produced each test at the upstream and downstream sites for 2011–12 fell within previously-recorded ranges in values for the in situ method (Figure 1B). The difference values recorded during the 2011–12 wet season continue the trend of greater egg production downstream since inception of toxicity monitoring, with an average difference value of -8.2 (Figure 1B). This is much closer to the historical mean of -5.8 (pre 2009–10), than to the means recorded in the previous two years (2009–10 = -22.3, 2010–11 = -12.8), resulting in a new mean difference value across the complete time series of -8.0.

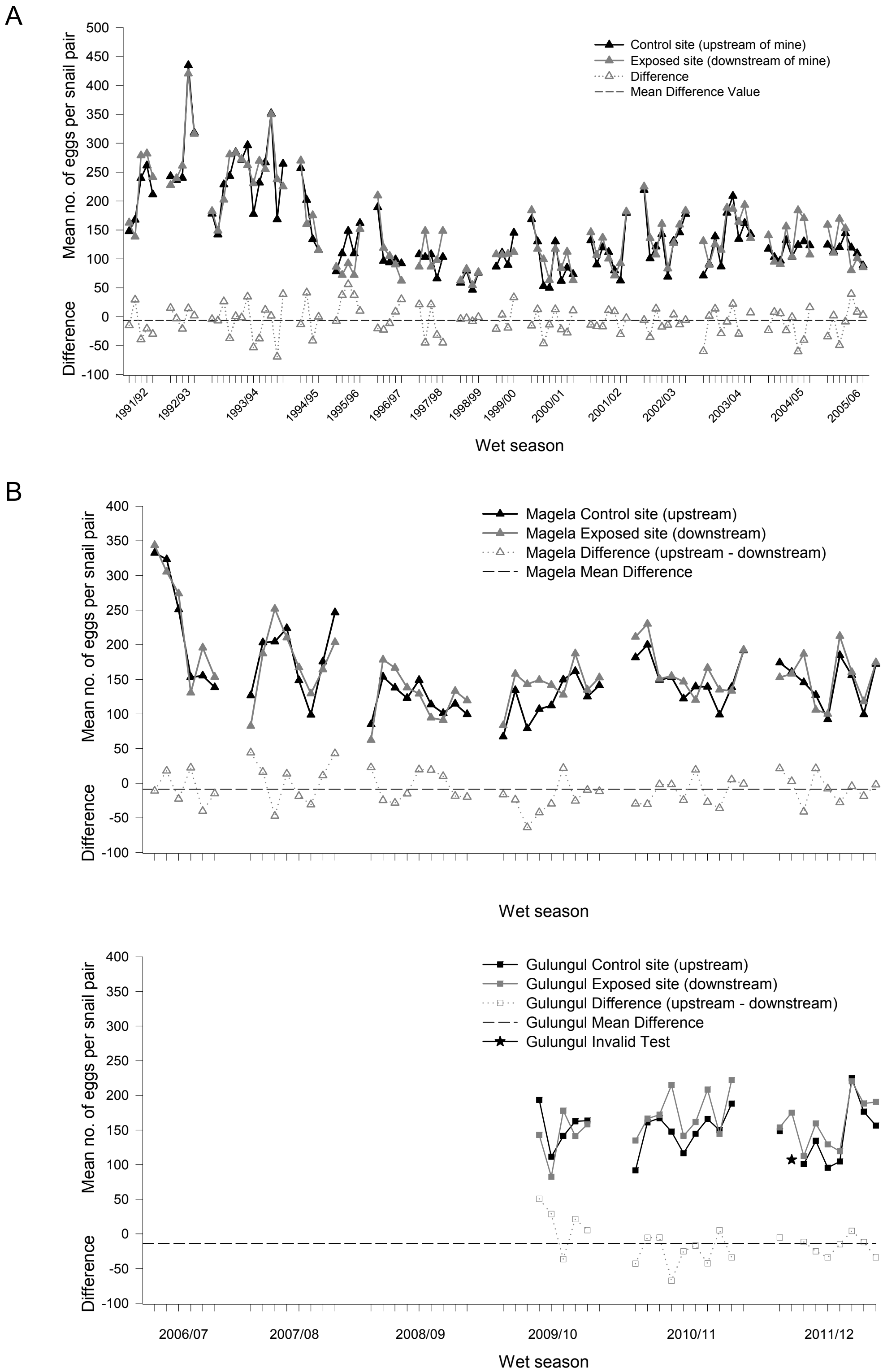

Figure 1 Time-series of snail egg production data from toxicity monitoring tests conducted in Magela Creek using A: (mostly) creekside tests, and B: in situ tests with Gulungul tests commencing in 2009–10

Unlike the previous two wet seasons (see Humphrey et al 2012 and Table 1), Analysis Of Variance (ANOVA) testing found no significant difference between the results (upstream-downstream difference values) of the wet season of interest (2011–12) and those from previous years (Table 1, P = 0.907). See also Humphrey et al, 'Ranger stream monitoring research: Analyses of toxicity monitoring and associated water quality data for Magela and Gulungul Creeks', this volume, where additional interpretation of Magela Creek toxicity monitoring results from recent years is provided.

Table 1 Results of ANOVA testing comparing Magela upstream-downstream difference values for mean snail egg number for different 'before versus after' wet season scenarios

Statistical comparison	Probability value (*P*)	Significance
2009–10 compared with all previous seasons	0.043	at 5% level
2010–11 compared with all previous seasons	0.434	NS
2010–11 compared with previous seasons excl 2009–10	0.315	NS
2010–11 + 2009–10 compared with previous seasons	0.044	at 5% level
2011–12 compared with all previous seasons	0.907	NS
2011–12 compared with previous seasons excl 2009–10	0.989	NS

NS = Not significant

Analysis of Gulungul Creek results

Gulungul Creek results for the 2011–12 wet season show consistently higher snail egg production at the downstream site compared with the upstream site, with seven of the eight valid tests producing a negative difference value (Figure 1B). Thus the results for this wet season are similar to those observed for the previous season (2010–11; eight of the nine tests with a negative difference value) but are in contrast to those observed during the initial wet season (2009–10) when four out of the five tests resulted in positive difference values, indicating higher upstream egg production.

ANOVA testing found no significant difference between the upstream-downstream difference values for the wet season of interest (2011–12) and those from the previous two years (Before period, 2009–10 and 2010–11; P =0.813). However, when data from 2010–11 and 2011–12 wet seasons are combined and compared with 2009–10 data, the ANOVA result is close to significant (P = 0.055) reflecting interannual variability in Gulungul Creek upstream-downstream difference values that has not been encountered in Magela Creek. Gulungul Creek toxicity monitoring results from recent years are discussed in more detail (see Humphrey et al, 'Ranger stream monitoring results: Analyses of toxicity monitoring and associated water quality data for Magela and Gulungul Creeks', this volume).

Conclusions

Concordance in the snail egg production responses between upstream and downstream sites in Magela and Gulungul Creeks for the 2011–12 wet season was high (Figure 1) with difference values consistent with, and not significantly different from, long-term mean values. Further analysis of the data is provided below (section 'Analyses of toxicity monitoring and associated water quality data for Magela and Gulungul Creeks') in the context of discerning possible effects of mine-input EC on enhancing snail egg production.

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Bioaccumulation of uranium and radium in freshwater mussels from Mudginberri and Sandy Billabongs

A Bollhöfer, C Doering, P Medley & T Fox

Introduction

Local Aboriginal people harvest aquatic food items including fish and mussels from Mudginberri Billabong, 12 km downstream of the Ranger mine (Map 3). Ongoing research and monitoring has shown that mussels accumulate some metals and radionuclides, in particular ^{226}Ra (Johnston et al 1987, Martin et al 1998, Bollhöfer et al 2011, Bollhöfer 2012). In contrast, nine years (2000–08) of monitoring metals and radionuclide concentrations in fish from Mudginberri Billabong has not shown any issues of potential concern with regards to bioaccumulation of radionuclides. Hence, the previous two-yearly fish sampling program has been discontinued since 2007 and the bioaccumulation monitoring effort has focussed on the mussels component.

This component of the project ensures that mussels are fit for human consumption with respect to concentrations of metals and/or radionuclides, and that the concentrations in mussel tissue attributable to mine-derived inputs from Ranger remain within acceptable levels. As the enhanced body burdens of mine-derived solutes in mussels could potentially reach limits that may harm the organisms themselves, the bioaccumulation monitoring program serves an ecosystem protection role in addition to the human health aspect. It also achieves this by the potential to provide early warning of bioavailability of metals and/or radionuclides to the creek system.

Uranium and radium bioaccumulation data have been obtained intermittently from Mudginberri Billabong between 1980 and 2000. Between 2000 and 2008, mussels were collected annually from Mudginberri (the potentially impacted site) and Sandy Billabongs (the control site in a different catchment, sampled from 2002 onwards). The results showed that radionuclide burdens in mussels from Mudginberri Billabong were generally about twice that in the reference Sandy Billabong (Ryan et al 2005). Two research projects that investigated factors controlling the sources and uptake of radionuclides in mussels along the Magela catchment concluded that this difference was due to natural catchment influences and differences in water chemistry, rather than any mining-related inputs (Bollhöfer et al 2011).

As a result of these projects the scope of the mussel bioaccumulation monitoring program was reduced from 2009. It now involves the annual collection and analysis of a bulk mussel sample from Mudginberri Billabong at the end of the dry season, rather than analysing age-classed mussels from both billabongs. In addition to this, every three years (starting with the October 2011 collection reported here) a detailed study analysing aged mussels from Mudginberri and Sandy Billabongs is conducted and results are compared with those from previous years. This project is continued primarily to provide ongoing assurance to the public that the consumption of mussels from Mudginberri Billabong does not present a radiological risk.

Methods

Mussels were collected in October 2011 by hand in Sandy Billabong and by dredge in Mudginberri Billabong (Figure 1). After collection, mussels were placed into acid-washed containers holding water from the respective billabongs. Surface water samples were collected at the same time in acid washed containers and sediments associated with the mussels were collected and stored in zip-locked plastic bags. The mussels, water and sediment samples were taken to the Darwin laboratories for processing.

Figure 1 Mussel sampling at Mudginberri Billabong

Mussels were purged in host billabong water for three days in the Darwin laboratories, before being measured for length, breadth and width, and dissected to remove the mussel flesh. The wet weight of the mussel flesh was recorded and samples were freeze dried then re-weighed to determine the dry weight. The age of each mussel was determined by counting the number of annual growth bands preserved in the mussel shell (Humphrey & Simpson 1985).

Mussels in the same age class from individual billabongs were then combined into one sample and three aliquots of each sample were taken. Aliquot 1 was used to determine the concentrations of uranium and other heavy metals via inductively coupled plasma mass spectrometry (ICP-MS). Aliquot 2 was taken for radium analysis involving radiochemical separation procedures and alpha spectrometry. The remainder of the sample was used to determine the activity concentrations of radioisotopes of radium (^{226}Ra & ^{228}Ra), lead (^{210}Pb) and thorium ($^{228^{Th}}$) using gamma spectrometry. Water and sediment samples were analysed for ^{226}Ra via alpha and gamma spectrometry, and for heavy metals using ICP-MS.

Uranium in freshwater mussels

Uranium concentrations in mussels, water and sediment samples collected concurrently from Mudginberri and Sandy Billabongs over the years are shown in Figure 2.

The concentration of uranium in mussels from Mudginberri Billabong are very similar from 2000 onwards, with no evidence of an increasing trend in concentration over time. The essentially constant levels of U in mussel tissues observed between 1989 and 1995 (Brazier et al 2008), consistently low levels from 2000 to the last sample taken in October 2011 and no observable increase over time indicates the absence of any mining related influence.

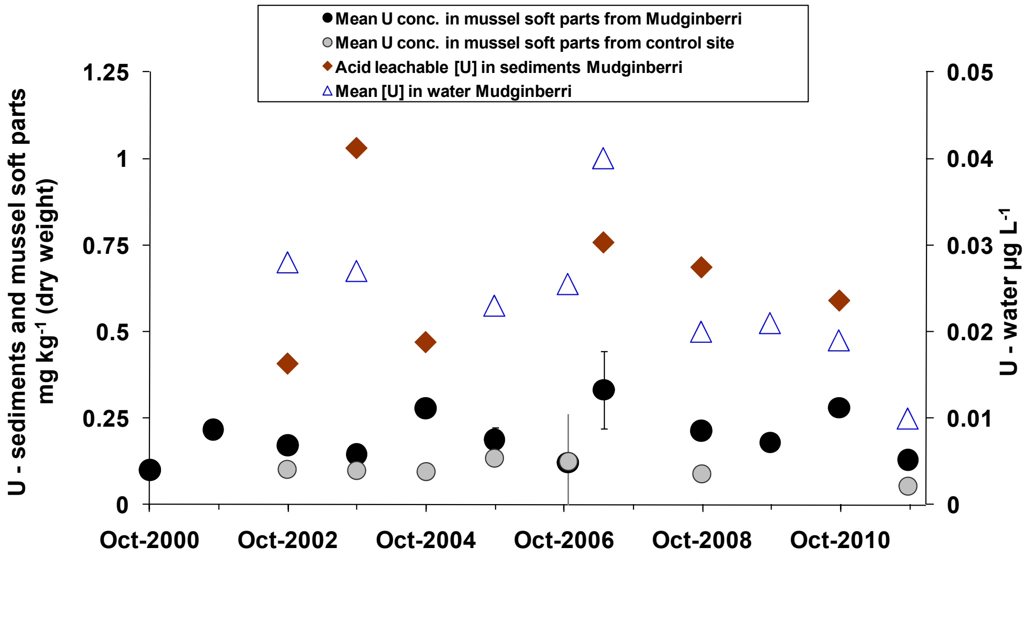

Figure 2 Mean concentrations of U measured in mussel soft-parts and sediment (dry weight basis) and in water samples collected from Mudginberri and Sandy Billabongs since 2000

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

In Figure 3, ^{226}Ra and ^{210}Pb activity concentrations in mussels collected from Mudginberri and Sandy Billabongs in October 2011 are compared with the average ^{226}Ra and ^{210}Pb activity concentrations measured in previous years. The graphs show that ^{226}Ra activity concentrations in aged mussels from Sandy Billabong are lower than from Mudginberri Billabong, as reported previously (Ryan et al 2005), and that activity concentrations were no different in 2011 compared with the average from previous collections.

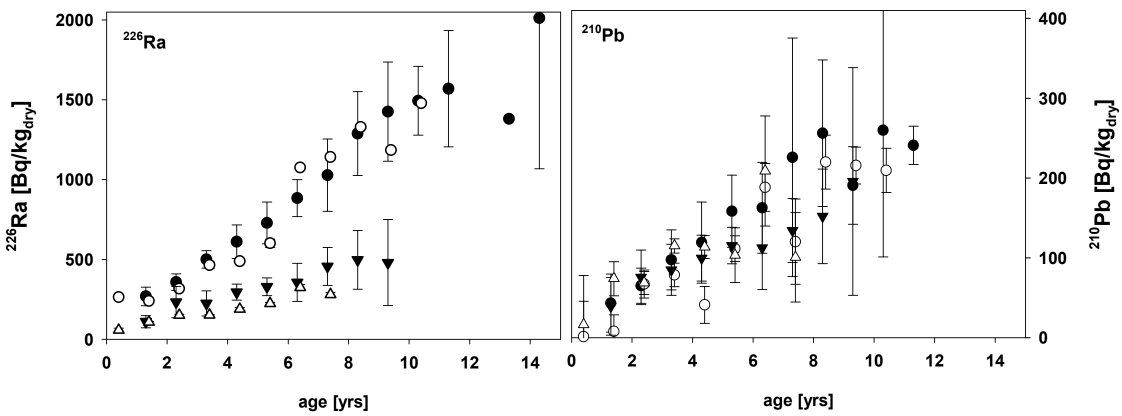

Figure 3 ^{226}Ra and ^{210}Pb activity concentrations measured in dry mussels flesh from Mudginberri (circles) and Sandy (triangles) Billabongs. The average of previous end of dry season collections (2000–2008) is shown as solid symbols, open symbols show the results from the 2011 collection.

In Figure 4, the ^{226}Ra activity concentrations are plotted for individual collections and the different mussel age groups. The figure shows that there has been no increase in ^{226}Ra activity concentration in mussels of the same age group since 2001.

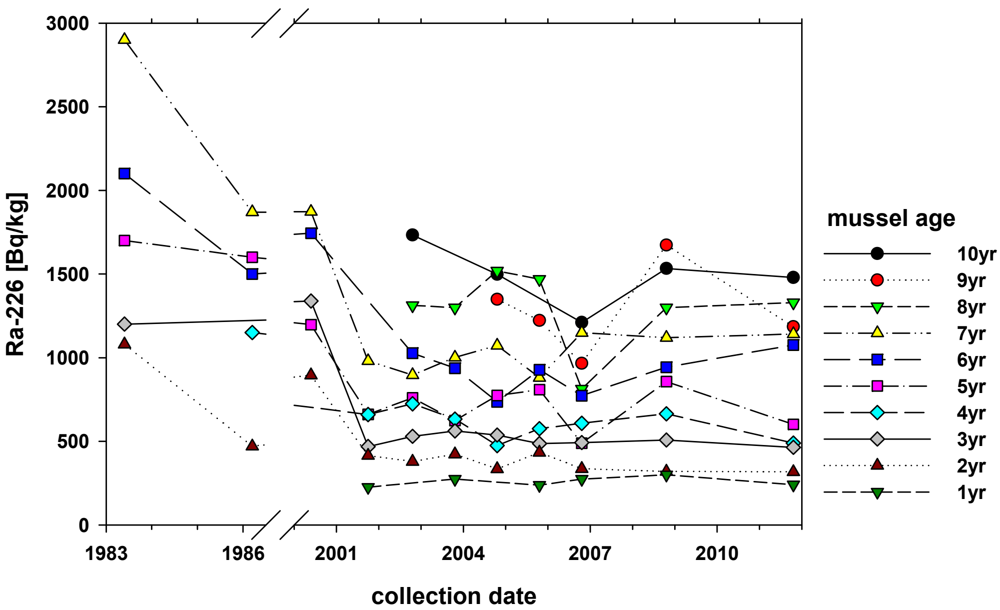

Figure 4 Mussel age and ^{226}Ra activity concentrations for various mussel collections

^{226}Ra activity concentrations in flesh from mussels collected in 1983, 1986 and 2000 are higher than measurements made from September 2001 onwards. However, the 1983, 1986 and 2000 collections were done earlier in the year, between February and May. Typically, mussel condition is poorer (ie the mussel weight is generally lower) in, or just after, the wet season, compared with the dry season (Humphrey & Simpson 1985). The ^{226}Ra activity in the mussel flesh however does not change because ^{226}Ra, once taken up by the mussels, has shown to have a relatively long biological half-life (Johnston et al 1987, Bollhöfer et al 2011). Consequently, the activity concentration of ^{226}Ra in mussel flesh is higher in May than in October, though the total ^{226}Ra activity per mussel remains the same.

Radium measurements via alpha and gamma spectrometry

^{228}Ra

Aliquots of pooled mussels of the same age were analysed for radium isotopes (^{226}Ra and ^{228}Ra) via gamma and alpha spectrometry to calibrate and test a new radioanalytical method. This method was developed to allow determination of ^{228}Ra activity concentrations in environmental samples via measurement of the activity of the gamma emitting daughter radionuclide ^{228}Ac in a radiochemically separated radium (^{226}Ra) sample, or after ingrowth and subsequent measurement of ^{228}Th in that same sample via alpha spectrometry (Medley 2010).

Radium in Mudginberri and Sandy Billabong mussels was radiochemically separated using the $BaSO_4$ co-precipitation method (Medley et al 2005). The ^{228}Ac activity was then measured as a surrogate of the ^{228}Ra activity using gamma spectrometry, whereas the ^{226}Ra activity was measured directly via alpha spectrometry. Figure 5 shows mussel ^{228}Ra activity concentrations determined using the new method plotted versus the conventional gamma spectrometry results. The plot also shows a regression fit (slope = 0.99) and the associated 95% confidence intervals. ^{228}Ra activity concentrations measured in the samples typically agree within 20% although there are larger deviations for a few individual samples. These deviations are most likely an effect of heterogeneities due to the small aliquot mass (< 1 g) used for the new method.

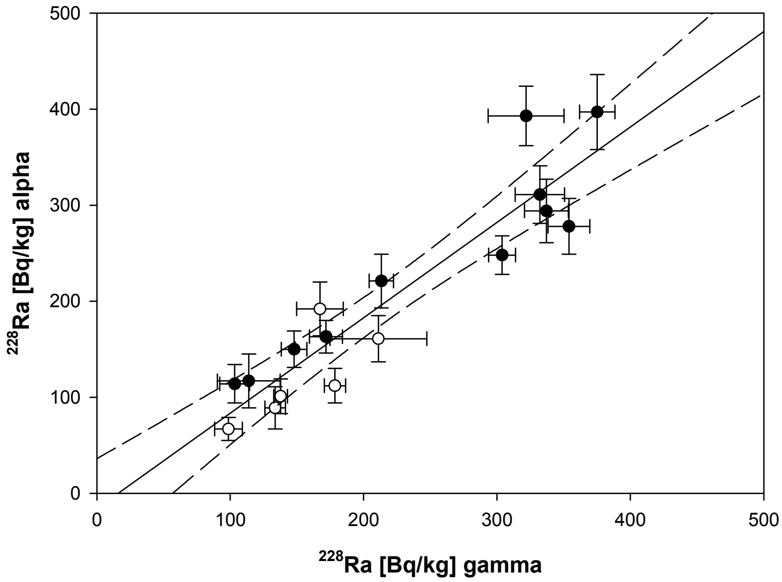

Figure 5 ^{228}Ra activity concentrations in mussel flesh (filled symbols: Mudginberri Billabong; open symbols: Sandy Billabong) determined using Medley (2010) plotted against conventional gamma spectrometry results. Errors shown are standard deviations from counting statistics only.

^{226}Ra

^{226}Ra activity concentration was also measured in mussel samples via alpha spectrometry in addition to the conventional gamma spectrometry measurements. Noticeably five samples displayed a large discrepancy between ^{226}Ra activity concentrations determined using the two methods, with alpha spectrometry results higher than results determined using conventional gamma spectrometry (Figure 6). The respective samples however did not show a similar discrepancy in their ^{228}Ra results (Figure 5).

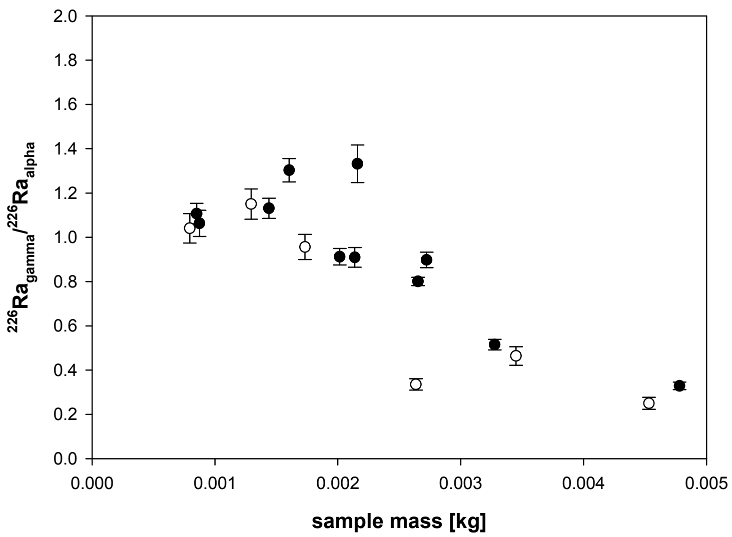

Figure 6 The ratio of the ^{226}Ra activity concentrations measured in mussel flesh using gamma and alpha spectrometry plotted versus the sample mass (filled symbols: Mudginberri Billabong; open symbols: Sandy Billabong). Errors shown are standard deviations from counting statistics only.

The technique for analysis of ^{226}Ra via gamma spectrometry uses indirect measurement of the radon (^{222}Rn) decay products ^{214}Pb and ^{214}Bi in the sample. Samples are cast in epoxy resin, and an ingrowth period of 21 days before measurement assures that radon is in equilibrium with its radioactive parent ^{226}Ra. It is generally assumed that the sample matrix (dried mussel flesh cast in epoxy resin) prevents radon from degassing from the sample (Marten 1992). Our investigation, however, showed that for samples with masses exceeding ~2.5 grams some of the radon may escape from the sample, resulting in lower ^{226}Ra activity concentrations measured via gamma spectrometry when compared with alpha spectrometry.

This 'radon leakage' from the sample matrix was confirmed via measurement of the ^{222}Rn activity leaking from the samples using a RAD-7 radon detector. Some leakage of radon will occur in samples prepared for analysis, however, the activity of radon escaping was much larger than expected for samples with a large mass. It is believed that large amounts of dried mussel tissue prevents the epoxy resin from hardening, and introduce preferential pathways for radon to escape from the sample. For future analysis of mussels via gamma spectrometry, sample masses will be kept below 2.5 grams.

Ingestion doses to the public from ^{226}Ra and ^{210}Pb in mussels from Mudginberri Billabong

Based upon the activity concentrations of ^{226}Ra and ^{210}Pb in mussel flesh and the age distribution of mussels collected, the average annual committed effective dose can be calculated for a 10-year old child who eats 2 kg (wet weight) of mussel flesh from Mudginberri Billabong. The average of all collections from 2000 to 2010 is 0.18 mSv. In 2011, the committed effective dose was slightly lower at 0.14 mSv (Figure 7).

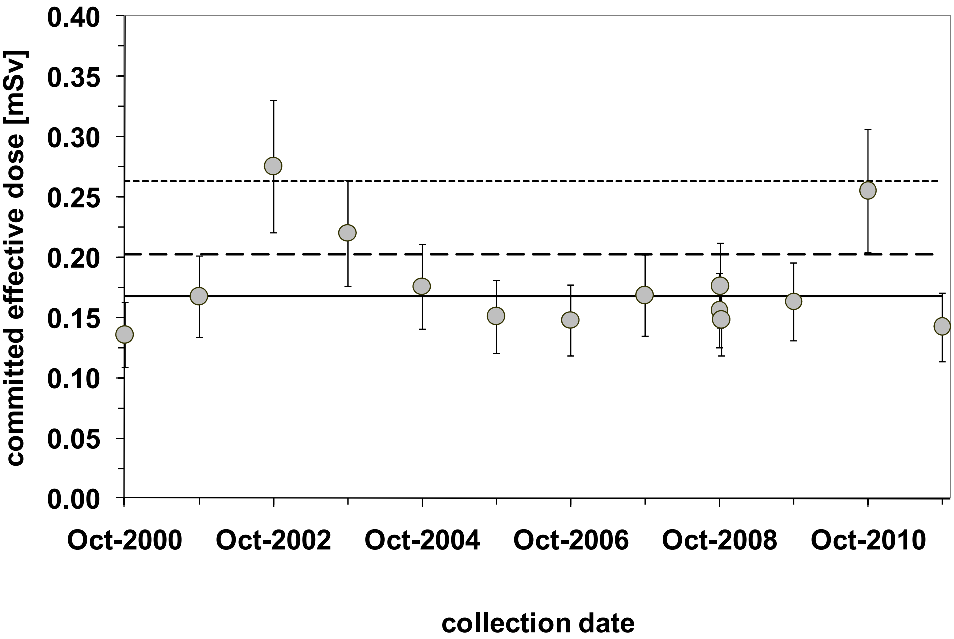

Figure 7 Annual committed effective doses (point data) from ^{226}Ra and ^{210}Pb for a 10 year old child eating 2 kg of mussels collected from Mudginberri Billabong. The median for all the data (solid line), the 80th percentile (dashed line) and 95thpercentile (dotted line) are shown for reference.

This dose is almost exclusively from natural background contributions and would be received irrespective of the operation of Ranger mine. This assertion can be made since: (1) the difference between ^{226}Ra activity concentrations measured in Magela Creek upstream and downstream of the Ranger mine is only very small (see Figure 1, in 'Surface water radiological monitoring in the vicinity of Ranger', this volume, and (2) the findings from previously reported research show that mussel radionuclide activity loads in Mudginberri

Billabong are due to natural catchment rather than mining influences. For Sandy Billabong mussels the inferred ingestion dose for 2011 was 0.06 mSv.

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Monitoring using macroinvertebrate community structure

CL Humphrey, L Chandler, C Camilleri & J Hanley

Background

Macroinvertebrate communities have been sampled from a number of sites in Magela Creek at the end of significant wet season flows, each year from 1988 to the present. The design and methodology have been refined over this period (changes are described in the 2003–04 Supervising Scientist annual report, section 2.2.3). The present design is 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 are collected from each site at the end of each wet season during recessional flows (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 '100%' indicates totally dissimilar communities, sharing no common taxa. Disturbed sites may be associated with significantly higher dissimilarity values compared with undisturbed sites.

Results

Compilation of the full macroinvertebrate dataset from 1988 to 2012 has been completed. Figure 1 shows the paired-site dissimilarity values using family-level (log-transformed) data, for the two 'exposed' streams and the two 'control' streams.

For statistical analysis, dissimilarity values for each of the five possible, randomly-paired, upstream and downstream replicates within each stream are derived. These replicate dissimilarity values may then be used to test whether or not macroinvertebrate community structure has altered significantly at the exposed sites for the wet season of interest. For this multi-factor ANOVA, only data gathered since 1998 have been used. (Data gathered prior to this time were based upon different and less rigorous sampling and sample processing methods, and/or absence of sampling in three of the four streams.)

A four-factor ANOVA model based on replicate, paired-site dissimilarity values, was run using the factors Before/After (BA; fixed), Control/Impact (CI; fixed), Year (nested within BA; random) and Stream (nested within CI; random) to determine if any change has occurred. The ANOVA showed no significant change from the before (pre 2012) to after (2012) periods in the magnitude of upstream-downstream dissimilarity between the control and exposed streams (BA and BA*Stream interaction not significant respectively).

These results confirm that the dissimilarity values for 2012 do not differ from previous years. While the Year*Stream interaction is significant in the same analysis ($p < 0.001$), this simply indicates that dissimilarity values for the streams show natural differences through time. This variation over time is evident in Figure 1, particularly for recent years in Gulungul Creek, and is further considered below.

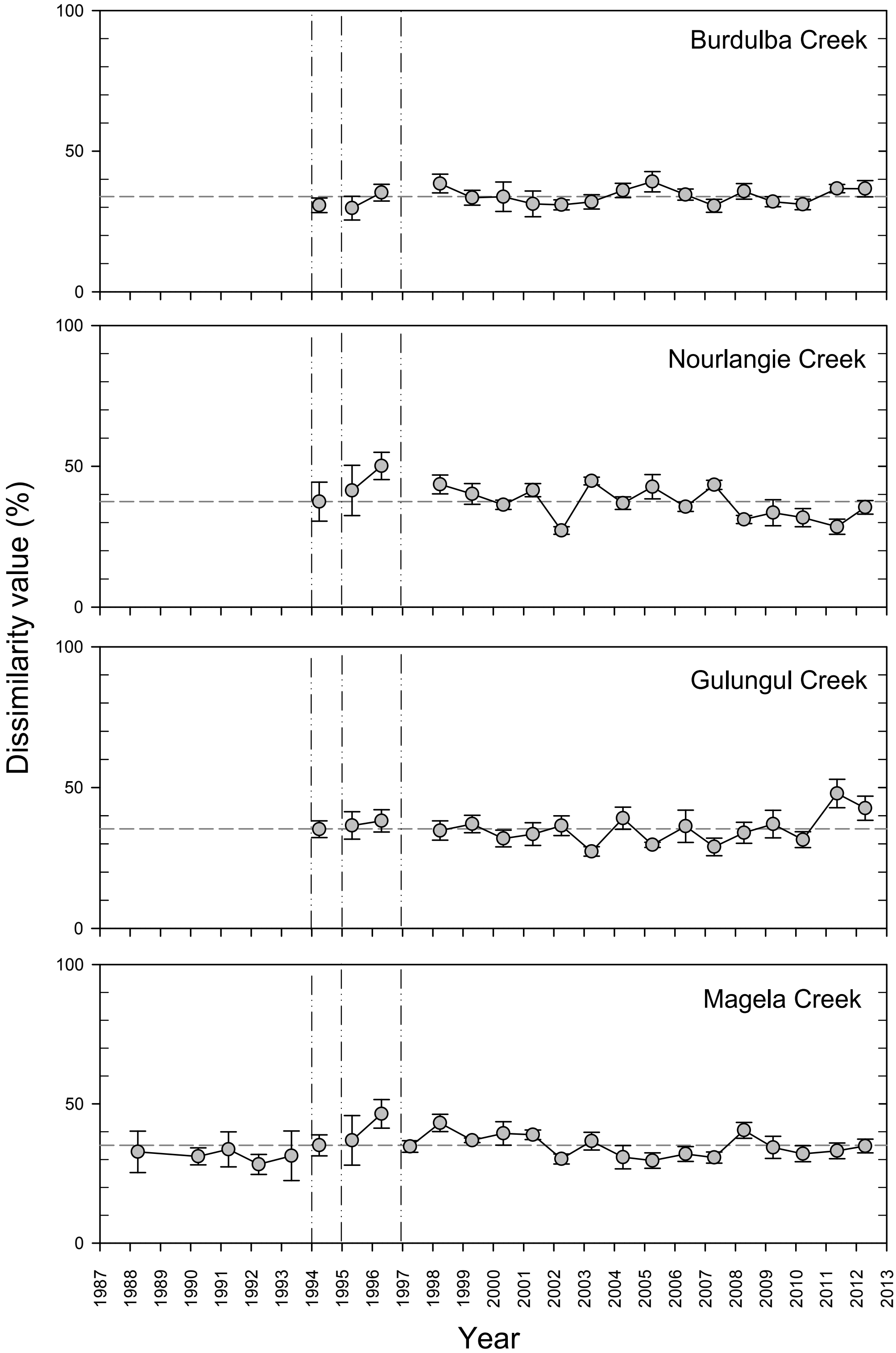

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2012. The dashed vertical lines delineate periods for which a different 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.

In ***eriss***'s Annual Research Summary for 2010–11, a sharp rise in dissimilarity for Gulungul Creek following the 2010–11 wet season was reported (Humphrey et al 2012; see Figure 1 below). Accompanying multivariate analyses showed that the *upstream* Gulungul site in 2011 was significantly different from the before (pre 2010–11) to after (2010–11) periods and that was related to an unusually higher proportion of taxa with a preference for high velocity waters (ie so-termed 'flow-dependent' taxa). The magnitude of paired-site dissimilarity for Gulungul Creek in 2012 has declined from its peak in 2011, but remains elevated compared with data prior to 2011 (Figure 1).). Despite the non-significant before versus after ANOVA result for 2012 (ie neither the BA or BA*Stream interaction significant, from above), the slightly elevated Gulungul dissimilarity has been examined further to ensure mine-related change was not a possible cause.

Figure 2 depicts the multivariate ordination derived using replicate within-site macroinvertebrate data. Data points are displayed in terms of the sites sampled in Magela and Gulungul Creeks downstream of Ranger for each year of study (to 2012), relative to Magela and Gulungul Creek upstream (control) sites for 2012, and all other control sites (Magela and Gulungul upstream sites, all sites in Burdulba and Nourlangie). Samples close to one another in the ordination indicate a similar community structure.

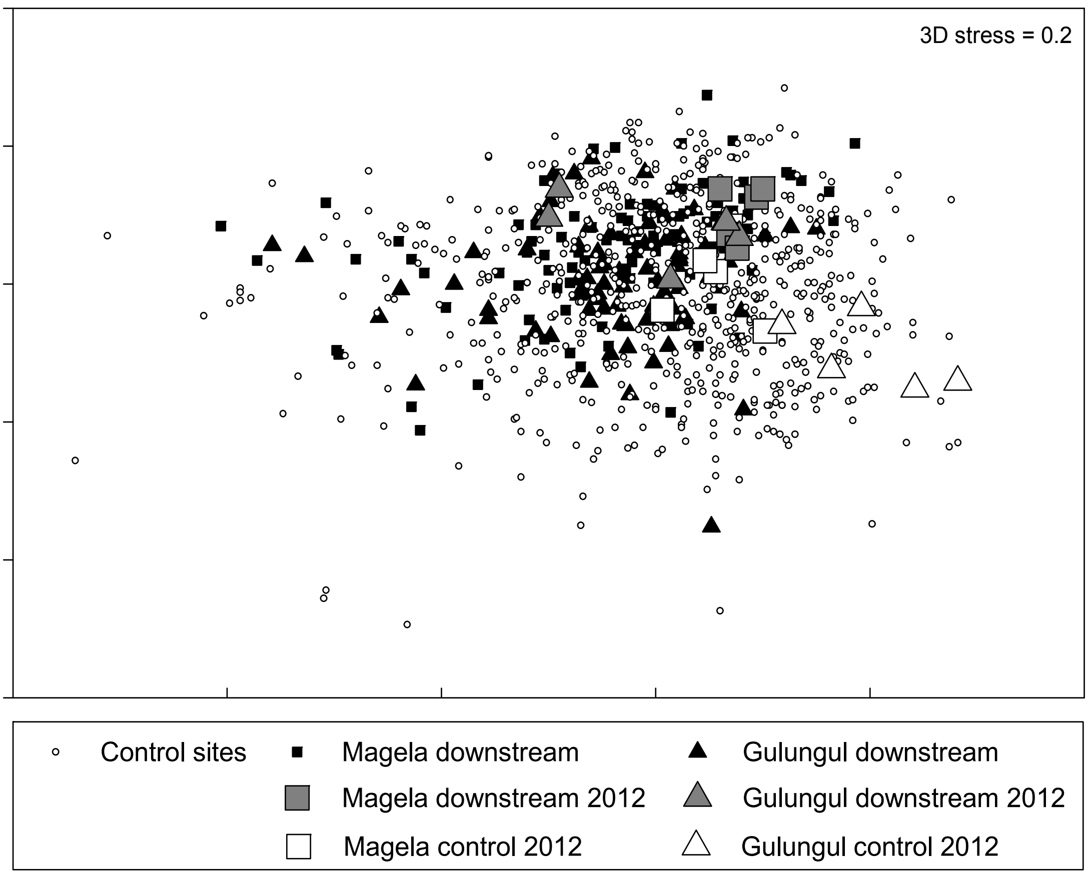

Figure 2 Ordination plot (axis 1 and 2) of macroinvertebrate community structure data from sites sampled in several streams in the vicinity of Ranger mine for the period 1988 to 2012. Data from Magela and Gulungul Creeks for 2012 are indicated by the enlarged symbols.

Examination of the three-dimensional ordinations for all paired axes combinations (results not shown) indicated that Gulungul Creek communities from the upstream site still differed in 2012 from other sites and times (Figure 2). However, the separation of the upstream replicate values was not as marked as that observed in 2011 (see Humphrey et al 2012, Figure 2). Data points associated with the 2012 Gulungul and Magela downstream sites are generally interspersed among the points representing the control sites, indicating that these 'exposed' sites have macroinvertebrate communities that are similar to those occurring at control sites.

A five-factor PERMANOVA (PERmutational Multivariate ANalysis Of Variance) was conducted on all replicate data gathered from all sites from 1998 to 2012 to determine if a priori groups, exposure type ('exposed' Magela and Gulungul Creeks versus control Burdulba and Nourlangie Creeks) and site location (upstream versus downstream) and the interaction between these two factors, show significant differences. Unlike the ANOVA performed on paired-site dissimilarities, PERMANOVA analyses data from *within* sites, hence an additional (fixed) factor, location (upstream-downstream). This has the advantage that, in the case of a change in paired-site dissimilarity in one or both of the exposed streams, it can be identified whether or not the change is associated with the downstream, exposed site(s) and thereby trigger further investigation to determine whether it could be mine-related. PERMANOVA showed no significant difference for both factors and their interaction ($P > 0.05$).

The PERMANOVA showed a significant result for the Stream(CI)*Upstream/Downstream interaction ($P = 0.0001$), which indicates that across years for streams within either, or both, exposure types, the changes are not consistent between the up- and downstream sites. This result combines all years for each site together (ie no before vs after). A significant result may indicate that the exposed downstream sites differed greatly or more so than control sites, and therefore requires further investigation. A pair-wise comparison was undertaken to determine the nature of the significant difference. These tests showed that this interaction was significant across all streams and is therefore unrelated to mining.

Collectively, these graphical and statistical results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining during the period 1994 to 2012 have not adversely affected macroinvertebrate communities.

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Monitoring using fish community structure

D Buckle & CL Humphrey

Assessment of fish communities in billabongs is conducted between late April and July each sampling year, the precise time of the monitoring being dependent on flow regime, using non-destructive sampling methods at 'exposed' and 'control' locations. Two billabong types are sampled: deep channel billabongs every year, and shallow lowland (mostly backflow) billabongs dominated by aquatic plants every two years. Details of the sampling methods and sites were provided in the 2003–04 Supervising Scientist annual report (Supervising Scientist annual report 2004, chapter 2, section 2.2.3). These programs were reviewed in October 2006 and the refinements to their design, respectively, are detailed in Buckle and Humphrey (2008, 2009).

For both deep channel and shallow lowland billabongs, comparisons are made between a directly-exposed billabong in the Magela Creek catchment downstream of the Ranger mine versus control billabongs from an independent catchment (Nourlangie Creek and Wirnmuyurr Creek). The extent of 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 in the 'Monitoring using macroinvertebrate community structure' section, above. 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) to those of Sandy Billabong (control site in the Nourlangie Creek catchment) (see Map 3) are determined using multivariate dissimilarity indices calculated for each annual sampling occasion. A plot of the dissimilarity values from 1994 to 2012 is shown in Figure 1.

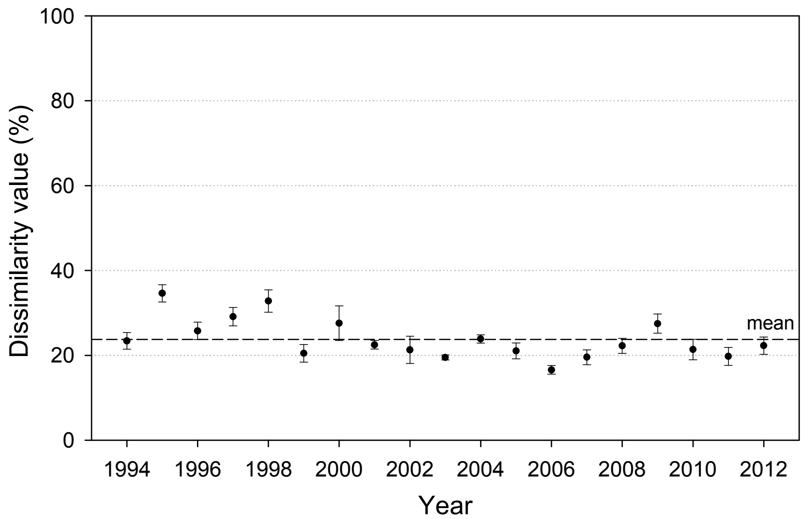

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of the Ranger Mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

The paired-billabong dissimilarity values have been analysed using a two-factor ANOVA (ANalysis Of VAriance), with Before/After (BA; fixed) and Year (nested within BA; random) as factors. In this analysis the 'BA' factor tests whether values for the year of interest (2012) are consistent with the range of values reported in previous years (1994 to 2011) while the factor 'Year' tests for differences amongst years within the before or after periods. The ANOVA results showed no significant difference between 2012 and other years (BA factor not significant, $p = 0.758$), indicating the relationship between Mudginberri and Sandy Billabong fish communities has remained consistent with relationships observed in previous years. However, the variation in fish assemblage dissimilarities between the two billabongs amongst years (tested by factor Year) was significantly different over the 1994 and 2012 period ($p < 0.001$). This variation over time is evident in Figure 1 and is further considered below.

In previous reports, possible causes of trends in the annual paired-site dissimilarity measured over time have been advanced and assessed. Because the dissimilarity measure is most influenced by numerically-abundant fish species, it was possible to demonstrate that fluctuations in the measure over time were directly associated with longer-term changes in abundance in Magela Creek of the chequered rainbowfish (*Melanotaenia splendida inornata*), the most common fish species in this creek system (Supervising Scientist annual report 2004, chapter 2, section 2.2.3). Thus, effort has been directed at understanding the possible causes of interannual variations in the abundance of this fish species in Magela Creek.

In previous Supervising Scientist annual reports, negative correlations between annual rainbowfish abundance in Mudginberri Billabong and the magnitude of wet season discharge (total for the wet season, January total and February total, GS8210009) have been observed in Magela Creek. The negative relationships between rainbowfish in Mudginberri Billabong and wet season conditions have been further tested using rainfall data (Jabiru airport records). Rainfall data have been used in place of discharge data because it is considered more representative of regional wet season conditions. The results support those from previous years with negative relationships observed between rainbowfish abundance in Mudginberri Billabong and both the total annual rainfall ($p = 0.019$) and the rainfall total for January ($p = 0.039$).

To this end, the results from 2012 continue to support previous suggestions that reduced rainbowfish abundances occur after larger wet season rainfalls which potentially allow greater upstream migration of rainbowfish past Mudginberri Billabong, thereby reducing the concentration of fish in the billabong during the recessional flow period. The relationship between annual rainfall and rainbowfish abundance in Mudginberri Billabong can be visualised in Figure 2.

Collectively, the analyses described above provide good evidence that changes to water quality downstream of Ranger as a consequence of mining during the period 1994 to 2012 have not adversely affected fish communities in channel billabongs.

Shallow lowland billabongs

Monitoring of fish communities in shallow lowland billabongs has previously been conducted every two years (Buckle & Humphrey 2008). The last assessment of fish communities in these billabongs occurred in May 2009 with results reported in Buckle et al (2010). The scheduled sampling of fish communities in 2011 was postponed to enable staff resources to be dedicated to an intensive sampling of other biota (phytoplankton, zooplankton and macroinvertebrate communities) in these billabong habitats. Results from sampling conducted in June 2012 are discussed below.

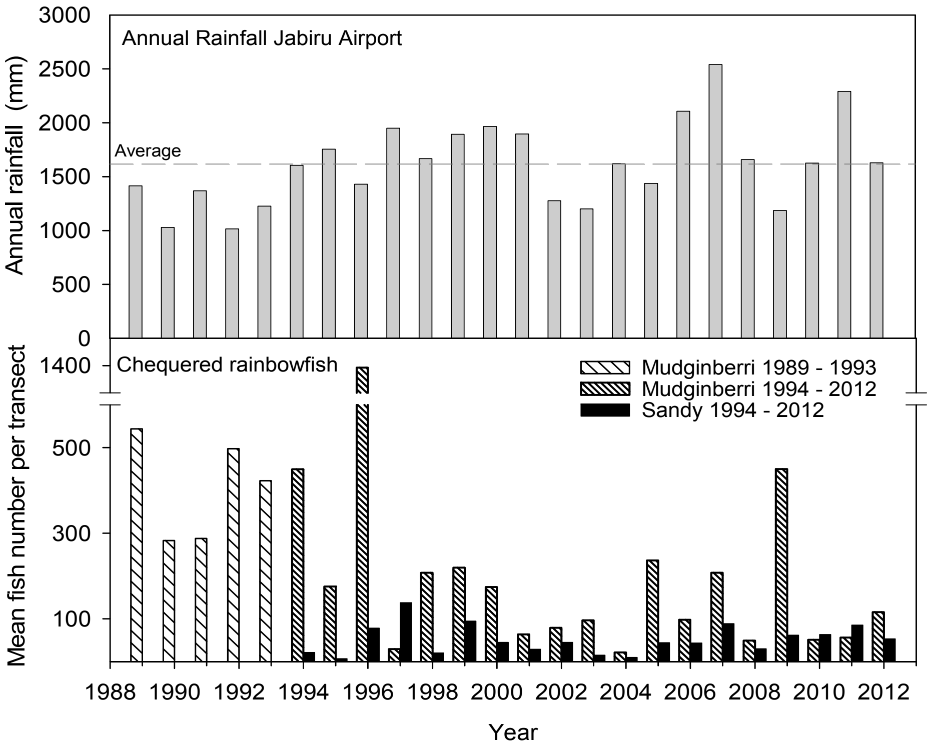

Figure 2 Relative abundance of chequered rainbowfish in Mudginberri and Sandy Billabongs from 1989 to 2012 with associated total discharge in Magela Creek (gauging station G8210009)

The monitoring program for fish communities in shallow billabongs is conducted 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 Shallow and Buba billabongs on Nourlangie Creek and Wirnmuyurr Billabong – a Magela floodplain tributary) (see Map 3) is 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 Shallow and Gulungul-Wirnmuyurr Billabongs – from 1994 to 2012, is shown in Figure 3.

The three sets of paired-billabong dissimilarity values measured since 1998 (when sampling of all three site-pairs commenced) have been analysed using a three-factor ANOVA with Before/After (BA; fixed), Year (nested within BA; random) and Site-pair (Fixed) as factors. In this analysis the BA factor tests whether values for the year of interest (2012) are consistent with the range of values reported in previous years (1998 to 2011), the factor 'Year' tests for differences amongst years within the before or after periods and the 'Site-pair' factor tests for differences amongst the three paired-billabong dissimilarities. The ANOVA results showed that across all three site-pairs there was no significant change from 2012 to other years (BA factor, $p = 0.508$) and that the change between 2012 and previous years within the individual site-pairs was consistent (BA*Site-pair interaction, $p = 0.433$). These results confirm that dissimilarity values for 2012 for all three site-pairs do not differ from those values from previous years. Significant differences do occur over time within site-pairs (Year*Site-pair interaction, $p = 0.001$) which reflects (natural) changes through time. This variation over time is evident in Figure 3 and is further considered below.

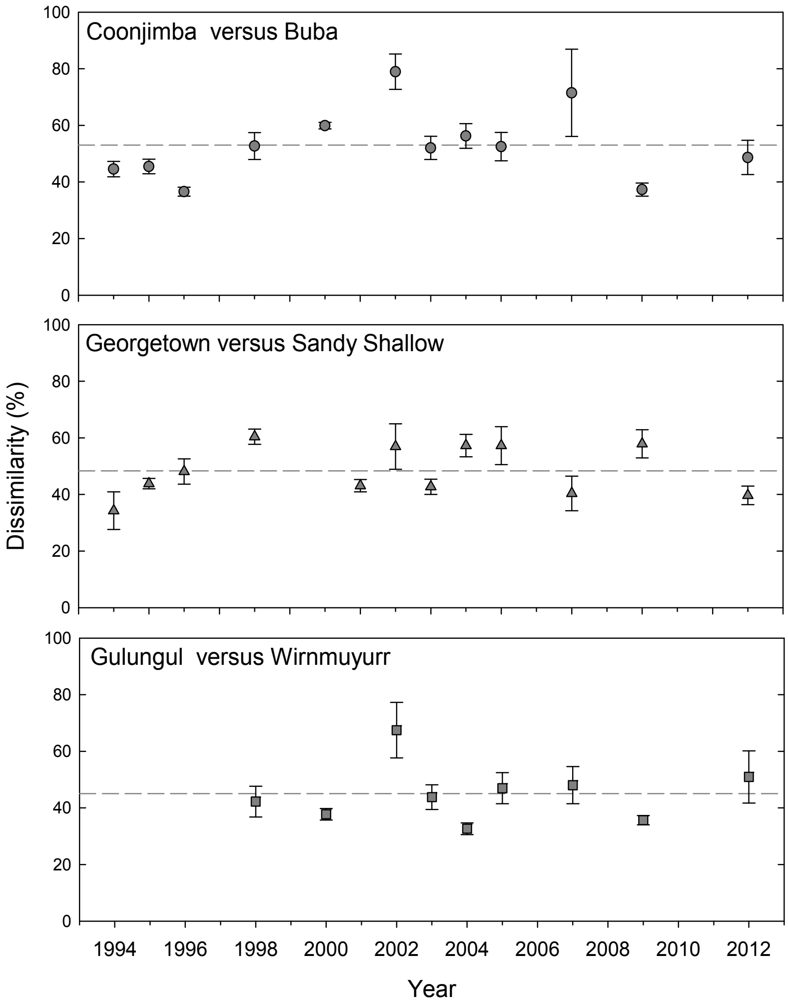

Figure 3 Paired control-exposed site dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in 'exposed' Magela and 'control' Nourlangie and Magela Billabongs in the vicinity of Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of average trap enclosure data between the pairwise billabong comparisons, Coonjimba-Buba, Gulungul-Wirnmuyurr and Georgetown-Sandy Shallow 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-pair are quite different from one another. The dissimilarity values appear to reflect differences in aquatic plant communities of the site-pair billabongs. Buckle and Humphrey (2008) noted 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 influences of aquatic plant communities in billabongs and their influence in turn over fish communities are being further assessed.

Collectively, the graphical and statistical results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining during the period 1994 to 2012 have not adversely affected fish communities in shallow billabongs.

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Ranger stream monitoring research: Analyses of toxicity monitoring and associated water quality data for Magela and Gulungul Creeks

C Humphrey, M Ellis, R van Dam & A Harford

Background and previous findings

Prior to reading this summary, it is advisable to read both the introduction and results sections of the routine toxicity monitoring for the 2011–12 wet season paper reported above (see 'Toxicity monitoring in Magela and Gulungul creeks', this volume) to provide context for the research and development project outlined below.

Toxicity monitoring evaluates the responses of aquatic animals exposed in situ in Magela and Gulungul Creeks to diluted runoff water from the Ranger minesite. Egg production by the freshwater snail, *Amerianna cumingi*, over a four day deployment period, has been the method used in Magela Creek since 1990–91 and in Gulungul Creek since 2009–10 (see 'Toxicity monitoring in Magela and Gulungul Creeks', this volume).

Following the 2010–11 wet season, analyses were undertaken to provide an improved understanding of environmental (viz water quality) conditions affecting the production of snail eggs during the toxicity monitoring tests (Humphrey et al 2012). This work was necessary to ensure that it is possible to distinguish between natural and mine-induced effects on snail egg numbers between upstream and downstream sites.

For toxicity monitoring data from Magela and Gulungul Creeks acquired between the 2006–07 and 2010–11 wet season (where corresponding continuous electrical conductivity (EC, a reliable surrogate of magnesium sulfate concentrations), water temperature and turbidity data are available), a number of significant correlations and interactions were found between mean egg number and both median EC and water temperature, but not turbidity:

1. A positive linear relationship was observed between EC and snail egg number.
2. A unimodal (second-order polynomial or quadratic) relationship was found between water temperature and snail egg number, with a peak in egg number observed near 29°C.
3. A significant interacting effect of water temperature and EC upon snail egg counts in Magela and Gulungul Creeks was observed. Plots of EC and snail egg number (for Magela and Gulungul sites combined) were prepared using one degree increments in median water temperature. The changing relationship of the snail reproduction response to EC with rising water temperature was noted, with enhanced egg production with increasing EC at lower water temperatures (27–29°C), an increasingly neutral effect at intermediate temperature (~30°C) and an increasingly reduced/negative effect at higher water temperatures (>30°C).

In the previous (2010–11) Supervising Scientist Annual Report it was also noted that trapping of suspended organic matter in the capsules containing the snails could possibly have produced a stimulatory effect on downstream egg production by virtue of providing a supplementary food source. To address this issue the detrital material accumulating in the snail containers in both Magela and Gulungul Creeks during the 2010–11 wet season was collected and analysed for its content of inorganic and organic matter. Suspended inorganic

(SIM) and organic (SOM) matter were both found to be higher at the upstream sites of both creeks with no strong relationships found between these variables and mean snail egg number measured in each creek. This work was repeated in the 2011–12 wet season.

Toxicity monitoring data from the 2011–12 wet season have been combined with the in situ toxicity monitoring data from previous wet seasons and the combined dataset reanalysed with associated water quality data (including SIM and SOM) to assess previous findings and conclusions.

Analyses following 2011–12 wet season

Correlates of variability in snail egg difference values

For the 2009–10 and 2010–11 wet seasons, greater variability was noted between upstream and downstream sites in the egg counts from Gulungul Creek compared with the same response measured in Magela Creek (see Figure 1B of Humphrey et al, 'Toxicity monitoring in Magela and Gulungul creeks', this volume). This higher Gulungul variability corresponded to similar and generally more variable (compared with Magela Creek) water quality in Gulungul Creek, attributed to the greater proportional influence of runoff to Gulungul Creek from catchment sources between the upstream and downstream sites in this relatively small drainage basin.

This water quality-biological variability relationship was examined more closely for Gulungul and Magela Creeks, and with the addition of 2011–12 wet season data. Water quality (temperature, EC, turbidity) differences were calculated from the medians of upstream and downstream values measured at a 10 minute frequency over each of the four-day tests conducted over the three (Gulungul) or six (Magela) wet seasons for which in situ toxicity and continuous (water quality) monitoring data were available. The standard deviation of the four-day upstream-downstream difference values for the continuously-monitored water quality variables and snail egg numbers in both Gulungul and Magela Creek were then derived for each wet season. Correlation analysis was conducted between the standard deviation of annual egg difference values and standard deviation of water quality (temperature, EC, turbidity) differences values for the data from both creeks combined.

The only significant water quality correlate of annual egg difference variability was electrical conductivity (EC), as depicted in Figure 1. Higher variability in upstream-downstream EC differences in both Gulungul and Magela Creeks is associated with significant mine-water discharge events in a particular wet season, most evident in Magela Creek in the 2006–07 wet season and, for the period of toxicity monitoring in Gulungul Creek, in the 2009–10 wet season. Given the significant relationship depicted in Figure 1, these specific mine-water discharge events are implicated in changes to the paired-site, egg number difference values, even though there may be no significant difference in overall mean snail egg difference values between wet seasons. The nature of the egg production response to EC is discussed in the following section.

Effect of EC and water temperature upon snail egg production

Toxicity monitoring results from Magela and Gulungul Creeks for the 2011–12 wet season were combined with previous toxicity monitoring data to determine the consistency of the recent data with previous findings of (i) water temperature-EC interacting effect upon snail egg number (ie changing relationship of the snail reproduction response to EC with rising water temperature), and (ii) a positive linear relationship between EC and snail egg number.

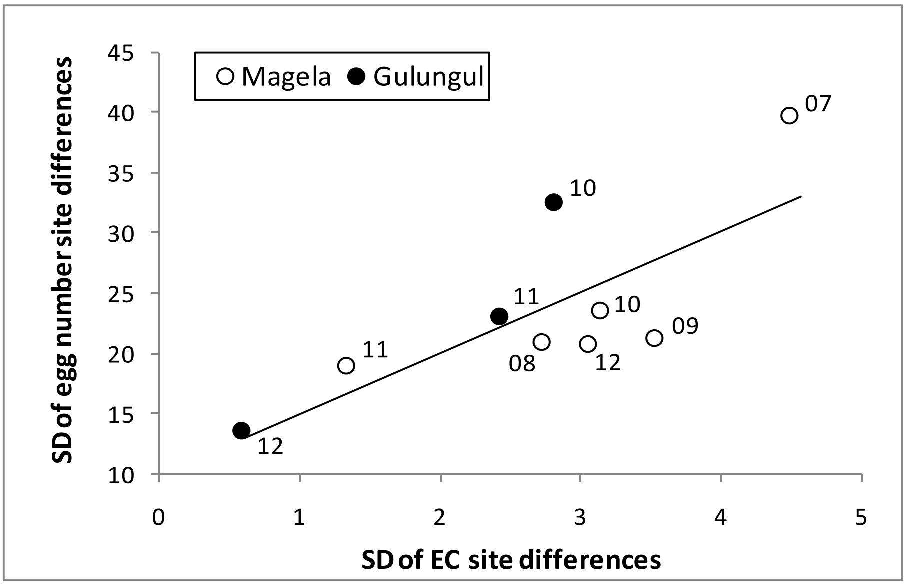

Figure 1 Regression relationship between standard deviations (SD) of upstream-downstream difference values for snail eggs numbers and electrical conductivity (EC) in both Magela (2006–07 to 2011–12) and Gulungul Creek (2009–10 to 2011–12) wet seasons. Subscript numbers against symbols refer to year (eg 07 = 2006–07 wet season).

The water temperature-EC interacting-effect upon snail egg number continued with addition of 2011–12 data. While the plots of EC versus egg number with incremental (1°C) increases in water temperature are not included here, the slope of the trend lines for the egg number-EC relationships are plotted against water temperature increments in Figure 2. Positive and negative regression slopes indicate enhanced and suppressed effects of EC upon snail egg production, respectively.

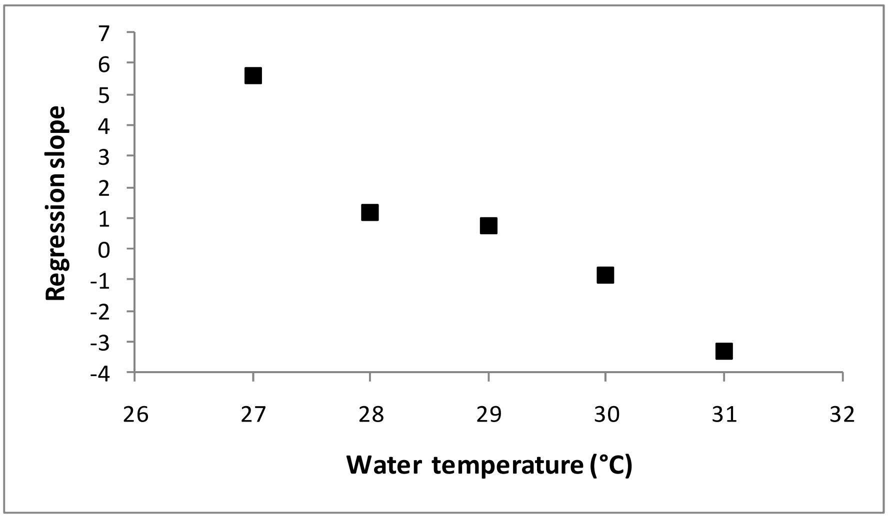

Figure 2 Relationship between slope of the regression relationships between mean snail egg number and median EC for toxicity monitoring tests conducted in both Magela and Gulungul creeks, 2007 to 2012, and median water temperature

Nevertheless, over and above the interacting effect of water temperature and EC upon snail egg counts, for the range of median (four-day) EC values recorded in the creeks (between 7 and 30 µS/cm), a positive non-linear relationship is observed, and has continued, between EC and snail egg numbers (Figure 3). This suggests a net, albeit small, enhancement to snail egg production with increasing additions of mine-derived solutes to Magela and Gulungul Creeks over the range of median EC values observed.

Relationships between snail egg response and suspended inorganic and organic matter

Very few significant relationships were found between SOM and SIM and mean snail egg number when considering Magela and Gulungul creeks, and years (2011 and 2012), separately and combined. Most correlations between SIM and mean snail egg number were negative while most between SOM and mean snail egg number were positive. However, it is unlikely that this result indicates possible inhibition and enhancement by SIM and SOM, respectively, upon snail reproduction given that SOM values were actually higher at the upstream sites in both years, whereas snail egg numbers were generally lower at the upstream sites (see Figure 1 of Humphrey et al, 'Toxicity monitoring in Magela and Gulungul Creeks', this volume). Thus, the hypothesis that SOM contributes to higher downstream egg production is not supported by the data. Few significant correlations were found between SOM and SIM and either EC or water temperature when considering Magela and Gulungul Creeks, and years (2011 and 2012), separately and combined.

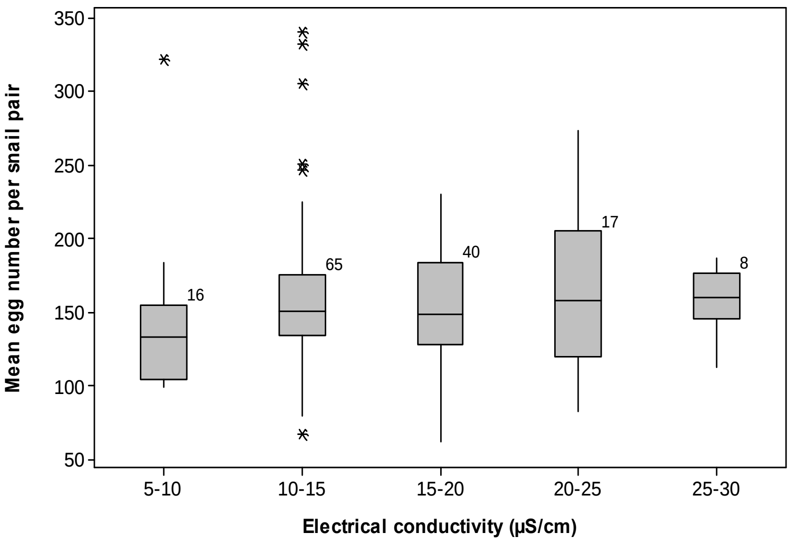

Figure 3 Box plots of mean snail egg number observed in in situ toxicity monitoring tests in Gulungul and Magela Creeks, 2007–2012, grouped according to ambient electrical conductivity in creek waters. Box plots show median, range, 25th and 75th percentiles. Points at a greater distance from the median than 1.5 times the Inter-Quartile Range are plotted individually as asterisks ('outliers'). Subscript numbers against boxes denote number of toxicity monitoring tests.

Conclusions

Reproductive responses of freshwater snails exposed in Magela and Gulungul Creeks in the wet season appear to be stimulated by small increases in EC across the range of median (four-day) values recorded in these receiving waters (7–30 µS/cm, Figure 3). Median EC values greater than 20 µS/cm are a consequence of mine water discharges, and so inputs of water from the minesite are implicated in at least part of the stimulatory response represented in Figure 3. In these very low solute streams, it is unsurprising that freshwater organisms would benefit from small additions of ions, including mine-derived magnesium sulfate. (Magnesium is an essential element in biological systems.)

There is some debate in the literature about the ecological significance of stimulatory effects of potential toxicants (especially for those that are essential ions or trace nutrients at low levels) at concentrations below those eliciting negative and adverse toxic effects. However, as an essential ion, and for chronic exposures at such very low concentrations, it is unlikely that Mg is resulting in ecological harm in the receiving waters of Magela Creek downstream of the Ranger mine. In support of this assertion, recessional flow sampling of macroinvertebrate and fish communities in Magela and Gulungul creeks has provided no evidence that changes to water quality downstream of Ranger as a consequence of mining during the period 1994 to 2012 have altered or adversely affected these communities (see Humphrey et al, 'Monitoring using macroinvertebrate community structure', this volume; Buckle & Humphrey, 'Monitoring using fish community structure', this volume, respectively. Nevertheless, the issue of Mg-related stimulation warrants further study through literature review and corroborative laboratory studies. To this end, initial laboratory investigations upon freshwater snails (possible EC and temperature interaction effect upon egg production) have commenced with results reported below.

A laboratory experiment was carried out in the ***eriss*** Darwin ecotoxicology laboratory in October 2012 to examine snail egg production using two water temperatures (27 and 32°C) and three Mg concentrations (equivalent to ECs of 10, 20 and 30 µS/cm). If there was to be concordance between field and laboratory observations, then these treatments might be expected to show, inferring from Figure 2 above, (i) increasing egg production with increasing EC at 27°C, and (ii) decreasing egg production with decreasing EC at 32°C. The experiment used an in-house protocol (Houston et al 2007), similar to the field protocol. This involves, exposing replicate pairs of snails for four days to Magela Creek water collected from upstream of the Ranger mine.

Results of the laboratory experiment are provided in Figure 4.

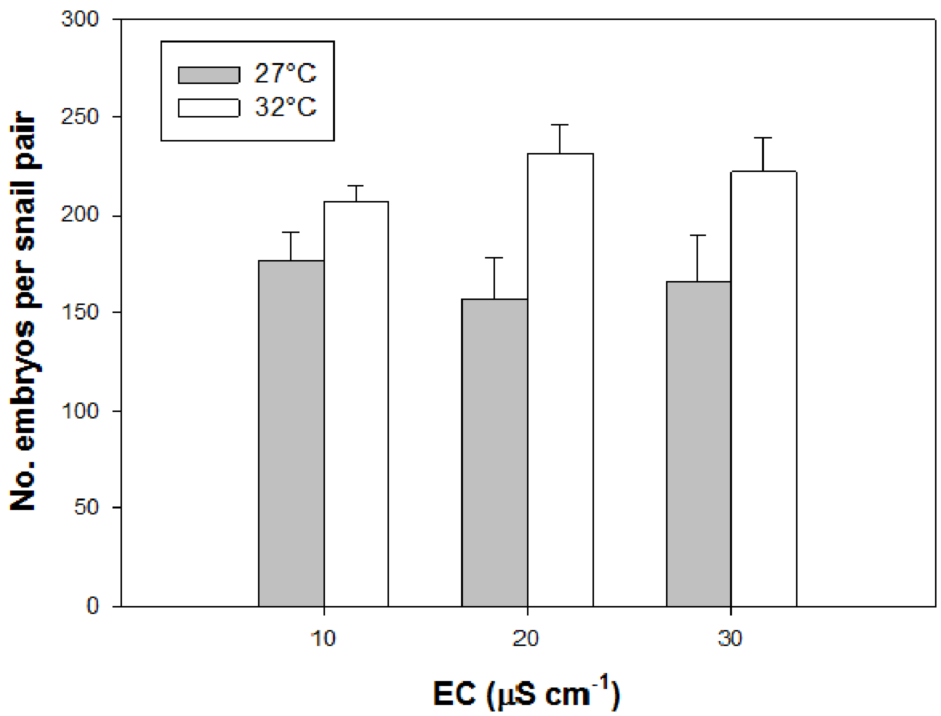

Figure 4 Effect of temperature and its interaction with EC on *Amerianna cumingi* egg production in a laboratory experiment, October 2012. Data represent mean ± SE where N for laboratory tests is 3 replicates, each of six snail pairs, per treatment.

A two-factor (EC and temperature) PERMANOVA (Anderson et al 2008) showed a significant difference in water temperature only, ie significantly greater egg production at 32°C; there was no significant EC nor EC-temperature interaction effect. Thus the laboratory study did not support the field observations. An assessment is yet to be made as to the adequacy of the experimental design for the laboratory experiment, and whether further improvements may be made that warrant a repeat of the experiment.

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Part 2: Ranger – rehabilitation

Assessing runoff, soil erosion and solute losses from the trial landform

MJ Saynor & WD Erskine

Introduction

Saynor et al (2012a,b) reported the results for rainfall and bedload (2009–2010 & 2010–2011 water years) from a long-term project to assess runoff, sediment and solute losses from a trial rehabilitation landform at the Ranger mine. ERA constructed the trial landform at the end of 2008 with the objective of testing over the long-term proposed landform design and revegetation strategies for the site, such that the most appropriate one can be implemented when the site is decommissioned (Saynor et al 2009, 2010, 2011). SSD is leading the erosion assessment project, and providing most of the staff resources, with a substantial level of field assistance and collaboration also being provided by technical staff from ERA.

The trial landform was designed to test two types of potential final cover layers for the rehabilitated mine landform: waste rock alone; and waste rock blended with approximately 30% v/v of fine-grained weathered material (laterite). In addition to two different types of cover materials, two different planting methods were originally going to be assessed: direct seeding and tube stock (Saynor et al 2009). Plots 1 and 4 were planted with tube stock in March 2009 with infill planting to replace dead specimens in January 2010. Plots 2 and 3 were direct seeded in July 2009. However, because of poor germination, these plots were infill planted with tube stock in January 2011. The trial landform surface was ripped on the contour before the erosion plots were constructed (Saynor et al 2009). Over the past three years eroded material has been washed into the rip lines but there is still a large amount of potential sediment storage that can occur before the rip lines are obliterated.

The locations of SSD's four erosion plots (approximately 30 m × 30 m) constructed during the 2009 dry season (Saynor et al 2009) are shown on Figure 1.

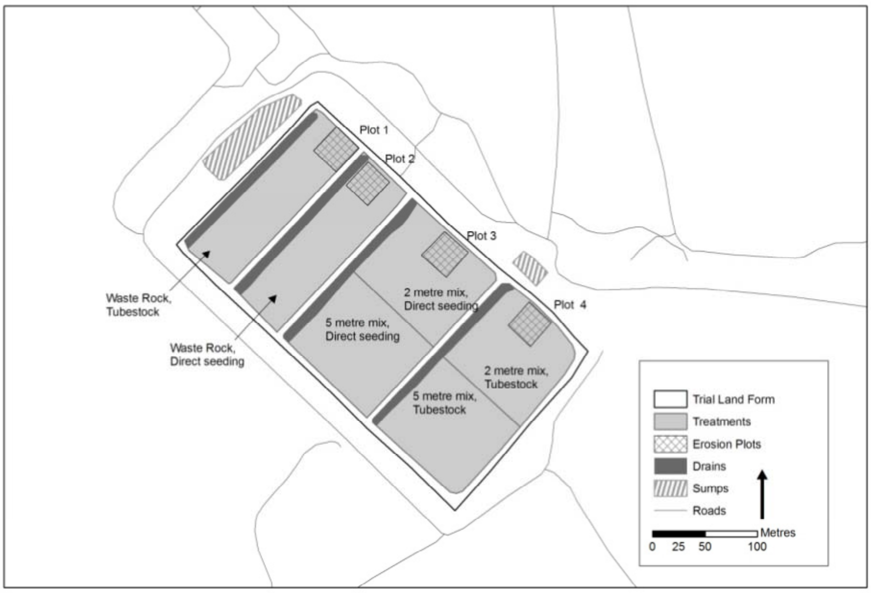

Figure 1 Layout of the plots on the trial landform

Erosion plots 1 and 2 contain waste rock, and erosion plots 3 and 4, mixed waste rock and laterite. The plots were physically isolated from runoff from the rest of the landform by raised borders.

Each erosion plot is instrumented with a range of sensors that were described in detail in Saynor et al (2012). In summary, these include: a tipping bucket rain gauge, a primary shaft encoder with a secondary pressure transducer to measure stage height; a turbidity probe to measure suspended sediment concentration; electrical conductivity probes located at the inlet to the stilling basin and at the entry to the flume to provide a measure of the concentration of dissolved salts in the runoff; an automatic pump sampler to collect event based water samples; a data logger with mobile phone telemetry connection and a rectangular broad-crested flume to accurately determine discharge from the plots.

The latest results from the trial landform project are described below.

Rainfall, runoff and bedload

Overview

Preliminary sediment and solute losses from the four erosion plots were presented for the first year of monitoring (2009–10 water year) in Saynor et al (2011). A water year extends from the driest month for 12 consecutive months, instead of being represented by a calendar year. Water years are used because the use of a calendar year would inappropriately combine data from two different wet seasons. This is because the wet season in the ARR typically extends over a six to seven month period from late October in one year to the end of April in the next. A 'water year' has been defined as the period from September in the first year to August in the next. Rainfall for all four plots and runoff from erosion plot 1 is reported here. Plot 1 is the only one for which runoff has been calculated to date for all three water years, 2009–10, 2010–11 and 2011–12.

Sediment is transported by flowing water as either suspended load or bedload. The results of the bedload measurements will be reported again this year because the suspended sediment and solute data are still being analysed.

Bedload samples were collected at weekly to monthly intervals during the wet season, depending on event magnitude and staff availability or on demand following isolated large rainfall events. The collected samples were transported to the ***eriss*** laboratory in Darwin for processing. The bedload grain size results for the first three years are also reported.

Rainfall and runoff results

The rainfall data for each plot for each water year are contained in Table 1 and the runoff data for plot 1 are summarised in Table 2.

Table 1 Rainfall data for the four erosion plots on the trial landform for the three years of measurement

Water year	Erosion Plot 1 Rainfall (mm)	Erosion Plot 2 Rainfall (mm)	Erosion Plot 3 Rainfall (mm)	Erosion Plot 4 Rainfall (mm)	Mean Annual Rainfall ± Standard Error (mm)
2009–10	1533	1531	1480	1528	1518 ± 13
2010–11	2227	2290	2205	2296	2255 ± 23
2011–12	1508	1531	1456	1489	1496 ± 16

Mean annual rainfall at Jabiru Airport (Station No. 014198) is 1576 mm[1] (Bureau of Meteorology). This BoM station is located 2.3 km from the trial landform and has an incomplete record for the period 1971–2012. Furthermore, the record since 1993 is not quality controlled.

Table 2 Rainfall and runoff data for erosion plot 1 on the trial landform for the three years of measurement

Water year	Maximum event rainfall (mm)	Number of runoff events	Runoff (L)	Runoff (mm)	Runoff coefficient (%)
2009–10	76.6	135	74612	81	5.3
2010–11	189.4	210	275748	300	13.5
2011–12	58.0	152	96991	106	7.0

The annual rainfall for the 2011–12 water year on the trial landform was similar to the 2009–10 water year (Table 1), and both were slightly less than average at Jabiru Airport. On the other hand, the annual rainfall for the 2010–11 water year was much greater than for the other two years (Table 1), and was comparable to the 95th percentile annual rainfall at Jabiru Airport.

The discharge data from plot 1 over the first three years are summarised in Table 2. The number of discrete runoff events is very large and reflects the rapidly responsive nature of the plots to small rainfall events. The number of runoff events that produced discharge over the crest of the weir was less in the first year after construction (2009–10) and was greatest in the wettest year (2010–11). Unusually, annual runoff was less in the 2009–10 water year than in the drier 2011–12 year (Table 2). This is most likely the result of the infilling with water of the initially empty pore space in the waste rock and laterite from which the trial landform was constructed. Annual runoff was greatest in the wettest year (2010–11) when 13.5% of rainfall was converted to runoff, and was least in 2009–10 when the trial landform was wetting up (Table 2). As expected for small areas (Pilgrim et al 1982; Pilgrim 1983), the runoff coefficient for plots is much less than for larger catchments in the ARR. We believe that areas up to about 1 km^2 behave as if they are drylands in the seasonally wet tropics.

There is a close curvilinear relationship between event rainfall and event runoff over the full range of rainfall for all three years for plot 1. Figure 2 shows this curvilinear relationship between total event rainfall and runoff for all 152 events on plot 1 for 2011–12. When event rainfall exceeds 30 mm there is proportionately greater runoff than for smaller events (Figure 2). These smaller events do not totally infill the rip lines with water and so runoff is only produced from a small part of the plot near the down slope border. Event rainfall greater than 30 mm can totally infill surface storage, hence generating runoff from the whole plot surface. Therefore, rainfall events approaching 30 mm represent a hydrologic threshold for the erosion plots and larger events produce proportionately greater runoff. Therefore, from an erosion perspective, the events greater than 30 mm are of greatest concern.

Bedload results

The annual bedload yields recorded for each plot for each water year are shown in Table 3. Annual bedload yields have declined greatly since construction (Table 3). Time since

[1] www.bom.gov.au/jsp/ncc/cdio/weatherData/av?p_nccObsCode=139&p_display_type=dataFile&p_startYear=&p_c=-47599587&p_stn_num=014198, accessed 11 July 2012

construction has had a much greater effect than cover material type and development of vegetation to date (Table 3). Major land disturbances, such as construction, initiation of gully erosion or occurrence of landslides, are usually characterised by an initial pulse of high sediment yield followed by a rapid decline in sediment yield (Duggan 1994, Erskine 2005). The nature of sediment yield decline following the initial large pulse has been reported as either exponential (Graf 1977) or logarithmic (Erskine 2005) or linear (this paper) (Figure 3).

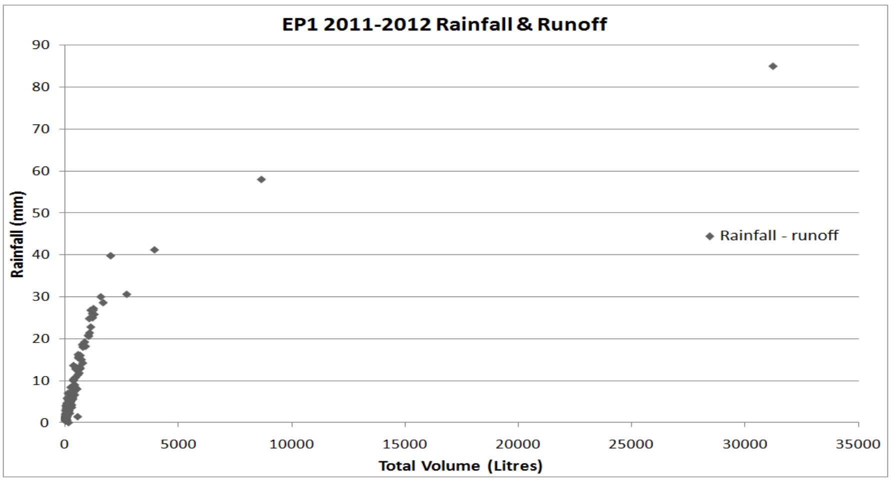

Figure 2 Relationship between total event rainfall and runoff for plot 1 for every runoff event in the 2011–12 water year

Table 3 Annual bedload yields for each plot on the trial landform for each year of measurement

Water year	Erosion Plot 1 Bedload Yield (t/km^2.yr)	Erosion Plot 2 Bedload Yield (t/km^2.yr)	Erosion Plot 3 Bedload Yield (t/km^2.yr)	Erosion Plot 4 Bedload Yield (t/km^2.yr)	Mean Annual Bedload Yield ± Standard Error (t/km^2.yr)
2009–10	106	147	111	143	127 ± 12
2010–11	59	113	54	56	71 ± 16
2011–12	34	48	38	15	34 ± 8

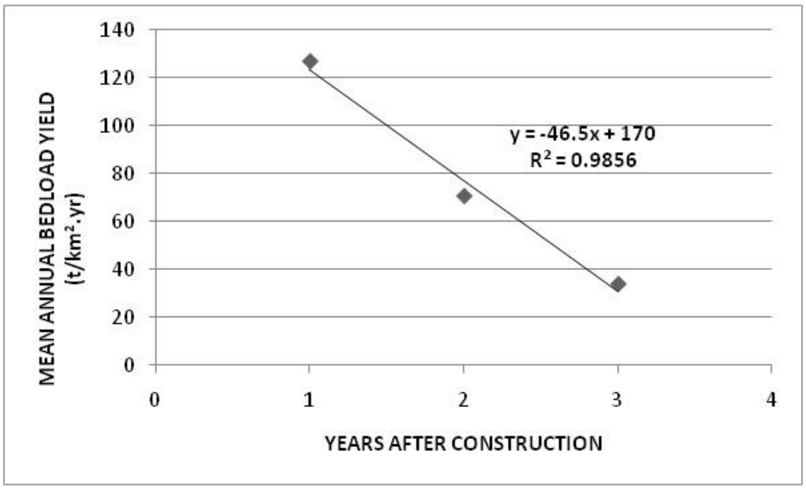

Figure 3 Linear decrease in mean annual bedload yield with time since construction for the four erosion plots on the trial landform

Previous research in the ARR has shown that sediment yields decline progressively over at least the first three years following a major surface disturbance as a result of initial washout of fine sediment and the subsequent formation of a gravel-armoured surface (Duggan 1994). Clearly, time since construction, rather than rainfall, is the dominant driver of bedload yield because by far the greatest rainfall occurred in the second year (Table 1) whilst bedload was substantially less than in the first year (Table 3). Using the average rainfall per rain day as an index of rainfall intensity, the values for the three years were 13.6, 15.2 and 11.8 mm/d for the 2009–10, 2010–11 and 2011–12 water years, respectively. Clearly 2010–11 was not only the wettest year but also had the most intense rainfall.

The highest bedload yields were always generated from Plot 2 (Table 3). This is most likely caused by this plot having the lowest ground cover with shallow rip lines dominate the lower part of Plot 2, resulting in direct connection of diffuse overland flow transporting sediment with the down slope plot drain.

In the third year since construction a clear signature of the effect of vegetation is starting to be seen in the annual bedload yields. The two plots (1 & 4) originally planted with tube stock now have the greatest shrub densities and, both now show lower bedload yields than the plots (2 & 3) initially planted by (failed) direct seeding followed a year later by infill planting with tube stock. Indeed, plots 1 and 4 also recorded the two lower bedload yields in 2011–12 (Table 3). Plot 4 has the lowest yield because it has also been invaded by weeds which densely cover about half of the plot.

Table 4 shows the particle size of the bedload for each year of measurement in terms of the Wentworth size fractions of gravel (> 2 mm), sand (63 µm to 2 mm) and silt and clay (< 63 µm) (Folk 1980). Each sample was dry sieved to determine the gravel and sand fractions and the hydrometer method was used to determine the silt and clay fraction (Folk 1980, Gee & Bauder 1986). For most plots, percentage gravel has progressively increased and percentage silt and clay has progressively decreased over time. Changes in the sand fraction compensate for these changes. Surface armouring of coarse gravel has occurred by the winnowing or washing out of fine sediment from the ground surface.

Table 4 Annual particle size of bedload for each plot on the trial landform for each year of measurement

Water year	*Erosion plot*	*% gravel in bedload*	*% sand in bedload*	*% silt and clay in bedload*
2009–10	EP1	34	60	6
2010–11	EP1	33	64	3
2011–12	EP1	44	53	3
2009–10	EP2	34	55	11
2010–11	EP2	40	55	5
2011–12	EP2	42	55	3
2009–10	EP3	37	59	4
2010–11	EP3	46	53	1
2011–12	EP3	47	52	1
2009–10	EP4	35	61	4
2010–11	EP4	50	49	1
2011–12	EP4	50	49	1

Future work

The most pressing outstanding work in the shorter term is to complete the calculation of runoff data from all plots since the runoff must be determined before suspended sediment and solute loads can be derived. Discharge from plots 2, 3 and 4 still remains to be determined. It is anticipated that sediment and solute load information from a subset of the plots will be available for presentation next year. Advances have been made in obtaining a detailed turbidity record for each plot. In addition, electrical conductivity data have been inspected to determine how to combine the record from the inlet probe with the probe mounted on the flume.

It is planned to continue monitoring the trial landform until at least 2013–14 to track the trajectory of runoff, sediment and solute yields from an evolving and revegetating landform. Objectives include quantifying the effect of developing vegetation on erosion rates, such that a much higher level of confidence can be placed in the predictions from the landform evolution models (see Lowry et al 2011, Saynor et al 2012b) that are being used to predict long-term erosion performance and assist with the design of the final mine landform. The runoff, sediment and solute loads that are being measured will also inform the design of sediment traps and wetland water quality polishing systems that will need to be incorporated into the rehabilitated mine footprint to manage the export of erosion products.

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LiDAR capture for the Alligator Rivers Region

RE Bartolo & J Lowry

Introduction

The Australian Government Department of Climate Change and Energy Efficiency (DCCEE) released a report titled 'Kakadu: Vulnerability to climate change impacts' in June 2011. The South Alligator River Catchment was used as a case study to examine the potential impacts of climate change and in particular sea level rise on Kakadu National Park's (KNP) World Heritage and Ramsar listed wetlands. The report concluded that the freshwater wetlands of the region are vulnerable to climate change impacts, and that sea level rise will increase saltwater intrusion events, thereby threatening the current status of the wetlands. It was identified that the most fundamental information gap is a Digital Elevation Model (DEM) of suitable resolution for this low relief area since: 'without this tool it is virtually impossible for Park managers to undertake a detailed assessment of the most important Park assets (values), determine which may be at the highest risk of climate change and which may need to be protected or conserved with any degree of certainty'.

In order to address this fundamental information gap, SEWPaC and DCCEE commissioned a LiDAR (Light Detection and Ranging) data capture for the Alligator Rivers Region (ARR) floodplains, through Geoscience Australia's Optical, Geospatial, Radar, and Elevation Supplies and Services Panel (OGRE). ***eriss*** provided advice for the development of the technical specifications for the project and liaised with the DCCEE coordinator on behalf of SEWPaC and the National Environmental Research Program, Northern Australia Hub (NERP NAH). In addition to the lowland ARR floodplain areas (including the Magela Creek floodplain) that were covered by the primary remit of the acquisition, ***eriss*** requested that LiDAR data also be acquired for the Nabarlek (decommissioned U mine) catchment, Myra Camp (U exploration camp in Arnhem Land located in the Tin Camp Creek catchment) and a targeted area of escarpment in the East Alligator area to assist with the uranium mining assessment and research activities of the Division.

Data acquisition

The total area (Figure 1) for data capture was approximately 4,000 km^2. Data acquisition commenced on the 22 October and was completed on 16 November 2011, with 100% of the specified area captured. The data capture was carried out as late as possible in the 2011 dry season and immediately prior to the start of the 2011–12 wet session rainfall, so that the minimum amount of inundation was present at the time of capture.

The ground control survey (carried out by the contractor's land survey crew) commenced on 7 November and was completed on 18 November 2011. In order to get ground returns in densely vegetated areas (mangroves and *Melaleuca*), the sampling intensity that was specified was 4pts/m^2, compared with an average of 2pts/m^2 for the less densely vegetated areas on the floodplain and for areas with greater topographic relief (Nabarlek site within the Cooper Creek catchment, Myra Camp and the East Alligator River landslips area). The East Alligator landslips occurred in March 2007 during the wettest wet season on record for Jabiru.

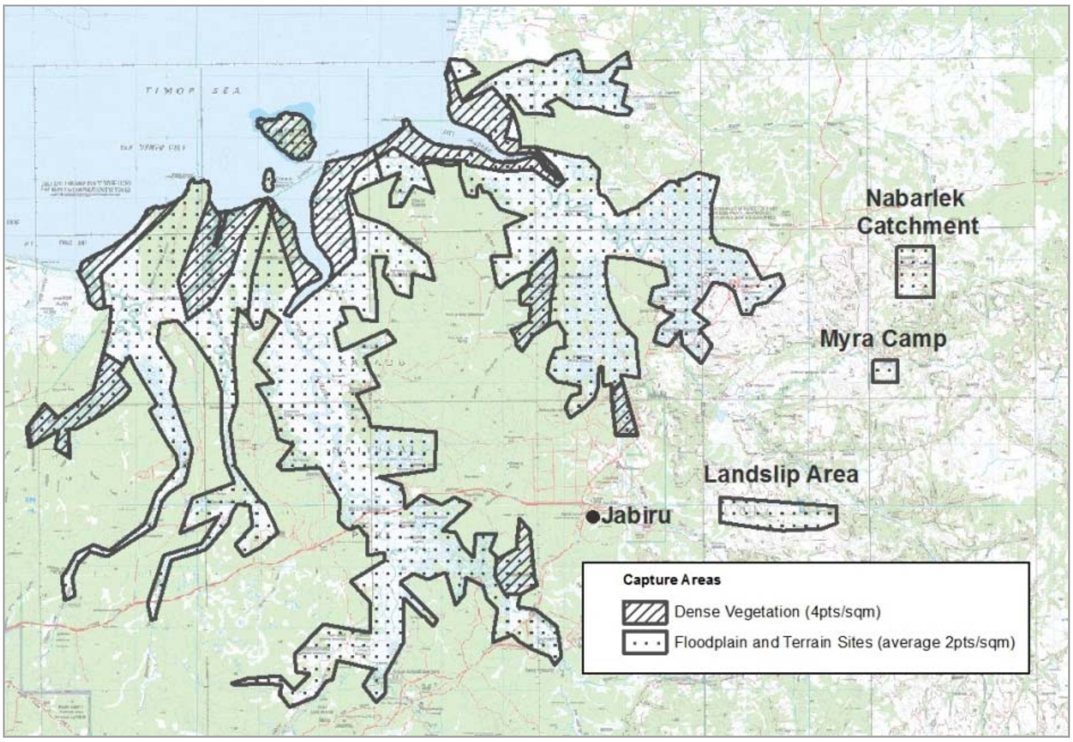

Figure 1 Alligator River Region data capture area showing those areas where data was required to be captured at a higher sampling rate (ie 4 pts/m^2)

Data products and preliminary data

The products to be supplied from this project are shown in Table 1. The data has been quality assured/quality checked by Geoscience Australia. Independent classification validation is currently being undertaken by the CRC for Spatial information and the final work package is being disseminated. However some preliminary data have been supplied for initial checking. Figures 2 and 3 show high resolution topographic images of land surfaces derived from the Digital Elevation Model (DEM) for the East Alligator River landslip area. Figures 4 and 5 show a mosaic of the DEM tiles for the East Alligator River catchment capture area, and a subset of the mosaic classified at 0.5m height intervals, respectively.

Table 1 Topographic LiDAR product summary

Data Product	Format	Resolution
Unclassified LiDAR points	LAS	Av 2 pts/m^2
Classified LiDAR points	LAS	Av 2 pts/m^2
LiDAR intensity (tiles and mosaic)	ECW (mos.) & Geotiff (tiles)	1 m
Surface model (DSM)	Grid	1 m
Elevation model (DEM)	Grid	1 m
Canopy elevation model (CEM)	Grid	2 m
Canopy foliage model (CFM)	Grid	2 m
Contours	Vector	0.5 m

A comparison of the LiDAR derived DEM and the Shuttle Radar Topography Mission (SRTM) 1' DEM for a subset of the Magela floodplain is shown in Figure 6. The LiDAR derived DEM has been classified at 0.5 m height intervals for the floodplain area as the resolution is 1 m with a vertical accuracy of 10–30 cm, while the SRTM derived 1' DEM has been classified at 1 m height intervals for the floodplain area as the resolution is 30 m with a vertical accuracy of approximately 3.8 m. The LiDAR derived DEM appears to be affected by the presence of dense floodplain grasses with elevations ranging between 3 and 4.5m and vegetation patterning evident as can be seen in Figure 6.

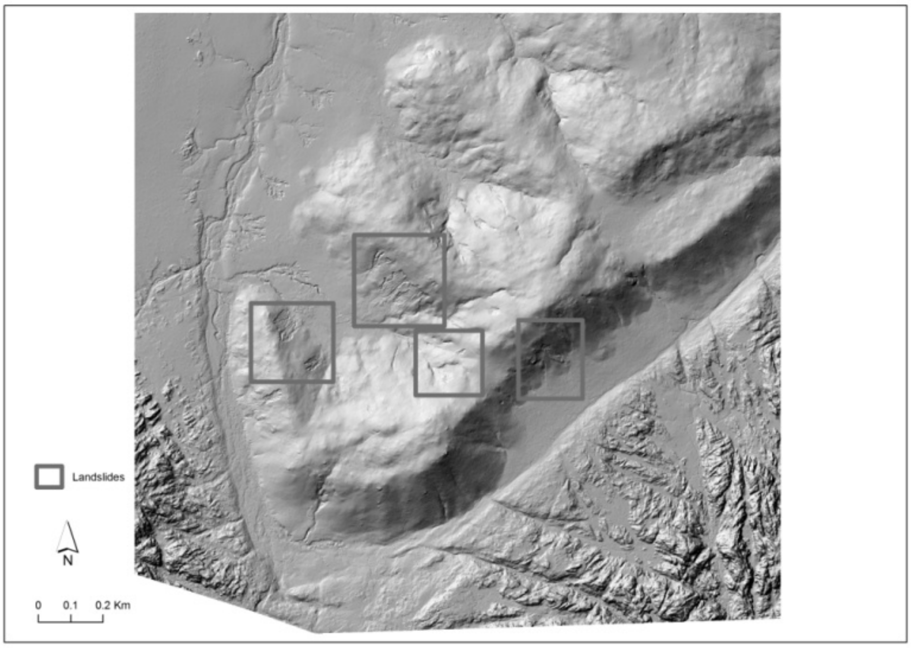

Figure 2 Hillshade image for the East Alligator landslip area with the landslips highlighted

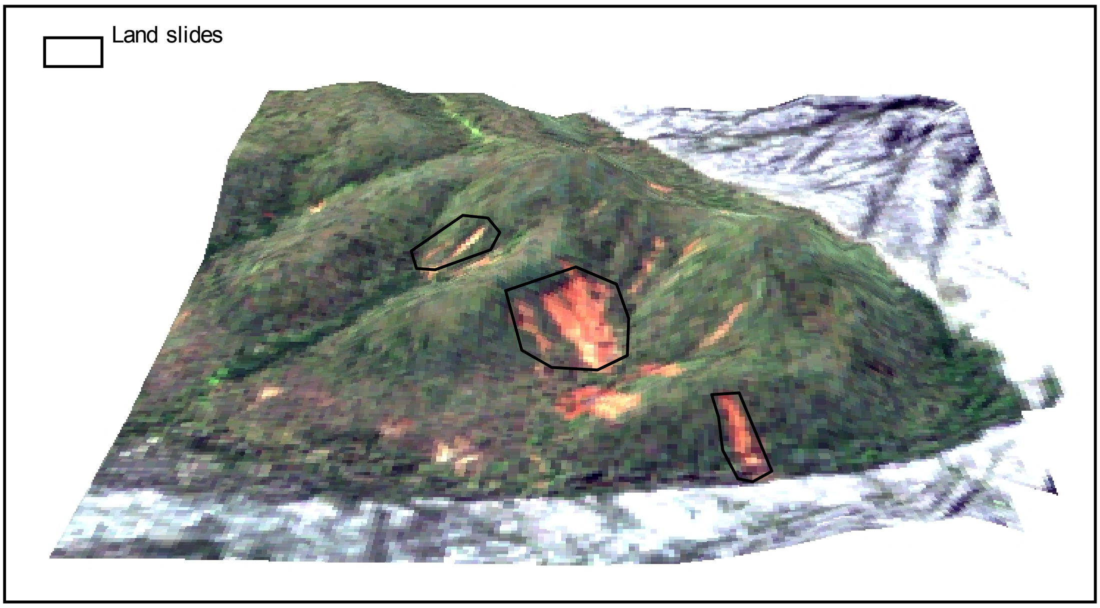

Figure 3 ALOS AVNIR-2 satellite image (2008) draped over the DEM for the East Alligator landslip area. The areas containing the landslips (lighter shade) have been highlighted.

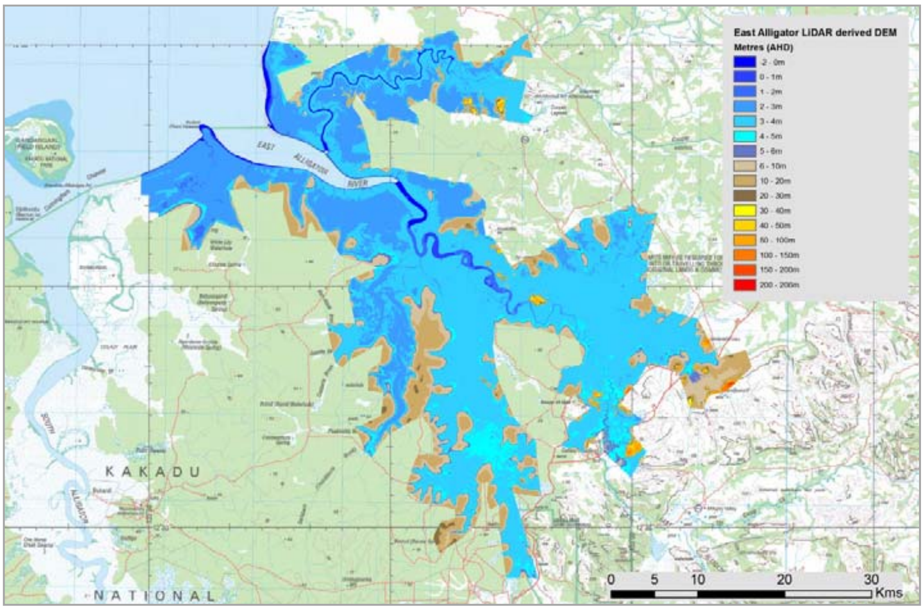

Figure 4 Mosaic of LiDAR derived DEM data for the East Alligator River catchment capture area

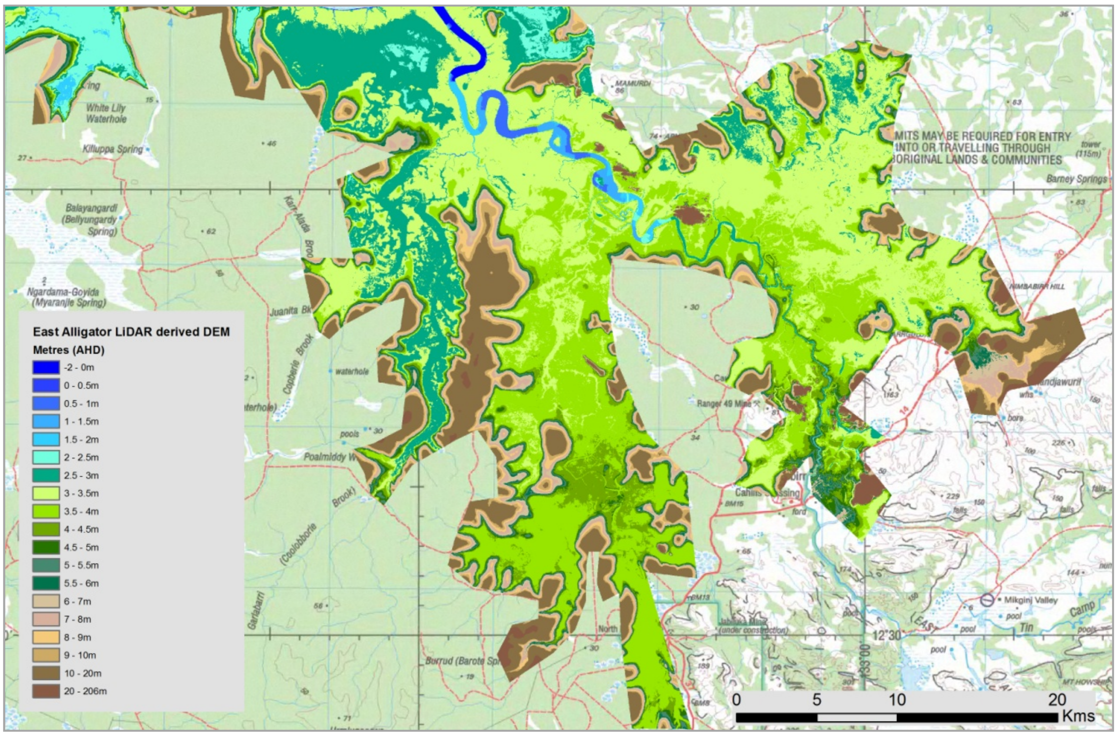

Figure 5 A subset of the East Alligator River catchment capture area DEM classified at 0.5 m height intervals

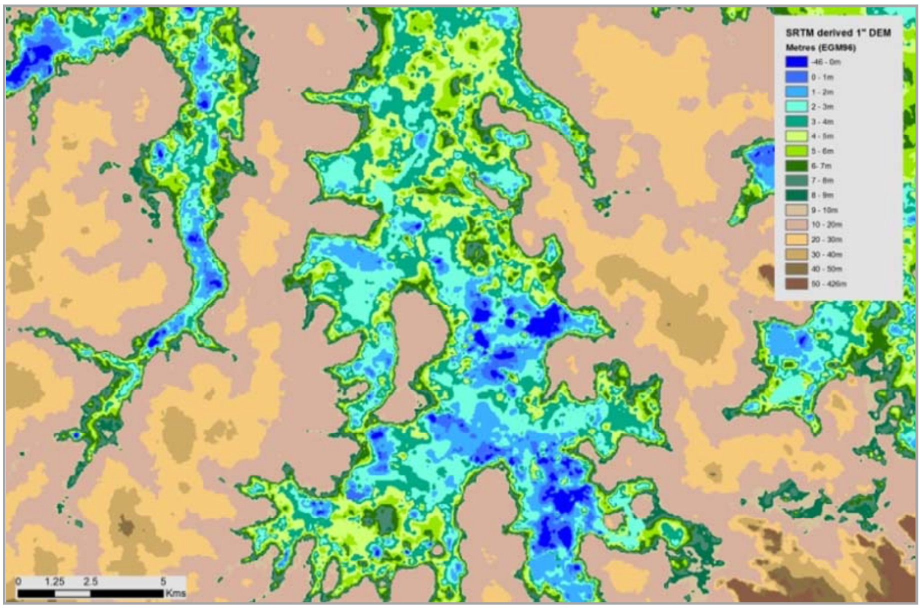

Figure 6 Comparison of LiDAR derived DEM (left) and Shuttle Radar Topography Mission (SRTM) derived DEM (right) for a subset of the Magela floodplain. Note that the LiDAR derived DEM has been classified at 0.5 m intervals for the floodplain and the SRTM derived DEM has been classified at 1 m intervals for the floodplain area. This is due to the differing resolutions of the datasets.

Future work

The floodplain LiDAR data will be used by NERP NAH researchers in developing hydrodynamic models and assessing the impacts of sea level rise in the Alligator Rivers Region. In terms of SSD specific project work the following projects will utilise the LiDAR data:

- Cooper Creek catchment – the LiDAR derived DEM provides sufficient vertical resolution for the application of geomorphic Landscape Evolution Models to predict the long term erosion potential for this catchment which includes the decommissioned and largely rehabilitated Nabarlek U mine.
- Myra Exploration Camp and Tin Camp Creek catchment – the data will be used in a long term erosion modelling and monitoring project being undertaken in collaboration between ***eriss*** and the University of Newcastle.
- East Alligator landslips – the LiDAR data will be used to assist with a geotechnical research project (collaboration with the University of Western Australia) on factors controlling occurrence of landslips in high rainfall tropical areas.

The classified LiDAR data are being validated by the CRC for Spatial Information (CRC-SI) to examine misclassifications such as where water and low vegetation have been classified as bare ground. The following datasets have been supplied to the CRC-SI for their validation process: high resolution (15 cm) control point air photos for Kakadu National Park provided by the Northern Territory Government (see Figure 7 for an example of these data); LiDAR data captured by ERA in 2010, specifically an area of overlap for the Magela floodplain; a 2004 air photo mosaic for KNP; the 2010 Magela floodplain vegetation map derived from WorldView-2 satellite imagery; and 2010, 2011 and 2012 WorldView-2 satellite imagery for the Magela floodplain.

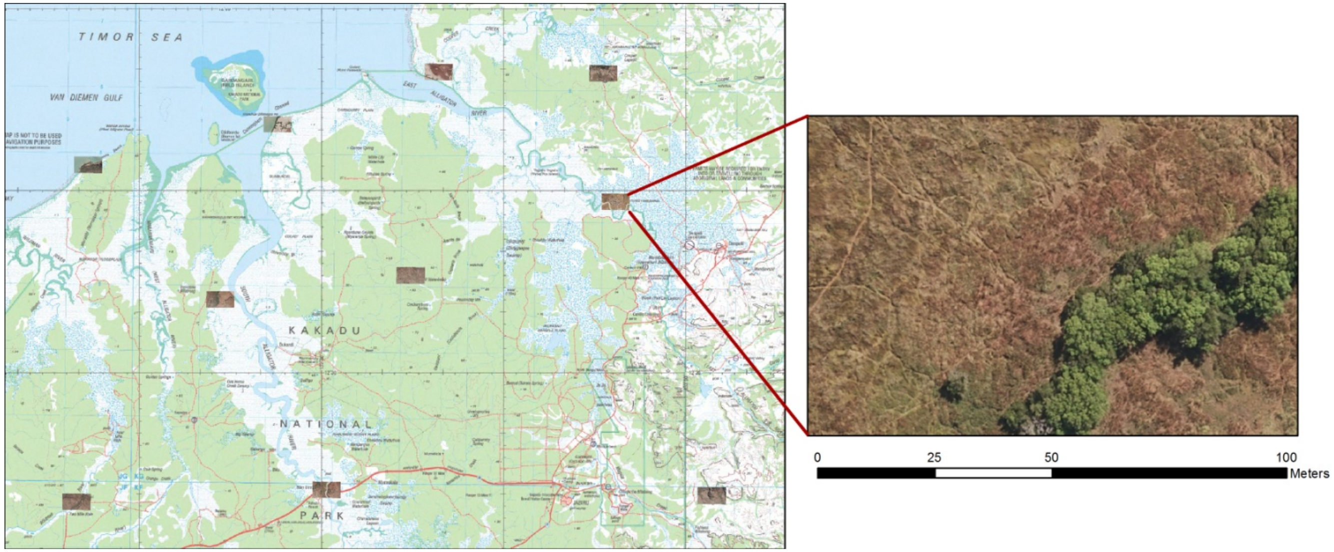

Figure 7 An example of the Northern Territory Government control point air photo data with a resolution of 15 cm. The image on the left shows the position of control point photos to be used for the LiDAR validation. The air photo on the right shows a small subset of one of these control point photo.

Assessing the geomorphic stability of the proposed rehabilitated Pit 1 landform

J Lowry, T Coulthard[1] & G Hancock[2]

Introduction

At the 27th meeting of the Alligator Rivers Region Technical Committee (ARRTC) in December 2011, ERA announced a concerted focus on rehabilitation and closure research needs for the Ranger uranium mine. It was agreed that ERA would identify the key operational and closure-related tasks, and associated knowledge requirements, and prioritise these against the current Key Knowledge Needs (KKNs) – specifically KKN 2.2.1 *Landform Design* and KKN 2.2.4 *Geomorphic Behaviour and Evolution of the Final Landform*. It was recognised that the focus on rehabilitation and closure research needs for Ranger would strongly influence, and hence be accounted for in, the 2012–13 research planning cycles for both ERA and SSD. Among the highlighted priority needs was an assessment of the geomorphic stability of the proposed landform lead by SSD, with results to be available by the third quarter of 2012. This information is required for use in the finalisation of the landform design, and the development of closure criteria.

The project commenced during the second quarter of 2012. The assessment initially focussed on the portion of the proposed rehabilitated landform that currently comprises Pit 1 (Figure 1) and that will be capped years before (according to the current schedule) the rest of the mine. The assessment was conducted by the Supervising Scientist Division (SSD) in conjunction with research partners at the University of Hull and the University of Newcastle.

Two conceptual scenarios representing a surcharged (to allow for consolidation of the capped tailings in the pit) and a non-surcharged (representing final consolidated land surface) landform were provided to SSD for assessment by ERA.

Methods

Two landform evolution models (LEM) – CAESAR-Lisflood and SIBERIA – were used to assess the geomorphic stability of the conceptual landforms supplied by ERA.

CAESAR-Lisflood combines the Lisflood-FP 2d hydrodynamic flow model (Bates et al 2010) with the CAESAR-Lisflood geomorphic model (Coulthard et al 2000, 2002, 2005, Van de Wiel et al 2007) to simulate erosion and deposition in river catchments and reaches over time scales from hours to thousands of years. The SIBERIA model was used to model the different landform scenarios over a simulated period of 1000 years to provide a comparison of this well-established model with the CAESAR-Lisflood simulations.

The fundamental difference between the two models is that the CAESAR-Lisflood model can utilise hourly recorded rainfall data from the study area, enabling the modelling of the effects of specific rainfall events, while the SIBERIA model can only use annual rainfall data for its inputs. Event modelling is critical, especially for the early stages of landform evolution, since it is well recognised that the majority of erosion typically occurs during a limited number of high

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intensity events (Saynor & Evans 2001). The ability to model specific rainfall events meant that the CAESAR-Lisflood model was the model of choice for this project.

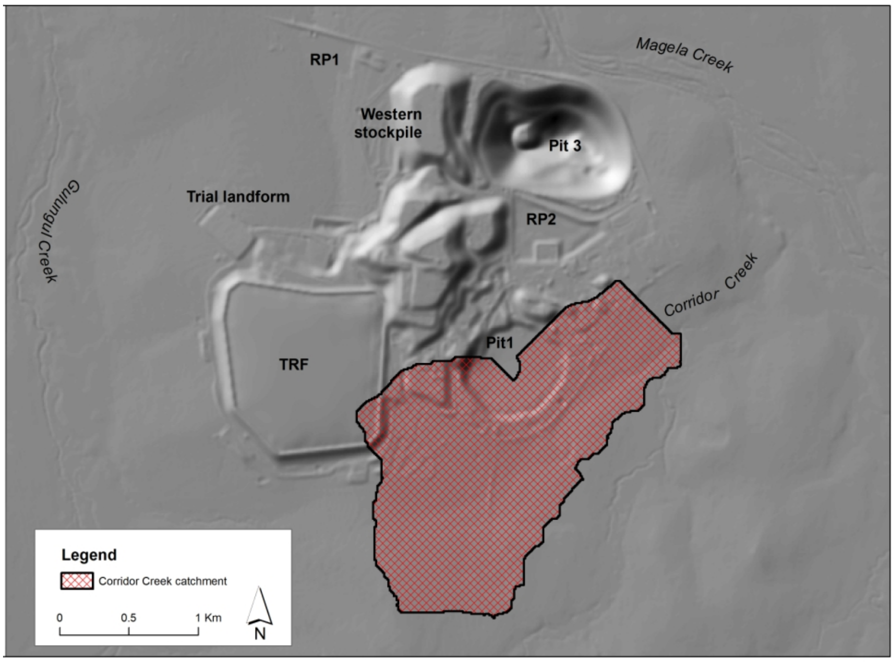

Figure 1 Corridor Creek/Pit 1 study area superimposed on DEM of current Ranger surface

An important aspect of this project was the ability to model the differential consolidation of the landform surface over Pit 1. Professor Coulthard, the developer of the CAESAR-Lisflood model was engaged to incorporate the ability to model the differential consolidation of the Pit 1 surface into the CAESAR-Lisflood model. Data produced by consultants ATC Williams (2009) for ERA was provided to SSD and Professor Coulthard and used to enable differential consolidation to be incorporated into the CAESAR-Lisflood model.

Three key data inputs are required by the CAESAR-Lisflood model: a high resolution digital elevation model; rainfall data; and particle size data.

Elevation data

Digital elevation models (DEMs) of the surcharged and non-surcharged versions of the conceptual landform were supplied by ERA. A number of significant issues were found with the ERA-supplied DEMs, which limited their utility as key inputs in the landform modelling process. These related primarily to the smoothing of the 'non-rehabilitated' areas of the study catchment included in the landform model adjacent to the Ranger Operational Area which produced data artefacts. In order to address these issues DEMs of the landform supplied by ERA were revised and enhanced by SSD through the integration of high-resolution LiDAR elevation data of the area surrounding the mine, and the elevation data of the conceptual landform. For modelling purposes, the DEMs were compiled to a horizontal spatial resolution of 10 metres. Ten metres was determined to be the optimal resolution at which the CAESAR-LisFlood model could function within the spatial extent of the study catchments, and over the temporal periods modelled.

Rainfall data

Twenty-two years of complete (ie no gaps in the annual dataset) hourly rainfall intensity data, is available for the period between 1971 and 2006 for Jabiru airport 2 kilometres north-west of the Ranger mine site. The one-hour rainfall totals (mm) for the 22 years were used to form the rainfall inputs for the different scenarios modelled using CAESAR-Lisflood.

Rainfall data was compiled for 2 scenarios:

- A 45-year simulation, in which the 22 years of rainfall over the period 1971–2006 was looped to run twice for a simulated period of 44 years, with the addition of the rainfall data from 2007 which included an extreme rainfall event at the end of the second loop.
- A 1000 year simulation was run in which the 22-year Jabiru rainfall was looped out to a period 1000 years. The 2007 extreme rainfall event was not used in this simulation.

Particle size data

Using the process described in Saynor and Houghton (2011), the particle size data for CAESAR-Lisflood was obtained from size fractionated bulk samples of surface material collected at eight points on the waste rock surface of the Ranger trial landform in the 2009 dry season. Grain size analysis was completed on these samples and the results averaged into nine grain size classes (Figure 2) which were used for input into CAESAR-Lisflood. The sub 0.000063m (ie 63 µm) fraction is treated as suspended sediment within CAESAR-Lisflood.

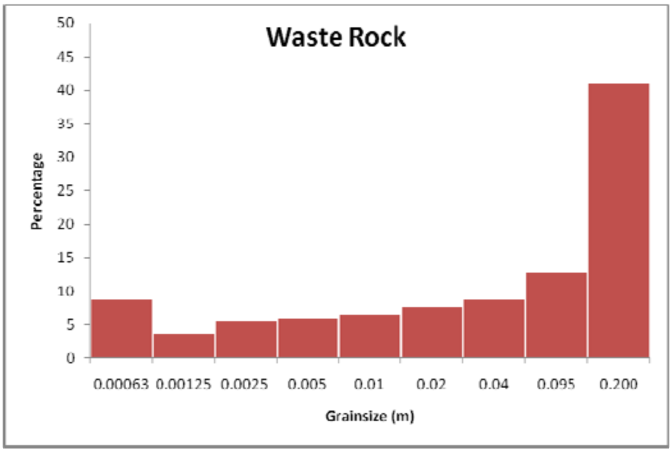

Figure 2 Grainsize distribution of waste rock used in simulations

For the modelling purposes of this study, the entire landform was assumed to be composed of the waste rock material.

Six different landform scenarios – static surcharged landform with/without vegetation, consolidated surcharged landform with/without vegetation, non-surcharged landform with/without vegetation – were modelled for periods of 45 and 1000 years each using CAESAR-Lisflood. Four scenarios – static surcharged landform with/without vegetation and non-surcharged landform with/without vegetation – were modelled for a period of 1000 years in SIBERIA.

Results

Results of the simulations on the Pit 1 component of the conceptual landform within the Corridor Creek catchment show that:

- For equivalent surface conditions, the CAESAR-Lisflood model predicts little difference between the surcharged and non-surcharged landforms in terms of total sediment yield. An example of sediment yield over a 1000-year period is shown in Table 1.

- Predicted denudation rates are higher than published rates (0.01 to 0.04 mm y^{-1}) of natural denudation for the region (Cull et al 1992,; Erskine & Saynor 2000). However, it is believed that the higher rates predicted by the simulation results may be due to the bare waste rock surface 'acclimatising'. That is, finer particles within the surface are preferentially eroded and irregularities in the model DEM surface are removed, resulting in initially higher sediment yields. This is a strength of the model and potentially provides a representation of what the surface behaviour may be when the landform is first constructed.
- The SIBERIA simulations (Table 2) for a surcharged/non surcharged surface produced lower denudation rates than the CAESAR simulations. The lower denudation rates produced by SIBERIA were expected, as the CAESAR simulations are based on individual events, whilst the SIBERIA outputs are based on averaged input data. The incorporation of a vegetation component into 1000-year SIBERIA simulations of the surcharged and non-surcharged landforms produced denudation rates which fell within the published rates for the natural landscape.
- The inclusion of hypothetical vegetation (a uniform grass cover in this instance) greatly reduced total sediment yield and denudation rates in all of the scenarios in which it was applied.
- None of the scenarios predicted gullies with a depth of greater than 5 metres over a time period of 1000 years. In the case of a vegetated consolidated landform, a 4-metre deep gully is predicted to form on the side of the landform over a period of 1000 years (Figure 3).
- Shallow gullies are predicted to form on the side of the surcharged Pit 1 landform within a period of 45 years, regardless of whether vegetation is present.
- The absence of vegetation appears to lead to channel widening and movement. The presence of vegetation reduces the number of gullies that are likely to form – although those that are formed incise markedly over time.

Table 1 Sediment yields and denudation rates after 1000 years from CAESAR-Lisflood

	Surcharged				No surcharge	
	No vegetation		Vegetation		No vegetation	Vegetation
	Consolidation	No consolidation (static)	Consolidation	No consolidation (static)		
Total Load (m^3)	262175	290704	610819	610819	258498	615593
Denudation rate (mm y^{-1})	0.086	0.095	0.2	0.213	0.08	0.202

Table 2 Predicted denudation rates on the Pit 1 landform after 1000 years from SIBERIA

	Surcharged		No surcharge	
	No vegetation	Vegetation	No vegetation	Vegetation
min (deposition)	3.2 m	1.6 m	2.5m	2.4m
max (erosion)	4.7 m	5.0 m	4.7m	6.6m
denudation	0.126 mm yr^{-1}	0.015 mm yr^{-1}	0.113 mm yr^{-1}	0.019 mm yr^{-1}

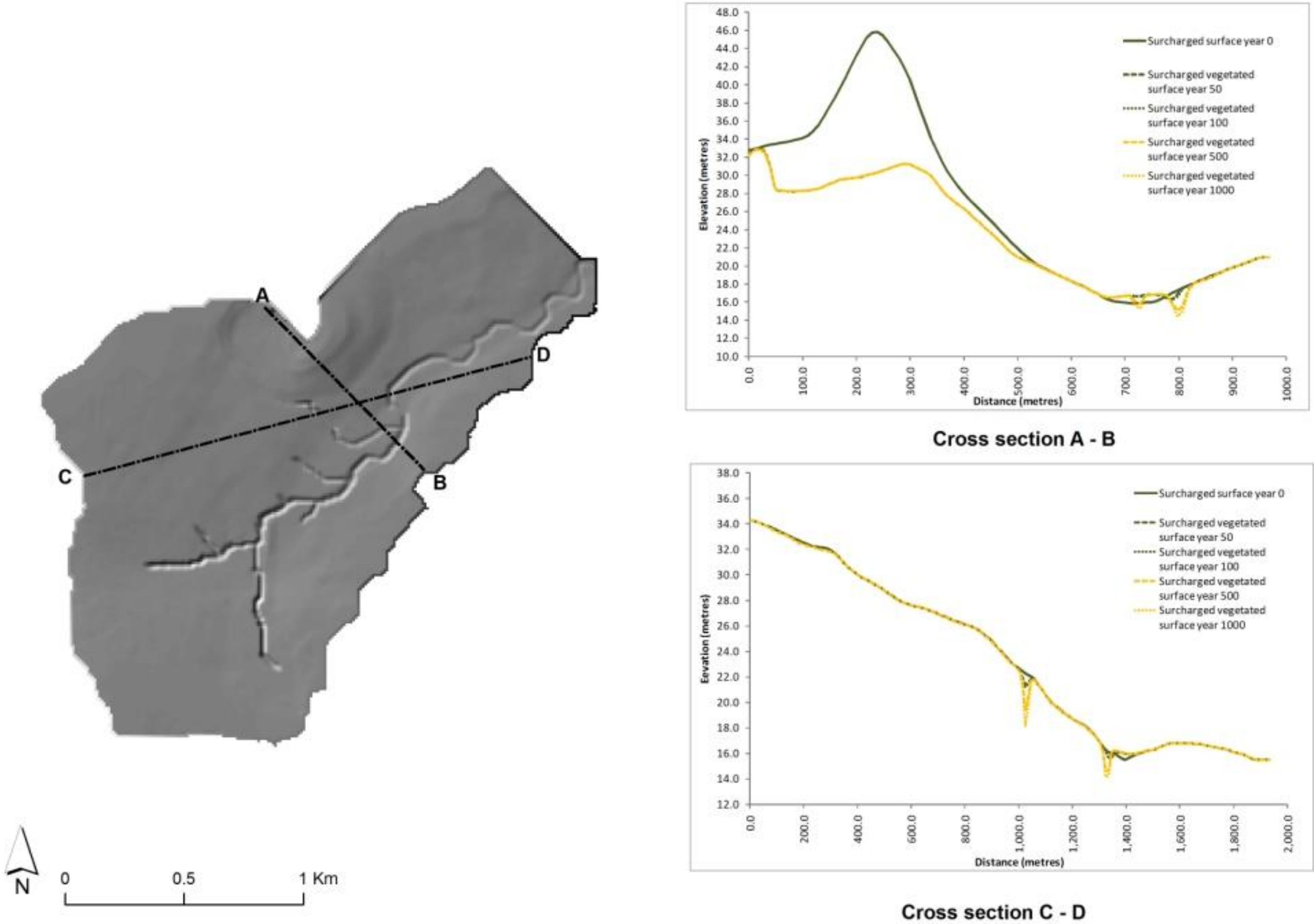

Figure 3 Vegetated, consolidated surcharged upper Corridor Creek catchment after 1000 years

Steps for completion

This study has focussed purely on the potential impacts of the Pit 1 component of the landform on one of the catchments that will be impacted by the rehabilitated landform. The potential impact of the rest of the landform on the remaining catchments that comprise the rehabilitated landform will need to be modelled to enable the performance of the proposed landform as a whole to be assessed.

Ultimately, the performance of the landform will need to be assessed for a period of up to 10 000 years. This will likely require the CAESAR-Lisflood model to be closely integrated with the SIBERIA model to enable model outputs to be produced for that time period.

The study to date has identified a number of issues which should be incorporated into any future landform designs and modelling activities. These include identifying and incorporating different surface treatments appropriate for specific functional elements (eg rock-armoured drainage collection channels, erosion control structures, sediment traps) into the final landform. Other areas which will require further research include better understanding and defining the vegetation parameters value to better reflect the effects of developing vegetation communities, and the impact of fires on these communities. Analysis of palaeoflood records in the Alligator Rivers Region may provide a guide to the magnitude of extreme climate events that could be incorporated into future model scenarios. Further work should be undertaken on the CAESAR-Lisflood model to enable accurate predictions of suspended sediment concentrations to be made under a range of scenarios.

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Pre-mining radiological conditions at Ranger mine

A Bollhöfer, A Beraldo, K Pfitzner, A Esparon & G Carr[1]

Introduction

The Ranger orebodies were discovered in 1969 by Noranda (Australia) during an airborne gamma survey (AGS) flown in an adjacent area (Ryan 1972, Eupene et al 1975). Several follow up AGS were flown in the region before mining started. The objective of the project summarised here was to acquire and assess pre-mining AGS data that included the Ranger Project area and a suitable undisturbed radiological anomaly. Once such an anomaly was identified and groundtruthed, it could be used to extrapolate to pre-mining radiological conditions in the greater Ranger region.

Data from a pre-mining AGS flown in 1976 were acquired by ***eriss*** from the Northern Territory Department of Mines and Energy in 2006. In addition, data from an AGS flown in 1997, and covering Anomaly 2 to the south of the Ranger lease, were supplied by Rio Tinto. Extensive measurements of external gamma dose rates, radon flux densities and soil activity concentrations were subsequently conducted by ***eriss*** staff between 2007–2009 at Anomaly 2, just south of the Ranger lease. The objective was to groundtruth the 1976 AGS data to retrospectively determine the pre-mining radiological source term for Ranger mine.

In 2011–12 the focus of this project was on write-up of the results and a draft internal ***eriss*** report has been finalised (Bollhöfer et al 2013). Results have also been presented at the 2012 South Pacific Environmental Radioactivity Association conference (Bollhöfer et al 2012a).

Methods

Detailed descriptions of the methods for the measurements of external gamma dose rates, radon flux densities and soil radionuclide activity concentrations, as well as the specifications of the surveys can be found in Bollhöfer et al (2013) and have previously been reported in ***eriss*** research summaries (Bollhöfer et al 2011, 2012b). In short, external gamma dose rates measured on ground at Anomaly 2 at a resolution of 10–15 m were scaled up to a resolution comparable to that of the 1997 AGS. This was done using ArcGIS and it was found that a buffer radius of 90 m resulted in the best correlation between averaged ground gamma data and airborne uranium data. Consequently, the established correlation is only applicable for a cell size larger than 4 ha. The 1976 AGS data were then clipped to the extent of the 1997 survey, with the mine footprint excluded, and a correlation was established between the 1976 and 1997 AGS data sets.

Measurements of the soil ^{226}Ra activity concentrations and external gamma dose rates allowed s correlation to be established between gamma dose rate measured 1 m above the ground and soil ^{226}Ra activity concentration. The correlation agrees well with the theoretical predictions by Saito and Jacob (1995). In addition correlations were established between radon flux density, gamma dose rate and soil ^{226}Ra activity concentration, that are similar to values reported by Lawrence et al (2009) for the Ranger region.

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Results

Figure 1 shows the uranium data in the region with some of the mine site features overlaid. Table 1 shows an estimate of the average pre-mining external gamma dose rates, soil ^{226}Ra activity concentrations and radon flux densities for those features calculated from equations given in Bollhöfer et al (2013).

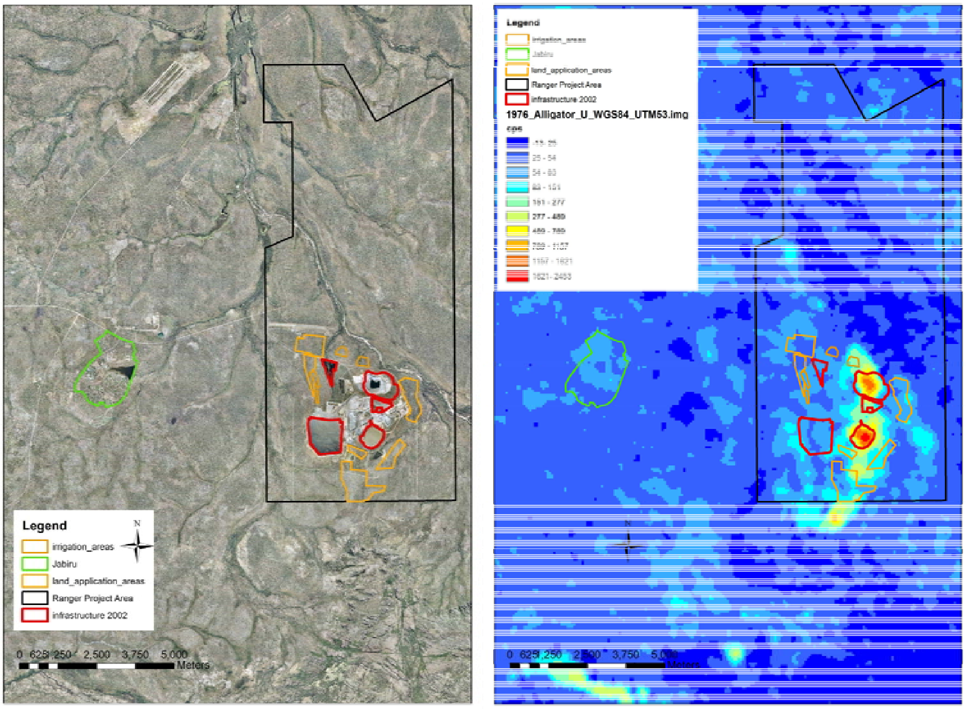

Figure 1 Mine features and Jabiru town overlaid on an aerial photo of the greater Ranger region (left) and on the counts in the eU channel of the 1976 AGS of the same extent (right)

Table 1 Pre-mining average (and 95% confidence intervals) external absorbed gamma dose rates (including cosmic component), radon flux densities and soil ^{226}Ra activity concentrations estimated using the GIS for features in the greater Ranger region shown in Figure 1

Infrastructure	Area [ha]	E_{ave} [µGy·hr^{-1}]	^{226}Ra [Bq·g^{-1}]	Rn_{ave} [Bq·m^{-2}·s^{-1}]
Pit 1	40	**0.87** (0.59–1.2)	**1.9** (1.2–2.5)	**2.7** (1.4–4.4)
Pit 3	77	**0.44** (0.26–0.62)	**0.9** (0.4–1.4)	**1.3** (0.5–2.3)
Anomaly 2	38	**0.33** (0.18–0.49)	**0.6** (0.2–1.1)	**0.9** (0.3–1.7)
Djalkmara LAA	9	**0.20** (0.07–0.32)	**0.3** (0–0.7)	**0.45** (0–1.0)
Corridor Ck LAAs	139	**0.13** (0.07–0.24)	**0.2** (0–0.5)	**0.23** (0–0.72)
Tailings Dam	110	**0.11** (0.07–0.21)	**0.1** (0–0.5)	**0.14** (0–0.58)
Magela LAAs	56	**0.11** (0.07–0.21)	**0.1** (0–0.5)	**0.15** (0–0.60)
RP1	17	**0.10** (0.07–0.20)	**0.1** (0–0.4)	**0.12** (0–0.55)
RP1 LAAs	38	**0.10** (0.07–0.20)	**0.1** (0–0.4)	**0.11** (0–0.54)
Djalkmara ext LAA	9	**0.10** (0.07–0.20)	**0.1** (0–0.4)	**0.10** (0–0.53)
Jabiru East LAA	50	**0.10** (0.07–0.20)	**0.1** (0–0.4)	**0.11** (0–0.54)
Ranger Project Area	7881	**0.11** (0.07–0.21)	**0.1** (0–0.5)	**0.14** (0–0.58)
Jabiru	306	**0.10** (0.07–0.20)	**0.1** (0–0.4)	**0.12** (0–0.55)

Discussion

As expected count rates in the uranium channel of the 1976 AGS are highest over orebodies 1 and 3 and Anomaly 2 (Figure 1) and thus these features exhibit the highest calculated external gamma dose rates, soil ^{226}Ra activity concentrations and radon flux densities. The average pre mining external gamma dose rates calculated from the radiological GIS for orebodies 1 and 3 and for the Anomaly 2 area are 0.87, 0.44 and 0.33 $\mu Gy \cdot hr^{-1}$, respectively. The average has been determined for the surface area of the two pits, rather than the surface area of the outcropping orebodies.

Kvasnicka and Auty (1994) have determined gamma dose rates for the two orebodies and report a gamma dose rate of 0.96 $\mu Gy \cdot hr^{-1}$ on orebody 1 and 0.58 $\mu Gy \cdot hr^{-1}$ on orebody 3. Whereas the dose rate on top of orebody 3 was measured directly, dose rates for orebody 1 were estimated from radionuclide activity concentrations in a drill core taken above the orebody. Differences in average values between our study and Kvasnicka and Auty (1994) most likely arise from the different areal extent of the anomalies investigated. The surface area of orebody 1 investigated by Kvasnicka and Auty (1994) was 44 ha, which is similar to the area used for averaging in our study, and external gamma dose rates agree well. In contrast, estimates for orebody 3 by Kvasnicka and Auty (1994) were only over an area of 66 ha compared to 77 ha in our study. Relatively lower values at the outer edges of the outcropping orebody have been included in our estimated average, leading to the difference observed between the two studies.

The average external gamma dose rates determined above Anomaly 2 using spatial analysis is 0.33 $\mu Gy \cdot hr^{-1}$, whereas the average of all field measurements is 0.40 $\mu Gy \cdot hr^{-1}$. However, the average of our field measurements is skewed towards higher values, as more measurements were taken in the vicinity of the anomalies as the field survey grid spacing was larger outside the anomalies. The typical external gamma dose rate, ie the geometric mean of the field measurements taken across Anomaly 2, amounts to 0.30 $\mu Gy \cdot hr^{-1}$. The typical environmental background gamma dose rate determined for the whole extent of the 1976 AGS data set and using the derived correlations is approximately 0.1 $\mu Gy \cdot hr^{-1}$. This compares well with typical background gamma dose rates published for the ARR, ranging from 0.08 to 0.15 $\mu Gy \cdot hr^{-1}$.

Kvasnicka and Auty (1994) also measured the radon flux density on top of orebody 3 at three locations and the average amounted to 4.4±1.0 $Bq \cdot m^{-2} \cdot s^{-1}$. Furthermore, they use these measurements and soil ^{226}Ra activity concentrations at those three locations to calculate a conversion factor of 1.85±0.23 $(mBq \cdot m^{-2} \cdot s^{-1})/(Bq \cdot kg^{-1})$ to determine the ^{222}Rn flux density across the two orebodies from the estimated average ^{226}Ra activity concentrations in soil. They estimate radon flux densities of 4.1 $Bq \cdot m^{-2} \cdot s^{-1}$ and 2.5 $Bq \cdot m^{-2} \cdot s^{-1}$ for orebodies 1 and 3, respectively, higher than pre-mining radon flux densities determined using spatial analysis.

Some of the difference in radon exhalation flux densities for orebody 3 can be explained by the smaller areal extent of the orebody 3 study area compared to our study (see above). In addition, the conversion factor used by Kvasnicka and Auty (1994) to calculate radon flux densities is larger than a conversion factor of (1.4±0.3) $(mBq \cdot m^{-2} \cdot s^{-1})/(Bq \cdot kg^{-1})$ estimated in our study. This factor was determined from direct radon flux density measurements, measured terrestrial gamma radiation dose rates and soil ^{226}Ra activity concentrations at Anomaly 2. Using our conversion factor, radon flux densities for orebodies 1 and 3 from Kvasnicka and Auty (1994) amount to 3.2 $mBq \cdot m^{-2} \cdot s^{-1}$ and 1.9 $mBq \cdot m^{-2} \cdot s^{-1}$, respectively, similar to the averages obtained in our study.

The average radon flux density determined from measurement in the field above Anomaly 2 was 2.2 $Bq \cdot m^{-2} \cdot s^{-1}$ compared with 0.9 $Bq \cdot m^{-2} \cdot s^{-1}$ determined using spatial analysis. This difference is due to the over-representation of radon exhalation measurements on top of the radiation anomalies compared to other areas. Repeated measurements on high radon flux areas will skew the average value of the field measurements towards a higher value. The typical radon flux density, ie the geometric mean of all individual field measurements taken above the Anomaly, is 0.7 $Bq \cdot m^{-2} \cdot s^{-1}$, similar to the average determined with the spatial model. Radon exhalation rates for typical environmental background sites amounts to 0.06 $Bq \cdot m^{-2} \cdot s^{-1}$. This is in agreement with values reported previously for the ARR (Lawrence et al 2009, Todd et al 1998).

Outstanding work

The pre-mining spatial analysis determines the magnitude of the terrestrial gamma and ingestion pathways, assuming radioactive equilibrium in soil between all members of the uranium decay series and using concentration ratios from the BRUCE database (see KKN 2.5.4 BRUCE). The magnitude of the inhalation pathways pre-mining needs to be determined, both for on-site and off-site scenarios. It is unlikely that the dust inhalation pathway contributed significantly to offsite inhalation doses, given that offsite inhalation doses from the inhalation of dust are very small even during the operational phase (see KKN 1.3.1 Monitoring). However, it may have been an important contributor when roaming the site, including the outcropping orebodies, for food gathering activities or camping. It is thus required to determine dust resuspension factors in the ARR and compare these with existing literature data and models (eg IAEA 2010) including earlier data for the ARR (Akber 1992, Akber et al 2011). This will be done in 2012–13 via measurement of radionuclide activity concentrations in soils and on filters collected for the atmospheric monitoring program.

Using computer codes such as RESRAD-Offsite (Yu et al 2009), which considers exposure pathways including the inhalation of particulates and radon, will allow inhalation doses to be determined for an individual who spends time in the vicinity of the Ranger site from inputs of data such as soil radionuclide activity concentrations or radionuclide fluxes to the atmosphere. ResRad training is being organised for the early part of 2013.

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Radon exhalation from a rehabilitated landform

A Bollhöfer & C Doering

Introduction

Details of the design of the trial landform have been provided in previous reports, in particular those that describe the work being done to quantify the rate of erosion (Saynor et al 2010). The trial landform also provides a unique setting to investigate seasonal and long-term changes in radon (^{222}Rn) exhalation, soil activity concentration and terrestrial gamma dose rate for the four different treatments, and dependency on cover type, weathering and compaction effects and developing vegetation.

Radon is part of the natural uranium decay chain and is produced by the decay of radium (^{226}Ra) in soil particles. It is a noble gas and some of the radon emanates from the particles, migrates through the soil pore space and eventually exhales from the soil surface. High soil radium activity concentrations and high porosity will lead to larger radon flux densities and vice versa (Porstendörfer 1994). In addition, particle size and soil moisture have an influence on radon exhalation, with smaller particles favouring radon emanation and high soil moisture generally inhibiting the migration of radon through the soil profile (Tanner 1980). Consequently, the radon exhalation rate may potentially change as the final landform evolves after rehabilitation of the site, in particular due to the highly weatherable nature of the waste rock at Ranger (Riley & Waggitt 1992). At the rehabilitated Nabarlek mine site for example, differences in radon flux densities immediately (Kvasnicka 1996) and five years after rehabilitation have been reported, although these differences could also be due to differences in experimental design between the two studies (Bollhöfer et al 2006).

Radon decays with a half life of 3.82 days to short-lived isotopes of the metals polonium (^{218}Po, ^{214}Po), lead (^{214}Pb) and bismuth (^{214}Bi). It is these radon decay products in the air, rather than the radon gas, that can deposit in the lungs and deliver a radiation dose upon inhalation (Kendall & Smith 2002). The knowledge of the radon exhalation fluxes from a rehabilitated landform thus forms the first step to predict potential inhalation doses received from the rehabilitated landform.

The objective of this project is to determine radon (^{222}Rn) exhalation flux densities for various combinations of cover types (two) and re-vegetation strategies (two) on the trial landform and to investigate seasonal and long-term changes in radon exhalation. Specifically, the ^{222}Rn exhalation from the four erosion plots constructed by SSD are measured over several years to investigate whether there are any temporal changes of radon exhalation, taking into account rainfall, weathering of the rock, erosion and compaction effects, and the effect of developing vegetation on the landform. The trial landform provides a unique opportunity to track radon exhalation over many years. The project will enable ***eriss*** and ERA to more confidently predict a long-term radon exhalation flux from a rehabilitated landform and contribute to the development of closure criteria for the site.

Methods

Radon flux density measurements have been made on the four erosion plots (30 m x 30 m) every four months since early 2009. In addition, radon flux density was measured on the original land surface before construction of the trial landform to determine the magnitude of radon exhalation from the substrate underlying the constructed landform (that is, the pre-construction background) (Bollhöfer et al 2009). A total of 765 measurements of radon flux densities have been made to date.

Brass canisters (15–20 per erosion plot) filled with activated charcoal are placed randomly over the surface of the erosion plots. Details on the charcoal canister methodology are provided in Bollhöfer et al (2006). Radon exhaling from the surface is trapped on the charcoal. After deployment for three days, the brass canisters are collected, sealed and sent to the SSD Darwin laboratories for analysis. Radon trapped on the charcoal decays and the activity of radon decay products collected in each brass cylinder is measured using a sodium iodide (NaI) gamma detector. The measurements coupled with the length of the deployment and measurement periods, enable radon flux densities from the surface of the soils to be determined (Spehr & Johnston 1983).

The gamma dose rate has also been measured across the entire trial landform using environmental dose rate meters. A 5 m grid was used for each of the four erosion plots with a 10 m grid being used for the remainder of the landform. A total of 921 measurements were made. The terrestrial component of the gamma dose rate was determined by subtracting the contribution from cosmic radiation to the gamma signal (Marten 1992). The terrestrial gamma dose rate can be used as a surrogate for the substrate ^{226}Ra activity concentration as the gamma signal from natural uranium series elements measured at 1m height originates predominantly from ^{214}Pb and ^{214}Bi, which are short lived decay products of ^{226}Ra (Saito & Jacob 1995).

In July 2012, soil samples (n=68) down to 10 cm depth were collected from underneath the locations where the radon exhalation measurements were made. These samples are being measured for their ^{226}Ra activity concentration via gamma spectrometry to obtain a quantitative relationship between radon flux densities and ^{226}Ra activity concentration for the two treatments.

Terrestrial gamma dose rates

Figure 1 shows the trial landform with the areas of the four different treatments clearly visible, the locations of the erosion plots (EP 1–4) and a contour plot of the terrestrial gamma dose rates measured across the surface in June 2012. This plot is an inverse distance weighted interpolation of the measured values and has been generated using the ArcGIS (version 9.3.1) Spatial Analyst tool. EP 1 and EP 2 were constructed on the waste rock cover treatments, while EP 3 and EP 4 are located on the waste rock–laterite mix.

Apart from a small area on treatment 2 on the opposite side of the landform to EP 2 where terrestrial gamma dose rates of up to 1 $\mu Gy \cdot hr^{-1}$ were measured, the dose rate is between 0.11–0.46 $\mu Gy \cdot hr^{-1}$, which is about 2–4 times higher than typical terrestrial background dose rates in the region. Average (both arithmetic and geometric) terrestrial gamma dose rate is lowest over EP 4 (0.20 $\mu Gy \cdot hr^{-1}$) and highest over EP 2 (0.24 $\mu Gy \cdot hr^{-1}$) (Table 1).

Dose coefficients given in UNSCEAR (2008) allow to estimate terrestrial gamma dose rates from measured ^{238}U, ^{232}Th and ^{40}K soil activity concentrations, assuming radioactive equilibrium of all members within the ^{238}U and ^{232}Th decay series and homogeneous distribution throughout the soil profile. Using the average ^{228}Ra (~42 $Bq \cdot kg^{-1}$) and ^{40}K (730–950 $Bq \cdot kg^{-1}$) activity concentrations measured previously in substrate from the erosion plots (Bollhöfer & Pfitzner 2011), ^{226}Ra activity concentrations can be estimated from measured terrestrial gamma dose rates using equation 1 below:

$$^{226}Ra = (D_{terr} - a_1 \cdot {}^{40}K - a_3 \cdot {}^{228}Ra)/a_2 \qquad (1)$$

With:

D_{terr}: measured absorbed terrestrial gamma dose rate [$\mu Gy \cdot hr^{-1}$]

a_1: $0.0417 \cdot 10^{-3}$ ($\mu Gy \cdot hr^{-1}$)/($Bq \cdot kg^{-1}$)

a_2: $0.462 \cdot 10^{-3}$ ($\mu Gy \cdot hr^{-1}$)/($Bq \cdot kg^{-1}$)

a_3: $0.604 \cdot 10^{-3}$ ($\mu Gy \cdot hr^{-1}$)/($Bq \cdot kg^{-1}$)

^{40}K, ^{226}Ra, ^{228}Ra: radionuclide activity concentrations [$Bq \cdot kg^{-1}$].

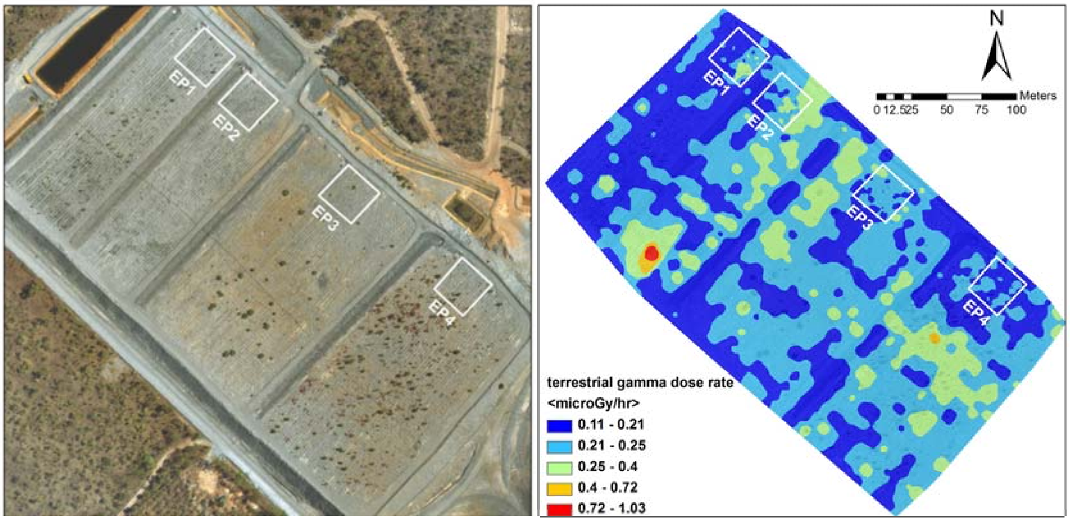

Figure 1 Aerial photo of the trial landform and contour plot of the terrestrial gamma dose rates measured across the trial landform in June 2012

Table 1 shows the arithmetic (and *geometric*) average terrestrial origin gamma dose rates measured across the different treatments and erosion plots, and the ^{226}Ra activity concentrations estimated using equation 1. Direct measurement of the ^{226}Ra activity concentrations is being performed via gamma spectrometry in samples collected across the erosion plots and analyses will be finished by the end of 2012.

Table 1 Average (arithmetic and *geometric*) terrestrial γ-dose rate measured on the trial landform in July 2012 and estimated substrate ^{226}Ra activity concentrations

Plot	γ-dose rate [$\mu Gy \cdot hr^{-1}$]	^{226}Ra [$Bq \cdot kg^{-1}$]
	arithmetic (*geometric*) average ± error (95% conf)	
Treatment 1	0.202*(0.198)* ± 0.005	295*(288)* ± 14
EP 1	0.212*(0.207)* ± 0.011	316*(307)* ± 25
Treatment 2	0.240*(0.232)* ± 0.014	378*(361)* ± 31
EP 2	0.241*(0.238)* ± 0.011	380*(372)* ± 25
Treatment 3	0.221*(0.219)* ± 0.004	358*(352)* ± 15
EP 3	0.222*(0.221)* ± 0.006	360*(357)* ± 19
Treatment 4	0.221*(0.219)* ± 0.004	358*(353)* ± 16
EP 4	0.204*(0.203)* ± 0.006	322*(319)* ± 18

Radon flux densities

Radon flux densities on the four erosion plots have now been measured over three years starting in May 2009 and covering both wet and dry seasons. Arithmetic (and *geometric*) averages for each plot and measurement campaign, and the associated standard errors, are shown in Table 2. Figure 2 shows a plot of the geometric average, or typical value, of the radon flux densities measured over time. The daily rainfall measured on the trial landform is also shown. Annual rainfall averages for the trial landform were 1518 mm, 2255 mm and 1496 mm in the 2009–10, 2010–11 and 2011–12 wet season, respectively (Supervising Scientist 2012).

The spread in measured radon exhalation fluxes is large due to the small surface area of the charcoal canisters used to collect the exhaled radon relative to the surface area of the erosion plots. Such variability is not unusual, and it has been shown that the geometric average is a better representation of the average, due to environmental radon flux densities typically being log normally distributed. Radon flux density from the pre-construction unmineralised substrate was measured late in 2008 (Bollhöfer et al 2009) and was around 70 mBq·m^{-2}·s^{-1}, similar to the average (±1SD) late dry season radon flux density of 64 ±25 mBq·m^{-2}·s^{-1}, which was previously determined for the region (Todd et al 1998).

Table 2 Average (arithmetic and *geometric*) radon flux densities measured on erosion plots 1–4, May 2009 to July 2012

Plot	**^{222}Rn flux density [mBq·m^{-2}·s^{-1}]** arithmetic (*geometric*) average ± error (95% confidence)									
	May 09	Sep 09	Feb 10	May 10	Sep 10	Jan 11	May 11	Oct 11	Mar 12	Jul 12
EP 1	22(*14*) ±11	15(*9*) ±8	7(*4*) ±3	43(*21*) ±25	60(*26*) ±47	100(*27*) ±76	60(*18*) ±63	68(*24*) ±47	47(*18*) ±27	112(*82*) ±40
EP 2	42(*27*) ±15	17(*9*) ±10	8(*5*) ±4	45(*28*) ±20	69(*40*) ±34	126(*50*) ±86	67(*38*) ±37	82(*43*) ±43	107(*41*) ±41	165(*121*) ±121
EP 3	18(*13*) ±7	14(*9*) ±8	5(*1*) ±2	51(*21*) ±35	102(*78*) ±35	64(*32*) ±50	65(*37*) ±33	63(*49*) ±19	69(*35*) ±38	199(*162*) ±59
EP 4	18(*14*) ±7	40(*19*) ±32	6(*3*) ±4	83(*42*) ±51	111(*68*) ±60	89(*25*) ±57	71(*55*) ±22	112(*79*) ±41	40(*20*) ±20	170(*106*) ±97

Measured radon flux densities were low in 2009, shortly after construction of the landform. The main substrate used for construction of the landform is waste rock, consisting of a chlorite-rich schist that weathers rapidly to smaller gravel and clay size fractions (Riley & Waggitt 1992). The higher surface area to volume ratio of weathered rock favours radon emanation, leading to the higher radon exhalation observed after the first wet season. In addition, radon exhalation from a soil surface is proportional to the gradient of the radon concentration in the interstitial soil space. Typical diffusion lengths for radon are 1.5 m up to 3 m, with diffusion coefficients between 10^{-8} to 10^{-5} m^2·s^{-1} (Porstendörfer 1994). This gradient needed time, after initial construction of the landform, to develop through the substrate profile until steady state conditions were achieved. Consequently, the initially measured radon flux densities were relatively low and in the order of 20–40 mBq·m^{-2}·s^{-1} only, as the radon originated predominantly from the top most layers, consisting of relatively coarse schistose material.

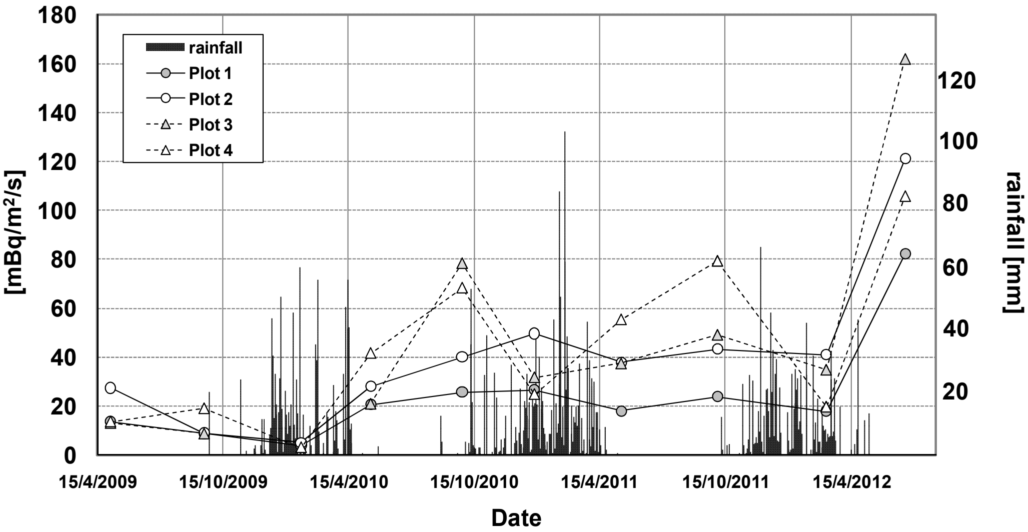

Figure 2 Time series geometric averages of the radon flux densities measured on erosion plots 1–4 on the landform with daily rainfall overlaid for comparison

Following the initial measurements in 2009, radon flux densities on EP 3 and EP 4 show the expected seasonal trend, with higher values in the dry and lower values in the wet seasons. In contrast, EP 1 and EP 2 show relatively little seasonal variation and dry season values on these two plots are similar to those measured during the wet season. However, in July 2012 average values were higher compared with the previous sets of data for EP 1 and EP 2. Radon exhalation from EP 2 is generally higher than radon exhalation from EP 1, which can be explained by the higher ^{226}Ra activity concentration of surface material between the two plots (Table 1).

Differences between waste rock only and the waste rock–laterite mix, both in average radon flux densities as well as its seasonal variation, can be explained by the different properties of the substrate materials. The average soil radium activity concentration is similar across the four erosion plots (Table 1). However, EP 3 and EP 4 have approximately 30 v/v % of fine grained laterite mixed with the waste rock. Soils collected from the waste rock–laterite mix surface in 2010 have a smaller average grain size, with an average percentage of silts and clays (< 63 µm) of 11% compared with 7% for waste rock only (Saynor & Houghton 2011). In addition, the percentage gravel (> 2 mm) is higher for waste rock only (67%) compared with the waste rock–laterite mix (61%). This favours radon exhalation from the waste rock–laterite mix and results in higher radon flux densities during the dry season compared with the waste rock only treatment. Further weathering of the waste rock on EP 1 and EP 2 may lead to smaller observed relative differences between the two treatments.

During the wet season, moisture is retained more effectively in the waste rock–laterite mix and radon flux densities decrease. Pooling of water is routinely observed on EP 3 and EP 4 during the wet season meaning that in some areas the soil is saturated with water. This leads to a decrease in radon exhalation (Tanner 1980). In contrast, no pooling has been observed on EP 1 and EP 2. Water infiltrates quickly into the waste rock cover after a rain event and consequently, the difference between wet season and dry season radon flux densities is small. This may change with time, with smaller grain weathering products slowly infilling cracks and voids on the waste rock only cover, perhaps leading to a seasonal variability similar to that currently observed on EP 3 and EP 4.

Conclusions and future work

Although average soil radioactivity is not markedly different across the four erosion plots, there is a difference in average radon flux densities for the two different surface treatments. In the dry season, typical average radon flux densities from the surface of the waste rock – laterite treatment are higher than radon flux densities from waste rock only, and they decrease markedly in the wet. In contrast, there was no obvious seasonal trend observed for radon exhalation fluxes from waste rock only. Ongoing seasonal radon flux density measurements are required to determine whether there will be further long-term changes. This is particularly important to ascertain for the waste rock only treatment, as this is the preferred option for rehabilitation at Ranger.

The measurement of ^{226}Ra activity concentrations in soil samples that were collected in July 2012 from underneath the locations where the radon exhalation measurements were made will be completed at the end of 2012. The investigation of the ratio of the radon flux densities to soil ^{226}Ra activity concentrations at these sites will aid in modelling radon exhalation from various cover types and predicting radon fluxes from the Ranger site after rehabilitation. A preliminary estimate of this ratio using Tables 1 and 2 shows that typical dry season ratios in 2012 were between 0.3 and 0.5 $(mBq \cdot m^{-2} \cdot s^{-1})/(Bq \cdot kg^{-1})$ similar to values given in Lawrence et al (2009) for rehabilitated waste rock dumps at the Ranger uranium mine.

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Developing water quality closure criteria for Ranger billabongs using macroinvertebrate community data

C Humphrey, D Jones, L Chandler & A Frostick

Background

Georgetown Billabong (GTB) is a natural waterbody located immediately downstream of Ranger minesite (Map 2) that discharges into Magela Creek. It has historically received low levels of minesite solutes (mainly magnesium sulfate ($MgSO_4$) and also uranium (U)) since the inception of mining. Slightly elevated U in the sediments of GTB sampled in 1978, (3x average value of U measured in sediments of adjacent reference waterbodies) also indicates that erosion from the surface expression of ore body number 1 located in the Georgetown Creek catchment has contributed U to the billabong historically and prior to mining (Humphrey et al 2009). Figure 1 shows the historical seasonal electrical conductivity (EC) record in GTB. Electrical conductivity is a useful surrogate for the concentration of $MgSO_4$, the main mine-derived solute.[1] As reported in Humphrey and Jones (2012), GTB is being used as a case study to develop surface water quality closure criteria, in this case for the solutes EC, $MgSO_4$ and U, for natural waterbodies within the mining lease.[2] (While EC is a reliable surrogate of Mg, it is also included because it can be readily measured in the field, including continuously, and hence provides rapid turn-around of emerging Mg contamination issues prior to laboratory confirmation.)

The approach for deriving water quality criteria from biological response data in local waterbodies is consistent with the framework outlined in the Australian and New Zealand Water Quality Guidelines (ANZECC & ARMCANZ 2000). Specifically, if the post-closure ecological condition in Georgetown Billabong is to be consistent with similar undisturbed (reference) billabong environments of Kakadu National Park (KNP), then the range of measured water quality data from the billabong over time that supports such an ecological condition – as measured by macroinvertebrate communities in this instance – may be used to derive the criteria.

Macroinvertebrate sampling in 1995, 1996 and 2006 among littoral macrophytes (representing water column habitat) supported the conclusion that biological conditions (viz relative abundances of different macroinvertebrate families) in GTB were consistent with those of reference waterbodies sampled elsewhere in the region. These results indicated that the corresponding water quality in GTB for the three sampling years was compatible with the maintenance of the aquatic ecosystem values of KNP. Derived water quality closure criteria for EC (for $MgSO_4$), Mg and U were subsequently reported (values not shown here), based on water quality and macroinvertebrate data acquired in 1995, 1996 and 2006.

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[1] R^2 values from Mg-EC regressions: for wet season = 88%; for dry season, for turbidity <200 NTU, = 79%.

[2] The derivation of (closure) criteria for turbidity in surface waters and uranium in sediments is being addressed by separate ***eriss*** KKN research projects reported elsewhere in this ARRTC compendium.

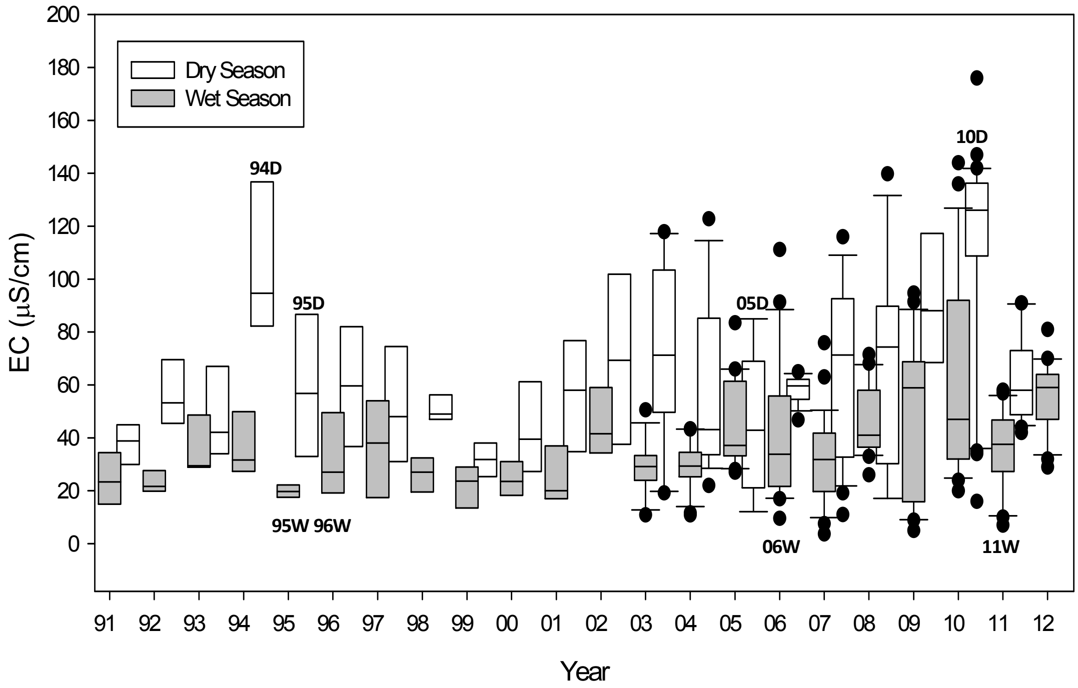

Figure 1 Summary box plots of generally weekly electrical conductivity (EC) values measured in GTB between 1991 and 2012. Box plots show median, range, 25th and 75th percentiles and outliers (points). Periods relevant to macroinvertebrate sampling are indicated by year and season (wet (W) or dry (D)). Data from Energy Resources of Australia. Wet season – January to May. Dry season – June to December.

Since 2006, the main criterion that has been applied to trigger the need for further assessment of biological condition of these waterbodies is if water quality in GTB deteriorates below the values observed in past sampling years (1995, 1996 and 2006). The values developed in 2006 have been used as minimum quality values, so that any deviation below these values has triggered further assessment of biological condition. In particular, where a deterioration in water quality is observed, one of two responses is possible: (i) where the biological condition remains similar to reference, there is potential to adaptively adjust (ie relax) the previously-derived criteria; or (ii) where the biological condition deviates from reference, implement remedial actions and continue to monitor to ensure that the previously-derived criteria adequately protect biota.

In the late 2010 dry season, water quality, based on EC, in GTB deteriorated to the worst that it has been in the past decade (Figure 1). Accordingly, the 13 previously-studied waterbodies (described in 'Sampling' section below) were re-sampled for biota during the wet-dry recessional flow transition period in May 2011 (ie end of the 2010–11 wet season). In addition to macroinvertebrates, phytoplankton and zooplankton communities were included in this sampling program to assess the relative sensitivities of these other important biological assemblages to water quality. Phytoplankton and zooplankton community results are only briefly reported here, given the low level of within-waterbody replication that was possible and the paucity of similar sampling in the past to provide a time series reference.

Sampling

Macroinvertebrate communities have been sampled by SSD four times between 1995 and 2011 with sampling conducted in up to 14 waterbodies during the wet-dry season transition period between April and May. The waterbodies have been classified as either mine water exposed, or reference that are not exposed to Ranger mine waters or influenced by natural expressions of uraniferous material. The exposed sites include the originally-natural Djalkmara (removed in the 1996 dry season by development of the open pit accessing the Ranger 3 orebody), Coonjimba, Georgetown and Gulungul billabongs and the constructed minesite waterbodies Ranger Retention Pond 1 (RP1) and Retention Pond 2 (RP2). Reference sites include Baralil, Corndorl, Wirnmuyurr, Malabanjbanjdju, Anbangbang, Buba and Sandy billabongs and Jabiru Lake. (Jabiru Lake, an artificial impoundment, provides a useful reference to RP1, a similar man-made structure.) See Maps 2 and 3 at the beginning of the current ARRTC compendium for locations of these waterbodies. Gulungul Billabong, while downstream of Ranger, has a negligible mine-water signal and for the purposes of the analyses conducted hereafter, is included amongst reference waterbodies.

In each waterbody, samples were collected from five locations and at each of these, separate samples were taken from littoral macrophyte and littoral sediment (benthic) habitats (thus 10 samples per waterbody). In 1995 and 1996, benthic and macrophyte samples were combined before processing, while for 2006 and 2011 samples from each of these two habitats were collected and processed separately. In 2011 (only), duplicate plankton samples were collected from each waterbody for phytoplankton and zooplankton analysis (species-level community structure). Water and sediment samples were also collected from the same five locations sampled in 2006 and 2011, but from only one of the waterbody locations sampled in 1995 and 1996.

In 2009, ERA commissioned a number of 'baseline' studies in and around the Ranger minesite in preparation for an EIS seeking approval to build a heap leach facility in the catchment of Georgetown Billabong. A component of those studies included characterisation of aquatic macroinvertebrate communities in Ranger mine-water exposed and adjacent reference billabongs (WRM 2010). The protocol used in that study was the same as used by SSD in the 2011 billabong study described above with the following exceptions: (i) macrophyte samples only (and not separate benthic samples) were collected; and (ii) sampling in reference billabongs was unreplicated (unlike five replicates per reference billabong collected in SSD sampling). With permission of ERA, data arising from April to May 2009 sampling were included in the current analysis and reporting.

Results

Water quality

The concentrations of mine-water-derived U in GTB have been generally low and importantly, in the antecedent periods (ie wet and/or dry season months prior to sampling) for 1995, 1996, 2006, 2009 and 2011, were at least an order of magnitude below the relevant trigger value (TV) for protection of local aquatic organisms (van Dam et al 2012).

Plots of the median of (generally) weekly EC values (direct correlate of $MgSO_4$) measured in GTB since 1991 are shown in Figure 1 for the wet season (January to May inclusive) and dry season (June to December) time periods. The antecedent wet season median EC values were low in 1995, 1996 and 2006 compared with the local EC TV of 42 µS/cm (based on an equivalent Mg TV of ~3 mg/L) for the protection of aquatic ecosystems (van Dam et al 2010). However, in the sampling periods for both 2009 and 2011 as well as in the several wet season months preceding these sampling periods , median EC values in GTB (Figure 1) approached the EC trigger value (weekly median of 3.3 and 2.5 mg/L Mg for 2009 and 2011 respectively).

Antecedent dry season EC is typically much higher than wet season values in GTB due to evapo-concentration of solutes during this period. During the wet season, solutes in surface runoff from the minesite (via Corridor Creek, see Map 2) are diluted by surface water contributions from the rest of the GTB catchment, as well as backflow from Magela Creek. Prior to 1982, EC naturally reached median dry season values of 43 µS/cm but at this time Na, K, Cl and alkalinity were the main contributors to total solute concentration and not Mg (median Mg value only 0.6 mg/L, ERA LIMS database) and sulfate. Dry season EC values in 1994, 2008 and 2010 (prior to 1995, 2009 and 2011 wet-dry transition season samplings, respectively) were the highest (median dry season values: 1994, EC of 95 µS/cm and Mg 2.9 mg/L; 2008, EC of 83 µS/cm and Mg 6.0 mg/L; 2010, EC of 128 µS/cm and Mg 8.8 mg/L) for the five years in which macroinvertebrate sampling occurred.

Macroinvertebrate communities

Macroinvertebrate community structure data (taxa and their abundances) from replicate sites of the different waterbodies have been summarised and analysed using community summaries and multivariate statistical techniques. Community summaries reported here are based on taxa (usually family) number and total abundance. One of the multivariate techniques – multi-dimensional scaling ordination (MDS) – graphically depicts the community structure of samples in a reduced (typically two or three) dimensional space. The closer samples (in this case replicate sites) are together in ordination space, the more similar is their community structure. Possible mine-related effects upon community structure would be inferred if mine-water-exposed sites were separated from those of reference sites, while interspersion of replicates of the different exposure types would imply no significant difference in community structure. Statistical tests of the separation of different community groups (in this case, different mine-exposure categories) in multivariate ordination space were quantified using Analysis of Similarity (ANOSIM) – effectively an analogue of the univariate ANOVA.

Apart from Ranger RP2, the mean taxa number and mean total abundance for macroinvertebrates sampled from littoral macrophyte habitat did not vary markedly amongst the waterbodies in 1995, 1996, 2006 and 2009 (Figure 2).

For these four years, the macroinvertebrate communities from GTB were very similar to those sampled from reference waterbodies. In 2011, however, both mean taxa number and total abundance in GTB were lower than values measured for reference waterbodies (Figure 2).

MDS ordinations, depicting the relationship of macroinvertebrate samples to one another, for each of the five years are shown in Figure 3. The RP2 data were excluded from the ordination analysis results plotted here as the high degree of separation of the RP2 replicates from those from all other waterbodies would have resulted in scaling compression of the data from other waterbodies. The MDS plots for these other waterbodies showed, generally, interspersion of replicate samples from GTB among reference waterbody samples in 1995, 1996, 2006 and 2009 but separation of GTB samples from reference waterbody samples in 2011 (Figure 3). The ANOSIM test statistic was used to compare the observed differences *between* groups – in this case exposure type, GTB, versus reference waterbodies – with the differences among replicates *within* the groups. The degree of separation between groups is denoted by the R-statistic, where R-statistic > 0.75 = groups well separated, R-statistic > 0.5 = groups overlapping but clearly different, and R-statistic < 0.25 = groups barely separable (Clarke & Gorley 2006). A significance level of < 5% = significant effect/difference.

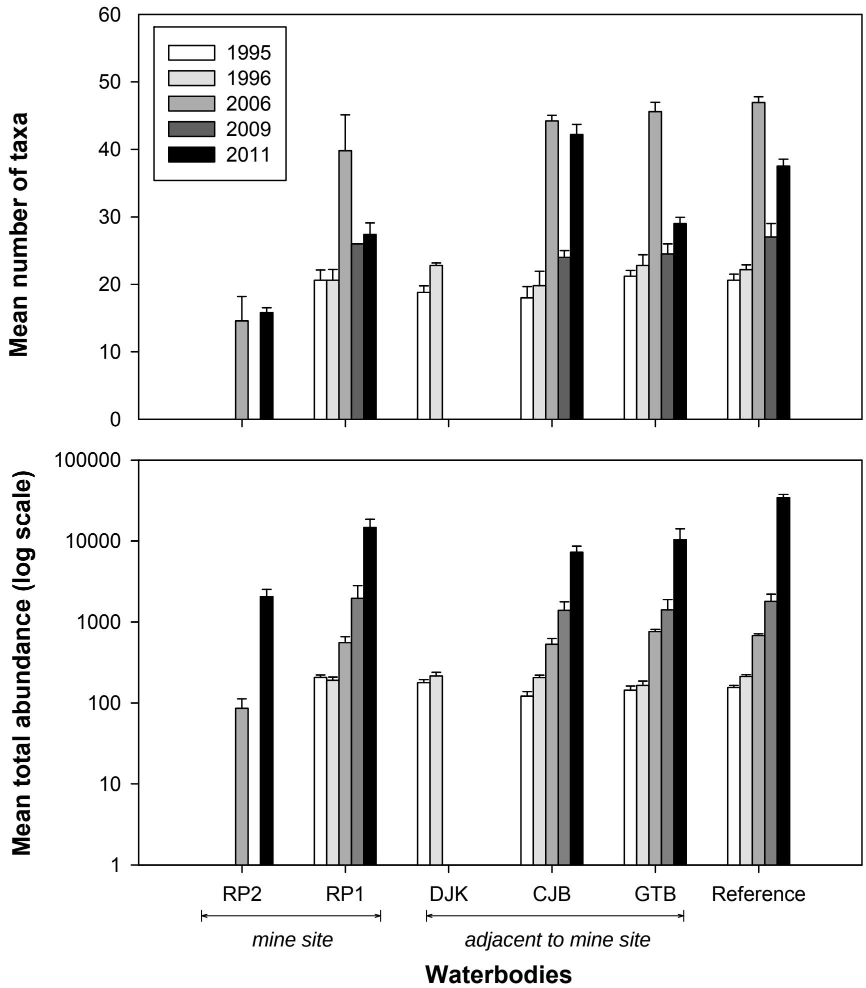

Figure 2 Histograms of mean (±SE) taxa richness (= number) and macroinvertebrate abundance amongst waterbodies on or near the Ranger uranium mine site for the five years of sampling. Site codes are Ranger Retention Pond 2 (RP2) and Retention Pond 1 (RP1), Coonjimba (CJB), Georgetown (GTB) and Djalkmara (DJK) billabongs. Reference waterbodies are Gulungul, Baralil, Corndorl, Wirnmuyurr, Malabanjbanjdju, Anbangbang, Buba and/or Sandy billabongs and Jabiru Lake.

The ANOSIM results are shown in Table 1. While GTB samples in 1995 and 2009 were significantly different from respective reference waterbody samples, the ANOSIM R statistics were generally low and near the criterion defined above as 'barely separable'. On this basis, the macroinvertebrate communities of GTB in 1995 and 2009 are not regarded as different from those in adjacent reference waterbodies in the region. For 2011, however, ANOSIM results indicate that GTB samples are clearly and significantly separated from reference waterbody samples. This result is consistent with the community summary results reported above which show reduced diversity of GTB macroinvertebrate communities compared with reference waterbody communities.

The power of phytoplankton and zooplankton community analyses was lower because only duplicate samples were collected from within each waterbody. In summary, the ordination analyses indicated that GTB phytoplankton and zooplankton communities more closely resembled those from mine-exposed waterbodies (Coonjimba Billabong and Ranger RP1) than those from reference waterbodies. These observations are consistent with those reported above for the 2011 macroinvertebrate community data.

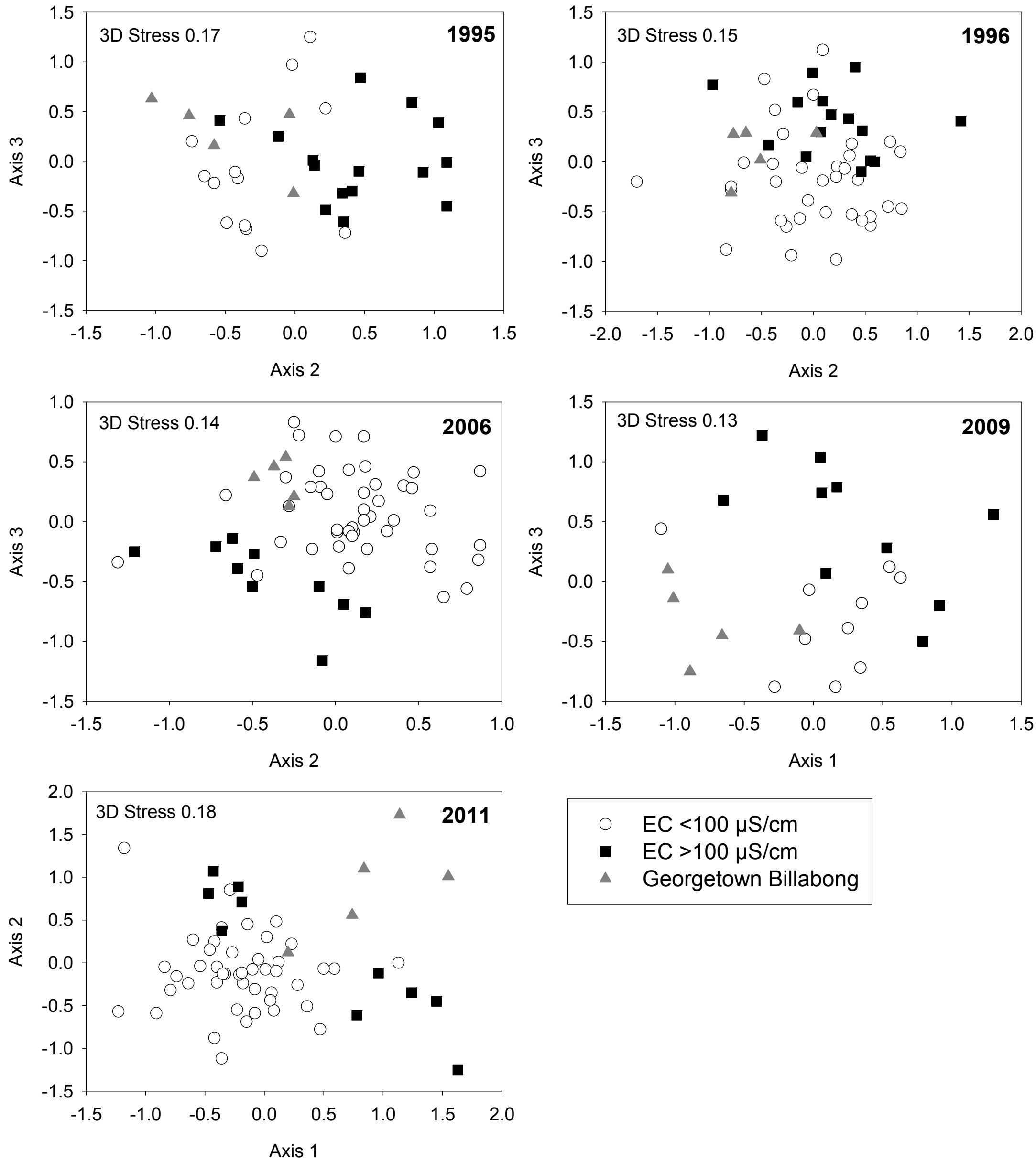

Figure 3 Ordination plots of macroinvertebrate communities from different sites and habitats in waterbodies on or near the Ranger minesite for four sampling years. Waterbodies are classified according to different degrees of exposure to mine waters indicated by electrical conductivity (EC): high for two waterbodies (RP1 and Coonjimba Billabong) and low for all other waterbodies including Georgetown Billabong. R statistic explained in the text.

Table 1 ANOSIM summary statistic results for each year comparing GTB with reference waterbodies (no mine influence)

Year	R Statistic	Significance Level %
1995	0.296	1.7
1996	-0.064	65.8
2006	0.07	30.8
2009	0.285	2.2
2011	0.594	0.1

Delineating thresholds in biological effects

To more precisely delineate thresholds in biological responses to mine-derived inputs of $MgSO_4$ to GTB, plots of taxa number and ANOSIM (GTB-reference waterbody) R values in relation to antecedent wet, dry and wet-dry combined season median Mg values were prepared. Taxa richness and not abundance was used because this community summary is least affected by variations (including biases) in sampling methodology amongst different years and studies. These plots showed that the best relationships between community responses and Mg were those that included median Mg values from antecedent wet and dry seasons combined.

GTB taxa number (as a percent of mean taxa number derived for reference waterbodies) in relation to sampling year and antecedent wet and dry season Mg is shown in Figure 4. Taxa number has declined slightly between 1995 and 2009, but a marked and significant decline was most evident in 2011, coincident with increasing Mg concentrations in GTB. Thus, based upon taxa number, biological effects are observed around a median antecedent wet and dry season Mg concentration of 5 mg/L. ANOSIM (GTB-reference waterbody) R values for each sampling year are plotted in relation to antecedent wet and dry season Mg in Figure 5. Using the derived significant ANOSIM R versus Mg regression (Figure 5) and applying ANOSIM authors' (Clarke & Gorley 2006) criterion of minimal group separation in multivariate space below an R value of 0.3, this would indicate biological effects are observed around a median antecedent wet and dry season Mg concentration of between 3 and 4 mg/L.

It would be expected that changes in community structure (rank abundances of taxa) would be elicited at lower toxicant concentrations than loss of taxa, and so it is not surprising that the ANOSIM biological effects threshold is lower than that evident in taxa number. The implications of this for deriving solute closure criteria for billabongs are discussed below.

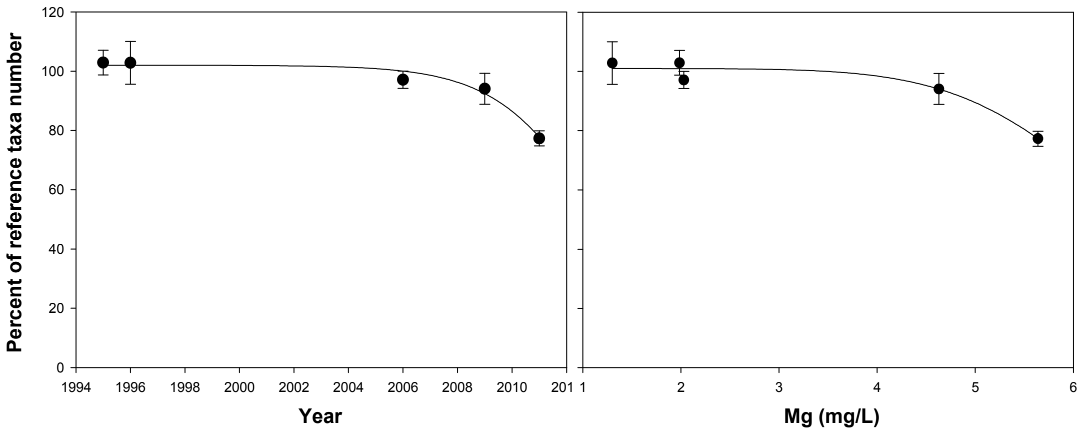

Figure 4 Mean (±SE) taxa number (% of mean reference waterbody taxa number) in GTB in relation to year of sampling and median antecedent wet and dry season Mg concentration. Line fitted according to logistic model.

In a related paper reported in this volume (Harford et al, 'The toxicity of uranium (U) to sediment biota of Magela Creek backflow billabong environments'), the authors review a number of additional assemblage-based, statistical approaches for change-point determination, useful for deriving sediment or water quality criteria. These include Non-linear CAP (NCAP), Threshold Indicator Taxa ANalysis (TITAN) and Gradient Forests (GF) – details of the methods are described in the KKN (1.2.4) paper. Just prior to finalising the present paper, TITAN and GF were applied to the full macroinvertebrate dataset used in the ANOSIM described above. Very similar thresholds for community effects were observed using these two independent approaches (ie Mg concentration of between 3 and 4 mg/L).

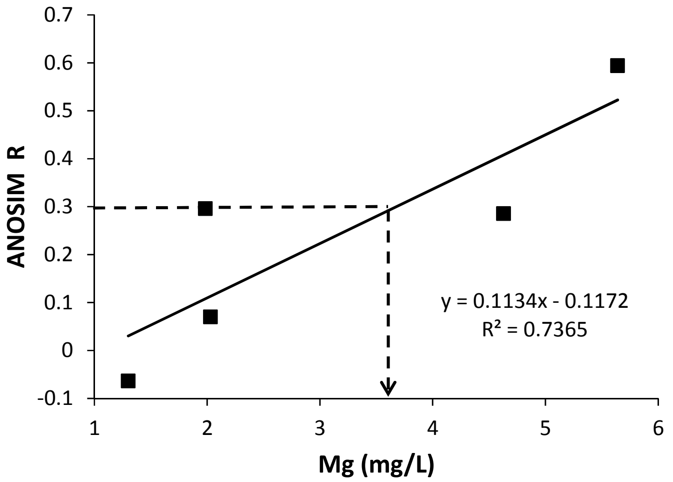

Figure 5 ANOSIM (GTB-reference waterbody) R values in relation to median antecedent wet and dry season Mg concentration

Conclusions

Macroinvertebrate sampling of waterbodies in 2011, in and around the Ranger mine lease, revealed for the first time, evidence of water quality impacts upon the fauna of GTB, corresponding to a reduction in water quality observed during the preceding dry and wet seasons. The EC and Mg concentrations measured during the entire antecedent dry season, and for several weeks in the latter part of the wet season prior to sampling, were consistent with those for which biological effects may occur based upon trigger values derived from laboratory ecotoxicity studies (van Dam et al 2010). Indeed, this deterioration in GTB water quality was evident even prior to 2009 sampling (antecedent wet and dry season Mg, 4.6 mg/L), with early indications of a reduction in taxa number (Figure 4) and differentiation in GTB-reference water body community structure (ie rise in 2009 ANOSIM R value, Figure 5). (See also comments on the 1995 elevated ANOSIM R value below.)

A Mg concentration below ~3.5 mg/L, was predicted from the ANOSIM R versus Mg regression (Figure 5) to provide minimal GTB-reference waterbody separation in multivariate space, this value being consistent with the laboratory-derived TV for Mg. Using this threshold, only the (‘no effects’) water quality data from 1995, 1996 and 2006 can be used to derive potential water quality closure criteria for U, Mg and EC. The proposed values are currently under review. It should be noted, however, that further consideration needs to be given as to whether the dry season EC and $MgSO_4$ data from 1994 should be included in the derivation of dry season TVs for those respective solutes, given: (a) their high values relative to reported guideline trigger values (Figure 1) and (b) the macroinvertebrate data from the end of the wet season in 1995 that showed some slight, but significant separation of GTB replicate samples from reference waterbody samples (Table 1).

In previous SSD reporting, it has been argued that antecedent wet season water quality, rather than antecedent dry season quality, would likely be of much greater significance to resident biota. The rationale given for this is was (i) the recency of exposure, (ii) the much higher biological diversity present in waterbodies in the wet season months compared with the dry season, and (iii) the greater perceived sensitivity of the local fauna to inputs of solutes during the wet season given that the natural exposure condition for this period is water quality similar to background receiving waters which are characterised by very low solute concentrations. However, on closer examination of the collective results to date, the influence of antecedent dry season water quality may be of greater importance for macroinvertebrate communities in billabongs than originally thought. The evidence for this is based upon: (i) the correspondence between poor dry season water quality in 1994, 2008 and 2010 and significant difference in GTB macroinvertebrate communities from reference communities in subsequent wet-dry season transition periods (1995, 2009 and 2011 respectively; albeit smaller shifts in 1995 and 2009); and (ii) the potential for poor dry season water quality to adversely affect taxa such as molluscs and worms which are resident in the billabong throughout the annual wet and dry season cycle.

A similar approach to that described above for Georgetown Billabong is being considered for deriving water quality closure criteria for natural (Coonjimba Billabong) or proposed-to-be-reinstated (Djalkmara Billabong) waterbodies elsewhere on the mining lease.

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Effects of fine suspended sediment on billabong limnology

A George & C Humphrey

Background

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 (Buckle et al 2010; see also related ecotoxicity project, last reported in Harford et al 2010). Backflow billabongs located immediately downstream of Ranger minesite, in particular, are at greatest risk from erosion of fine particulate matter from newly-rehabilitated landforms. Historical evidence for impacts that can arise in these waterbodies has previously been reported by the SSD; specifically, an intense rain-storm event passing over the Ranger site in February 1980 resulted in the failure of new earthwork structures in the catchment of Coonjimba Billabong. A consequence of this was high suspended solid loadings and significant sedimentation occurring within Coonjimba Billabong (Humphrey 1985, Nanson et al 1990).

This project aims to draw upon field-effects and observational limnological data from a local backflow billabong to assist in developing closure criteria for suspended sediment. The criteria are developed in alignment with the Australia/New Zealand Water Quality Guidelines framework (ANZECC & ARMCANZ 2000) which supports deriving criteria from biological response data collected from local waterbodies. The premise is that when a local waterbody supports the desired ecological condition, the associated water quality record can be used as success measures in rehabilitated or restored ecosystems by inferring the direct association between water quality and ecological condition.

Turbidity is a commonly-used measure to characterise suspended sediments, and thus is often used as a surrogate for suspended sediments. It may directly or indirectly affect a range of additional water quality variables such as dissolved oxygen (DO), water temperature, underwater light environment and nutrient load, with potential, therefore, to affect biological or whole ecosystem condition. The literature supports two primary pathways by which turbidity can disrupt ecological function in aquatic ecosystems, as illustrated in Figure 1.

The *first pathway* indicates that an increase in suspended sediments may result in increased surface water temperatures within the photic zone. This, in turn, promotes thermal stratification leading to decreased dissolved oxygen (DO). Diurnal temperature variation may then be insufficient to mix the stratified layers and raise the DO to biologically-sufficient levels.

The *second pathway* for turbidity-mediated disruption to aquatic ecosystem function is through reduced light availability. Reduction in available light results in consequent decreases in primary productivity, thereby disrupting the trophic flow of energy (including significant changes to diurnal DO cycles) within the ecosystem.

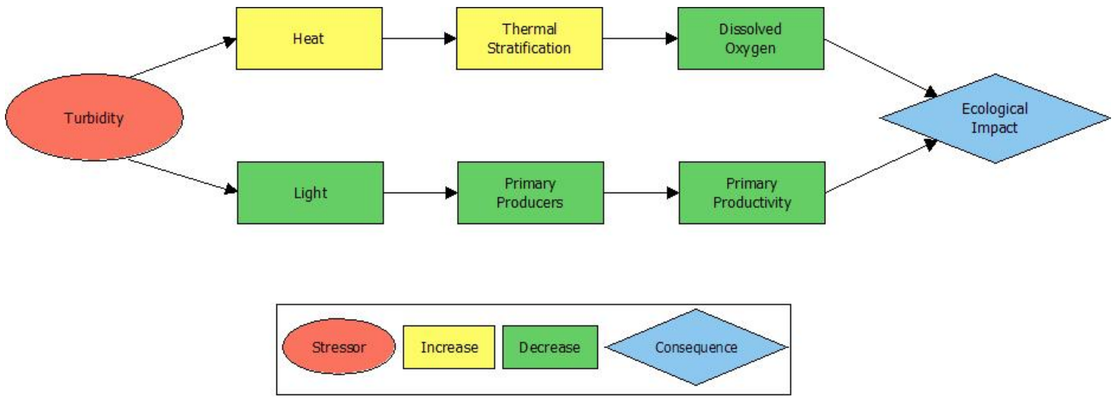

Figure 1 Two primary pathways by which turbidity, as suspended sediments, can disrupt natural ecological function within aquatic ecosystems

This study is a continuation of work conducted during 2009 (Buckle et al 2010) which examined possible relationships between turbidity, chlorophyll-a and dissolved oxygen in Georgetown Billabong (GTB), a shallow backflow billabong located adjacent to Ranger. The aim of this study is to define a threshold or range of turbidity which will maintain an ecological condition in billabongs adjacent to Ranger similar to those ecosystem values found in reference billabongs in nearby Kakadu National Park.

Buckle et al (2010) collected continuous data for DO and turbidity (using Sondes), as well as regular chlorophyll-a samples for laboratory analysis, from GTB over the 2009 dry season. They also analysed monthly chlorophyll-a and turbidity data from GTB gathered in 1981 by Humphrey and Simpson (1985). A (skewed) unimodal relationship was observed between near-surface chlorophyll-a and turbidity from data collected in 1981 suggesting inhibited phytoplankton production at turbidity values around 50 NTU (see Figure 4 below). The data collected in 2009 appeared to support the historical (1981) relationship and threshold effect (Figure 4). However, the 2009 threshold could not be precisely defined due to the absence of extreme high turbidity conditions during this year compared to earlier years. Previous years exhibited turbidity values exceeding 200 NTUs while in 2009, only one turbidity value recorded was above 30 NTU. Given the lack of sampling events in 2009 for turbidity values over 30 NTU, further measurement of chlorophyll-a and turbidity in GTB was recommended by Buckle et al (2010) to better define the relationship.

Buckle et al (2010) did not fully analyse the 2009 dataset, nor did they fully review other local data for other possible limnological effects of high turbidity. To address this issue, an analysis and review were conducted in 2012 of turbidity-mediated changes to billabong limnology in GTB between 1978 and 2012 (unpublished ***eriss*** report). This was deemed important in determining which limnological variable(s) in GTB would be useful for basing turbidity closure criteria upon. A particular focus of the analysis was to determine whether increasing dry season turbidity would lead to DO suppression in the billabong, viz one or both pathways identified in Figure 1. This relationship was implicated in GTB data gathered in 2006 and reported by Jones and Ragusa (1997). The key findings included:

1. Over the period 1978-1982, DO values were usually relatively high except during the early dry season when macrophyte decomposition was most intense (Walker et al 1984). Only partial stratification occurred in these shallow backflow systems (Walker et al 1984) and in GTB, anoxia was never evident. DO was relatively independent of both turbidity and water temperature. Thus there is no support in the early physico-chemical record of GTB for turbidity-induced DO suppression.

2. After the 1980s, DO values were only measured again in GTB in 2006 and 2009. Generally lower DO values were reported in 2006 (Jones & Ragusa 1997) and 2009 (Buckle et al 2010) compared with values reported in the earlier years (1979–82), consistent with an increase in aquatic plants in GTB observed between these two time periods (see Buckle et al 2010). During the dry season, these aquatic plants generally senesce, creating a significant oxygen demand; which may be a key factor explaining the lower DOs in GTB in more recent years.
3. Using the extensive 2009 dataset (based upon continuous logged data), an expected inverse relationship was found between DO and water temperature. This was a stronger relationship than that between DO and turbidity in GTB. The fall in DO associated with increase in water temperature was consistent with (i) reduced DO holding capacity of waters and (ii) increased rates of decomposition, at higher water temperatures.

The evidence for a direct causal link between increasing turbidity and decreasing DO in the water quality record for GTB is not strong. Jones and Ragusa (1997) suggested that high turbidity in GTB in 2009 led to a fall in billabong DO. While that is certainly plausible, other (unmeasured) factors may also have played a significant role (eg decrease in water depth, wind, increase in water temperature).

Even if an unequivocal relationship was observed between DO and turbidity in GTB, DO thresholds in themselves are not a particularly useful basis for deriving closure criteria for turbidity without a direct link to local biological effects data. To this end, the most useful information for deriving closure criteria for turbidity in GTB is based upon established surrogates for primary production; in this case using chlorophyll-a as a direct measure of phytoplankton biomass. Thus a sampling program was re-initiated in GTB over the dry season of 2012 with a particular focus on measurement of chlorophyll *a*, as a direct measure of phytoplanktonic biomass and an indirect contributor to surface water primary productivity and photosynthetic activity, over a broad range of turbidity values.

Two potential limitations of the present approach to deriving turbidity closure criteria should be noted:

(i) Opportunities to examine natural increases in turbidity in ARR surface waters (ie sufficiently high and sustained turbidity to elicit biological responses) are only available in the dry season, whereas erosion-induced, elevated turbidity associated with rehabilitation will occur during the wet season. Whether the sensitivities of the different biological communities inhabiting local billabongs in the wet and dry seasons are similar is not known; and

(ii) The source of dry season turbidity in GTB, re-suspended fine silt and clay from billabong sediments, differs from the source that would potentially arise from erosion of rehabilitated mine landforms, ie laterite and Cahill schists. Within the mine source, moreover, turbidity/sediment concentration relationships may vary significantly (Riley 1997). Whether suspended sediment from different sources elicits ecological responses at the same concentration or turbidity values, is not known.

Methods

The current Georgetown billabong study that re-commenced in 2012 has been located at the same site as that established in 2009 by Buckle et al (2010), ie in the deeper billabong waters near the billabong outflow (S 12^{0}40.705', E 132^{0}55.885'). The investigation uses in situ continuous measurements of water quality parameters supported with weekly, field-based water quality sampling. Sampling commenced in August 2012 and at the time of reporting (December 2012), aspects of the measurement program were still in place.

The in situ continuous measurements are collected from two depths: surface (10 cm below the water surface) and depth (1 m below the water surface). These depths represent the active photic zone in which the majority of primary production should take place. Each depth is equipped with a multiprobe (Hydrolab datasonde 5X) to measure temperature, dissolved oxygen, turbidity, EC and pH at ten minute intervals. Data are downloaded weekly to track changes in the measured parameters, and to detect any equipment malfunction.

The weekly field sampling is conducted between 8.30 am and 10.30 am to standardise the time of day with baseline studies conducted in 1980 and 1981 (eg Humphrey & Simpson 1985) and in subsequent routine water quality monitoring carried out by ERA. Each week water samples are collected from 10 cm, 50 cm and 1 m depths for analysis of chlorophyll-a, nutrients (NH_3, NO_3, PO_4, Total N, Total P), major ions (Cl, K, Ca, Na, HCO_3, SO_4, CO_3) and metals (Pb, Mg, Mn, Fe, Cu, Al, Zn, U).

The weekly sampling also includes collection of light profile data. Measurements are made at 10 cm below the surface, 25 cm, and then every 25 cm until the bottom of the billabong is reached. Measured parameters at each depth interval include temperature, dissolved oxygen, turbidity, EC and pH (Hydrolab quanta multiprobe) and photosynthetically-active radiation (PAR; LI-193 underwater Spherical Quantum Sensor). The instrument 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 gathered for this project (to December 2012) and hence only preliminary analysis and interpretation of available 2012 results are provided here, by way of comparison with results gathered in earlier years.

Changes in turbidity over time

Figure 2 illustrates the historical turbidity data collected between 1980 and 2012 and is plotted for each month.

As discussed by Buckle et al (2010), the historical data illustrate that turbidity during 1980 and 1981 increased earlier in the dry season, and reached higher levels, compared to later years. In later years, significant increases in turbidity occurred only towards the very end of the dry season (see 1983–2008 plot line). The earlier peaks in 1980 and 1981 were associated with trampling of the banks and wallowing by water buffalo. Water buffalo were mostly removed by 1982, reducing early dry season turbidity. As shown in Figure 2, the data collected in 2009 are consistent with the seasonal pattern of post buffalo turbidity in the Georgetown Billabong while the data for 2012 indicate a slight shift towards the patterns seen in 1980 and 1981.

Buckle et al (2010) suggested that both the removal of buffalo and associated changes to aquatic vegetation may have a role in the compressed time period of higher turbidity. The culling of buffalo eliminated a direct sediment-disturbance agent and allowed sedges to become more widespread in the backflow billabongs, thus increasing the deposition of silts that contribute to high turbidity (Finlayson et al 1994). This shift in turbidity and the suggested role of vegetation highlight the need to further consider the larger scale ecological responses associated with turbidity, such as annual and seasonal vegetation changes.

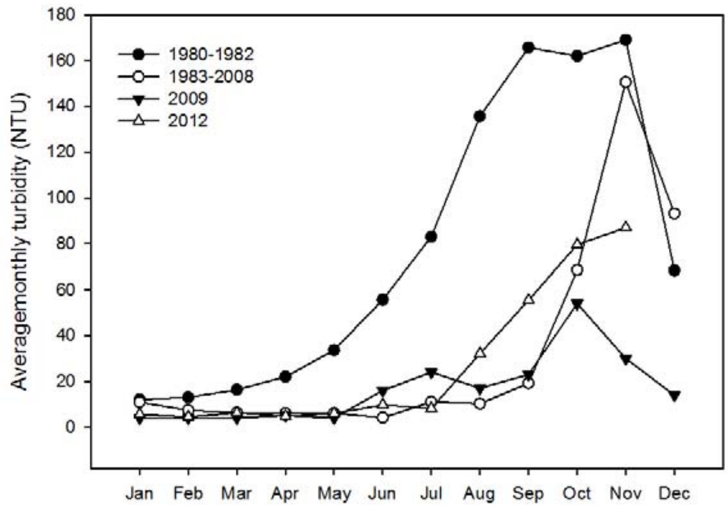

Figure 2 Average monthly turbidity (NTU) for Georgetown billabong for selected years between 1980 and 2012. Data for 1980, 1981, 2009 and 2012 are those collected by SSD. Energy Resources of Australia data were used for 1982–2008.

Turbidity and chlorophyll-a interactions

Surface turbidity and chlorophyll-a data gathered to date in 2012 are plotted according to time in Figure 3.

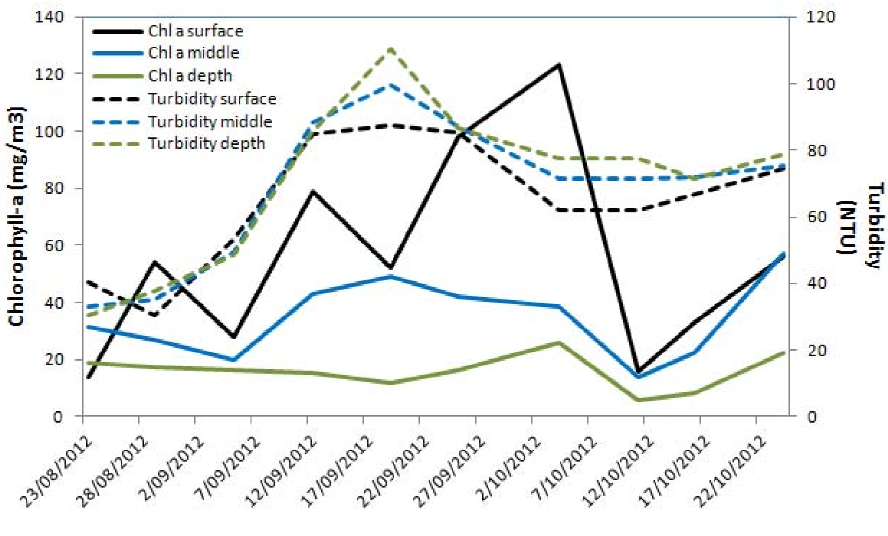

Figure 3 Chlorophyll-a and turbidity measured at different depths during each weekly field sampling period

Turbidity data represent spot readings taken at the same time as the (generally) weekly chlorophyll-a samples. Little variation is evident in turbidity at different depths, with a peak for all depths observed in mid September. The chlorophyll-a response differed amongst depths with chlorophyll-a values declining with increasing depth, unsurprising as the amount of light reduced exponentially between 10 cm and 1 m (data not presented here) (Figure 3). While chlorophyll-a did not vary much at depth over time, there was marked weekly variation observed at the surface (Figure 3).

Surface chlorophyll-a and turbidity data from each of the three sample years are presented in Figure 4. Data from 2009 suggested that phytoplankton are inhibited at values between 25 and

70 NTU (Buckle et al 2010). This was supported by data collected in 1981 by Humphrey and Simpson (1985) which suggested inhibited phytoplankton production at values around 50 NTU (Figure 4). The 2012 data, however, do not reflect any such threshold effect, with phytoplankton biomass peaking at turbidity values approaching 100 NTU. Field staff observed surface and near-surface red algal 'clouds' or scum in GTB in the late 2012 dry season. Kessell and Tyler (1982) have previously identified these natural scums and clouds that may occur in ARR billabongs in the late dry season as comprising almost pure cultures of phytoflagellates, including species from either *Pyramimonas*, *Chlamydomonas*, *Chlorogonium* and/or *Eugena* genera. Samples have been collected to verify these indications. If indeed the late 2012 chlorophyll-a values are reflecting low diversity and turbidity-tolerant phytoplankton populations and communities, the data are of no value in determining threshold effects of turbidity upon phytoplankton communities.

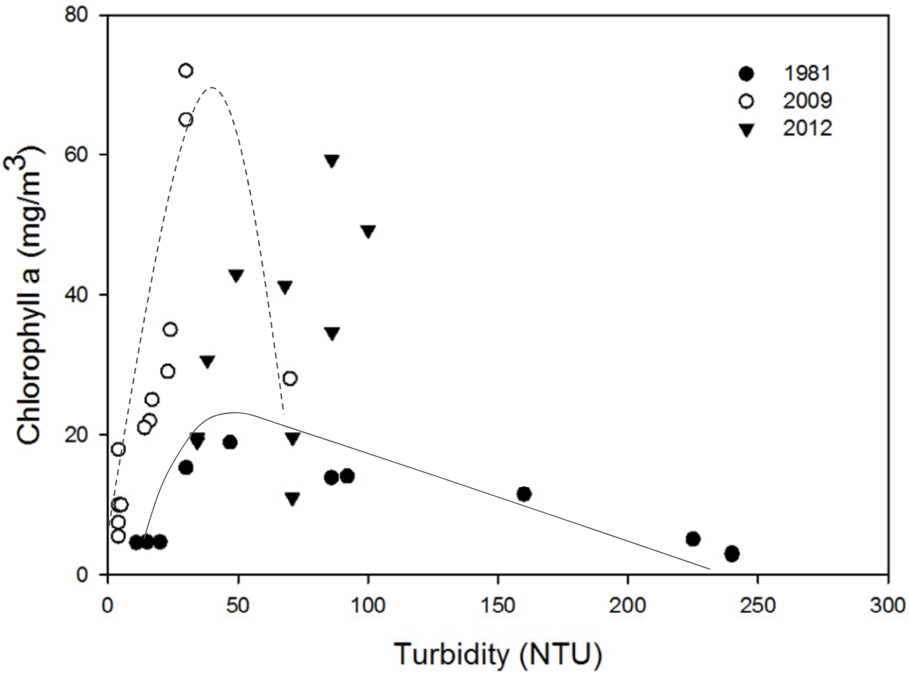

Figure 4 Relationship between turbidity (NTU) and chlorophyll-a (mg/m³) in Georgetown Billabong in 1981, 2009 and 2012. Data shown for 2009 are fortnightly measures, data for 2012 are weekly field measures and data for 1981 are monthly measures. Curves for 1981 and 2009 drawn by eye for illustrative purposes only.

Further work

It is unlikely that 2012 chlorophyll-a and turbidity data collected from GTB will contribute any useful information towards the setting of closure criteria for turbidity in receiving water billabong environments. Interim criteria may be derived from the 1981 and 2009 chlorophyll-turbidity data reported above, together with other regional information on this topic (eg from Stowar (1997) who reported the effects of high suspended sediments on stream macroinvertebrate communities in Jim Jim Creek, Kakadu National Park).

Nevertheless, with the recent purchase of chlorophyll-a probes for continuous measurements, further closure-criteria-related work in GTB is proposed:

(i) to characterise chlorophyll concentrations during the wet season in relation to natural elevated turbidity events (associated with backflow from adjacent Magela Creek, albeit relatively brief episodes); and

(ii) to enhance the dry season chlorophyll-turbidity record, including the capture of responses of phytoplankton to short-pulse exposures of high turbidity that characterise the dry season turbidity record in GTB.

Continuous measurement of chlorophyll-a in GTB will improve understanding of factors affecting high temporal variability of phytoplankton communities, such as observed in 2012. Further, continuous and contemporary water quality data for GTB may also be used to inform re-establishment techniques and water quality criteria for sentinel and re-instated waterbodies, the subject of new research proposed by SSD and ERA (see George & Clark, 'Aquatic ecosystem establishment: A KKN review', this volume).

The 2012 results also support the need to more closely consider the role of nutrients in influencing chlorophyll-a concentrations. High nutrient concentrations are characteristic of the dry season and the internal loading of nutrients (N and P) are maximised by sediment re-suspension during periods of increased turbidity (Walker & Tyler 1982). However, while suspended sediments may account for a substantial portion of biologically-available nutrients, the high nutrient values concurrent with high turbidity are not likely to induce a biological limitation. The potential link between the peaks and falls of chlorophyll-a need to be evaluated against nutrient availability (data collected but not yet analysed) during each of the sampling periods.

The potential toxicity of Mg to phytoplankton in GTB needs to be also considered in interpreting turbidity-chlorophyll data collected from GTB in recent years. In 2011, for example, high Mg concentrations in GTB were implicated in reduced macroinvertebrate diversity in the billabong (Humphrey et al, 'Developing water quality closure criteria for Ranger billabongs using macroinvertebrate community data', this volume).

The data presented within this paper represent only a minor amount of the data collected for this timeframe. The results are only preliminary and have subsequently resulted in the identification of additional research questions that can be addressed using the existing data. For example, the high variability in surface chlorophyll-a values indicates a gap in understanding the dynamics of chlorophyll-a response. To address this in the on-going sampling, chlorophyll-a will be added as a continuously monitored parameter. In addition, a pilot project has been initiated using flow cytometry to assess chlorophyll-a and phytoplankton communities.

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

C Humphrey

Background

A number of projects are currently underway to address aspects of rehabilitation associated with the future closure of the Ranger Project Area, including ecosystem reconstruction and final landform design and revegetation. Revegetation research using the concept of analogues has been progressed by both ERA Pty Ltd (ERA) and ***eriss***, separately or in collaboration, since the early 1990s, with key ARRTC summary papers published in every ***eriss*** research summary since 2004–2005 (Humphrey et al 2006). The sequence of research has shifted from broad regional (ARR) to smaller scale, and has considered the role of soil chemistry, texture and structure, as well as landscape terrain variables, in accounting for vegetation community patterns. The research to date has only considered shrubs (>1.5 m in height) and trees, in accordance with the recommendations of Reddell and Meek (2004) that early establishment focus just on this vegetation category. Work on trees and shrubs is drawing to a close and the following report summarises research conducted on the (small-scale) Georgetown analogue area located next to the Ranger site.

The Georgetown analogue area, a ~400 hectare area of natural vegetation located on the south-eastern edge of the Ranger mine, is providing much of the reference data about local vegetation communities. These vegetation data have been gathered separately by ERA and ***eriss*** but have been combined for data analysis. Unlike the flat lowland Koolpinyah surface found over most of the Ranger lease, this area has terrain characteristics that better match those of the (draft) proposed final landform. In particular, the vegetation communities associated with the area's low relief are representative of the variety of plant forms found in lowland and lower hill terrain across the Alligator Rivers Region (ARR) (Humphrey & Fox 2010).

The primary objectives of the work being conducted in the analogue area are to:

1. Identify and derive quantitative terrain parameters (eg elevation, relief, aspect) which provide a landscape-based reference for specifying design criteria for the final rehabilitated landform.
2. Characterise the plant communities and identify the key environmental determinants of those communities from the terrain descriptors derived in 1.
3. Use the findings from (1) and (2) to assist with,
    a. selecting the most appropriate species for revegetation of the Ranger mine landform post decommissioning, and
    b. the development of revegetation closure criteria and a suitable post-closure, performance monitoring regime.

In relation to item 1 above, derivation of landscape terrain descriptors of the analogue area by SSD using a high resolution DEM (from LiDAR) was reported to ARRTC in December 2011 (Humphrey et al 2012) and is not further considered here.

Relationships need to be identified between the occurrence of key vegetation communities representative of the environments likely to exist in the rehabilitated landscape and key geomorphic features of the landscape (eg soil type, slope, effective soil depth, etc.). Identifying the intrinsic environmental features associated with particular vegetation community types will inform two aspects of the rehabilitated landform at Ranger:

(i) the environmental conditions required to support key communities, and alternatively,

(ii) the community types that best suit particular environmental conditions.

The caveat to apply here is that the range of likely environmental conditions within the proposed rehabilitated landform needs to be similar to the natural analogue area; otherwise the natural analogue will be unable to provide information relevant to site rehabilitation.

Summary of progress reported to ARRTC27

Key aspects of this study reported to ARRTC27 (Humphrey et al 2012) included:

1. Derivation of landscape terrain descriptors of the analogue area using a high resolution DEM (± 0.25 m horizontal; ± 0.15 m vertical);
2. The use of multivariate classification techniques to identify four distinct vegetation communities on the Ranger analogue site: *Melaleuca* woodlands associated with riparian and floodplain zones subject to seasonal inundation, a common mixed eucalypt woodland community, and two dry mixed eucalypt woodland types with dominant species that are (semi-)deciduous in nature.
3. Modelling of plant-environment relationships for communities and individual species from up to 54 analogue sites using soil chemistry, texture and structure, depth to groundwater and landform (terrain) variables.

The species- and community-level modelling conducted in the study (from 3 above) were consistent with one another in highlighting key – but obvious – differences between *Melaleuca* woodlands and the dominant mixed eucalypt woodland type. It was noted that the species-level modelling was based upon data from the small and confined Georgetown analogue area, such that apparent local 'preferences' of species for particular landform conditions might not necessarily reflect the wider environmental ranges over which the species are known to occur in northern Australia, nor accurately reflect the full range of conditions that favour particular species.

Whilst noting this issue, it was also highlighted to ARRTC that the most useful aspect of the modelling was that it defined the local environmental conditions for which common plant species in the adjacent natural landscape occurred. The Ranger Environmental Requirements for revegetating the site according to assemblages and structure similar to the adjacent natural landscape would best be met by mimicking these plant-environment relationships on the revegetated landform. In doing so, it was noted that, this match might have no stronger basis than resemblance and aesthetics, as distinct from a strong eco-physiological basis for the occurrence of particular species in the landscape. It was suggested to ARRTC that further modelling might need to be no more sophisticated than defining the environmental ranges (viz statistical ranges and medians for landform variables) for the occurrence of dominant plant species.

A note on soil-plant relationships

A caveat raised above for this analogue study was that the range of likely environmental conditions within the proposed rehabilitated landform needs to be similar to the natural analogue area(s) or else the natural analogue(s) will be unable to provide information relevant to site rehabilitation. An important difference between analogue sites and the Ranger final landform will be in soil characteristics – chemistry, particle size distribution, water retention properties, texture, morphology (including depth) and microbiology. While these differences have not been quantified, there is evidence that (i) soil properties (as opposed to position in the landscape) are not important determinants of plant communities of the ARR, and (ii) Ranger mine-derived substrates can support sustainable growth of local plant communities. Thus:

1. As part of the investigations conducted under this KKN, up to 39 soil variables were collected for representative natural analogue sites. These variables reflected soil chemistry (major ions and nutrients, 18 variables), particle size distribution (4 classes), soil water retention properties (10 variables) and soil morphology and surface drainage classes from published classifications representing horizon thickness, gravel and texture, and soil permeability (total of 7 classes). The relationship between soil properties and the four plant communities (from above) was sought from multivariate analysis, at two spatial scales:
    a. For 28 analogue sites dispersed across the ARR, Humphrey et al (2009) observed independence of vegetation community composition and structure from the underlying soil properties that were measured and used in the analysis; and
    b. For 22 analogue sites confined to just the Georgetown analogue site, Humphrey et al (2012) similarly observed that most of the significant soil and variables only appeared to distinguish sites of seasonal inundation, where *Melaleuca* woodland occurs, from the other woodland community sites. (The occurrence of *Melaleuca* woodlands on low-lying, seasonally-inundated locations is well understood and hence this result is not particularly useful.)
2. ERA and its consultants have demonstrated over the years that local trees and shrubs can be successfully grown (and will flower and fruit) on harsh and stony mine-derived substrates (including neat waste rock) on the Ranger minesite. Some revegetation plots on the minesite have matured over a period of more than two decades. The most recent revegetation demonstration, occurring on the Ranger trial landform, has been underway for over 2.5 years. Growth rate of tubestock plants on waste-rock-only and on waste rock mixed with laterite materials has been similar. Higher incidence of mortality of tubestock plants has occurred on the mixed waste rock and laterite materials than waste-rock-only materials, due to waterlogging which is exacerbated by the properties of exposed laterite. This has lead to a proposal for the final landform to create a thin layer of 30–50 cm of laterite material ‘sandwiched’ between waste rock materials at about 1.5–2 m below surface (Energy Resources of Australia 2012). The trial landform provides an essential adaptive-management platform, to monitor ecosystem establishment and sustainability, including vegetation and soil development, over the long-term.

Werner and Murphy (2001) observed the two dominant tree species of the common mixed eucalypt woodland of northern regions of the NT, *Eucalyptus tetrodonta* and *E. miniata*, in a diverse range of soil types. In one (ARR) location, vigorous growth was reported in gravelly sandy soils just 0.5 m thick, overlaying an impermeable and root-impenetrable ferricrete layer. Noting the presence of mixed eucalypt woodlands in such extreme soil conditions (with negligible late dry season water and nutrient availability), the authors remarked on the unusual and adaptable physiologies of northern Australian savanna tree species. In comparable situations on other continents, the savannas comprise treeless grasslands (Werner & Murphy 2001).

Progress made since ARRTC27

The ARRTC27 summary paper detailing study components 1-3 from above and conclusions (Humphrey et al 2012) was peer reviewed by two well-regarded northern plant ecologists, Drs Jeremy Russell-Smith and Owen Price. They concurred with the results and conclusions drawn from the study, noting:

> Many of the ARR woodland tree and shrub species are quite tolerant of position and occur over a range of different environmental conditions. Russell-Smith (1995) observed that, generally, northern woodland vegetation communities are floristically continuous rather than readily definable / mappable discrete types;
>
> Over broad geographical ranges, individual species can have quite different occurrences in the environment, eg *Eucalyptus tectifica* appears to prefer more poorly drained, heavier soils in the ARR but, in southeastern Arnhem Land, it is distributed all over the hills (like *Corymbia confertiflora*).

Instead of defining the environmental ranges for the occurrence of dominant plant species, both researchers offered less prescriptive approaches to revegetation strategies. Apart from placing *Melaleuca* woodland species in locations predicted to be seasonally-inundated / poorly draining, matching the particular woodland species with their correct positions in the landscape (according to the plant-environment relationship model) would not be essential. Rather, a possible strategy would be to simply plant out species across most of the rehabilitated site in similar proportions to the densities and frequencies found in natural adjacent woodlands.

Based upon these recommendations, this body of work for trees and shrubs appears to have reached a conclusion, though the influence of fire (frequency and intensity), weeds (particularly high fuel load grasses), climate change and Traditional Owners views may be additional agents that may guide final species selection.

ERA is currently considering establishment needs for understorey species, including grasses and forbs. They note (P Lu, pers comm):

1. Efforts to establish understorey will not need to be as intensive as those necessary for trees and shrubs. ERA plans to wait 3 years after the establishment of trees and shrubs before actively introducing an understorey, ie until most tree species achieve a diameter of over 5 cm at which stage survivability (after fire and without dry season irrigation) is high.
2. An available inventory viz the extensive analogue information (species and relative abundances) on understorey species gathered by ERA and ***eriss***.
3. The passive colonisation of revegetation plots, including the trial landform, over the years by local native understorey species. This recruitment is currently being documented for the trial landform. Such passive colonisation will hopefully reduce efforts to actively plant understorey species.
4. ERA will be sowing seeds of some grass species this wet season on the trial landform.

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Aquatic ecosystem establishment: A KKN review

A George & K Clark[1]

Background

The aim of this discussion paper is to initiate the development of a collaborative research program between SSD and ERA targeting aquatic ecosystems constructed for Ranger minesite rehabilitation. Research is required across all phases of rehabilitation and closure, and will draw upon and integrate the aquatic ecosystem components embedded across various KKNs.

Freshwater aquatic ecosystems are regarded as the most threatened ecosystem type in the world (Abramovitz 1996). Often values are placed on these ecosystems in an attempt to protect them. However, this generates an idea that one aquatic ecosystem can be simply replaced with another with similar or equal values (Mitsch & Gosselink 2000) (ie the removal of one ecosystem is easily replaced by another through remediation or restoration activities or that replacement is unnecessary). Such a view fails to consider the landscape-level importance of aquatic ecosystems and the value that extends beyond the presence or absence of these ecosystems. The value placed on aquatic ecosystems can occur at various scales (population, ecosystem or biosphere) and may have different levels of benefit (land owner, local/regional, global) (Mitsch & Gosselink 2000). Once such value is determined for a region, the framework for restoration, re-creation or rehabilitation can be initiated. In the context of Ranger, high conservation values are assigned to Kakadu National Park, the area into which the rehabilitated minesite will be ultimately incorporated.

The design of restoration or re-creation programs for aquatic ecosystems often uses reference river reaches. However, where there are no directly comparable reference areas or when the relationship between an available reference site and an impacted site is uncertain, an analytical reference approach can be applied (Downs et al 2011). The analytical reference approach integrates site-specific baseline data to conceptual models as a means of linking habitats to ecosystem processes. This approach provides a means of constraining the restoration design to key processes and ecosystem drivers which should improve the likelihood of restoration or rehabilitation success (Downs et al 2011).

Aquatic ecosystems around the Ranger mine site have been extensively studied over the last 30 years. There is a large amount of historical information along with current data that may be used as baseline data for designing aquatic ecosystem restoration projects. In order to understand the types and amount of baseline-type information available for restoration, an evaluation of the current KKNs was initiated.

A review of the current KKNs found that components related to aquatic ecosystems are included across multiple KKNs (Table 1). Where aquatic components are embedded within KKNs whose focus is primarily on terrestrial aspects, the aquatic ecosystem programs have not received the same on-going research attention beyond that required for water quality monitoring and periodic event-based surveys. While this may simply reflect program priorities at the time, there is now a need to prioritise aquatic ecosystem programs to ensure sufficient information is available for developing a rigorous rehabilitation framework.

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Table 1 KKNs identified as directly containing relevant aquatic ecosystem components or contributing to aquatic ecosystem learnings

KKN	Aquatic component	Relevance for proposed KKN
1.2.1 Ecological risks via the surface water pathway	• Aquatic conceptual models • Surface water hydrodynamic model	Input and additional information
1.2.3 Wetland filters	• Attenuation • Biogeochemical pathways	Direct
1.3.1 Surface water, groundwater, chemical, biological, sediment, radiological monitoring	• Water quality (surface water and groundwater)	Input and additional information
2.2.3 Water quality in seepage and runoff from the final landform	• Groundwater quality	Input and additional information
2.3.4 Hydrological/hydrogeochemical modelling	• Groundwater flow model	Input and additional information
2.4.2 Passive treatment of water from the rehabilitated landform	• Review of research • Wetland filters	Direct
2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective	• Surface water quality • Flora and fauna • Sediments	Direct
2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites	• Aquatic analogue ecosystems • Rehabilitation objectives	Direct
2.5.3 Establishment and sustainability of ecosystems on mine landform	• Aquatic vegetation, fauna and flora habitats • Environmental requirements for rehabilitation • Reinstated and reconstructed waterbodies	Direct
2.6.1 Monitoring of the rehabilitated landform	• Monitoring of vegetation	Indirect – methods for monitoring aquatic vegetation
2.6.2 Off-site monitoring during and following rehabilitation	• Assess rehabilitation success • Dispersion of contaminants by water	Indirect – methods to identifying appropriate aquatic analogues
2.7.1 Ecological risk assessments of the rehabilitation and post rehabilitation phases	• Conceptual pathways models (aquatic)	Indirect – providing framework for rehabilitation direction and assessment

Habitat restoration is the premise used to support a program of aquatic ecosystem establishment. Restoration is a complex process for which there are still large gaps in understanding, even following several decades of active and applied research (Hilderbrand et al 2005). The concept of restoration has gained significance through legislative policy requiring restoration or mitigation in response to activities which significantly alter the natural landscape. Often these regulatory requirements lack a sound underpinning of ecological function necessary to set appropriate targets, measures and criteria for restoration success.

Restoration science requires the integration of information on physical, chemical and biological indicators as well as their interrelationships. In the Alligator Rivers Region, large databases have been gathered on water quality, fish and macroinvertebrate communities of aquatic ecosystems. Only sporadic work has been conducted on aquatic vegetation, despite this ecosystem component being a popular measure of relative restoration success within aquatic habitats. Aquatic plants serve important aesthetic and water-cleansing roles, and provide important habitat for resident biota, particularly in this region. The relative ease of identification and well-established quantitative methods for evaluating vegetation communities contributes to the selection of plants as indicators of success.

Of key importance for restoration and rehabilitation, is the understanding of the link between ecosystem drivers and vegetation dynamics. The dynamic nature of tropical systems implies that community assemblages will be equally dynamic, changing with wet and dry season cycling, the predisposition of the timing and intensity of seasonal events and competition (Finlayson et al 1989). Not all species represented in the seedbank or in surveys will be represented in any given year. Also, as research related to aquatic vegetation has not progressed significantly since 1990 (Finlayson et al 1989, 1994, Cowie et al 2000) possible changes related to recent or longer term drivers (eg cumulative effects of high rainfall years, large storm events) have not been assessed. For example, at least one species of prominent emergent macrophyte not previously noted in the region, *Eleocharis sundaica*, was found in both on-site and off-site billabongs during the 2012 dry season (SSD and ERA unpublished data). As this particular species was not reported by Finlayson et al (1989, 1994), it potentially highlights the dynamic nature of these aquatic ecosystems.

Key Knowledge Needs should address specific research questions aimed at providing specific required information. As written, the current KKNs relevant to the role of aquatic ecosystems for minesite rehabilitation are too diffuse to achieve such outcomes.

KKN Revision

We propose that aquatic ecosystem components of five current KKNs be re-organised into two targeted KKNs for rehabilitation purposes. As shown in **Table 1**, the five KKNs directly relevant to aquatic ecosystems include:

- KKN 1.2.3 Wetland filters
- KKN 2.4.2 Passive treatment of waters from the rehabilitated landform
- KKN 2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective
- KKN 2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites
- KKN 2.5.3 Establishment and sustainability of ecosystems on mine landform.

The KKNs should be refocused to reflect more specific, measurable and achievable research agendas. Key Knowledge Needs underpinned by relevant research questions which aim, specifically, to address the knowledge gaps will assist in designing focused research programs. The restructuring process for the Aquatic Ecosystem KKNs should also incorporate an overall review of the pertinent KKNs to determine the on-going relevance of the current list. Such a review is currently being undertaken, and will be further progressed following the mine rehabilitation/closure risk assessment in early 2013. The revised KKNs should be written in a manner conducive to achieving expected outcomes represented within the KKN title. For now, the proposed changes will remain within current KKN 2.5.3 until they can be more appropriately captured elsewhere.

The proposed, revised rehabilitation-related KKNs are as follows:

4 Aquatic Ecosystems – provision of environmental services

5 Aquatic Ecosystems – characterisation and functionality (for use in aquatic ecosystem design and closure criteria)

Revised KKN1 draws upon KKNs 1.2.3 (wetland filters) and 2.4.2 (passive treatment of waters from rehabilitated landform) and relates to how aquatic ecosystems provide environmental services, specifically, how they protect the environment by ameliorating the effects of contaminants. While focus is on the service that wetlands provide in sequestering contaminants, the implications of processes that could result in remobilisation of initially-bound contaminants also need to be considered. For example, what are the ecological risks associated with an aquatic ecosystem (natural or rehabilitated) switching from a contaminant sink to source?

Revised KKN2 incorporates parts of KKNs 2.4.2 (passive treatment of waters from the rehabilitated landform), 2.5.1 (closure criteria), 2.5.2 (analogue sites) and 2.5.3 (establishing sustainable ecosystems). KKN 2.4.2 is included in both revised KKNs because the wet/dry cycling affects not only how contaminants are mobilised within an aquatic system, but is also a key driver for ecosystem development and function. The other current KKNs were selected because they each contained some requirement for understanding the information needs for aquatic ecosystem establishment. These freshwater systems are dynamic across various temporal and spatial scales. So, with such drivers as wet/dry cycling, this revised KKN aims to use aquatic analogue sites to characterise aquatic ecosystems and determine their range of natural ecological variability.

The interrelationship between, and integration of, the two KKNs ensures that a consolidation of relevant aquatic ecosystem information is obtained for closure planning and the development of closure criteria. Integration and collaboration between these revised KKNs and other single component studies (eg bushtucker, sediment quality, macroinvertebrate and fish surveys) will provide ecological context to research findings.

Potential research areas and questions have been scoped for the revised KKNs (Table 2). The topics to be included are not limited to these questions since they simply illustrate the types of information that may address key knowledge needs for aquatic ecosystems. For example, within *KKN1 Aquatic Ecosystems – provision of environmental services* much of the focus is on aquatic plants. However, ecosystem services encapsulate more than plants. These questions illustrate the immediate need for information about plants, but the title illustrates an on-going requirement for broader aquatic ecosystem representation. This is of particular importance since services are often key elements of ecological risk assessment.

In addition, further review is required to determine the extent to which information is already available to address research questions. A review will identify a way to prioritise where additional research may be undertaken. SSD has ecological information on wetlands and ERA has been very active in wetland filters and chemical attenuation properties, both areas that can contribute to such a 'gap analysis'.

Workplan for 2013

In the context of rehabilitation, there is a need for a standardised, holistic approach to aquatic ecosystem research being conducted by ERA and SSD to ensure not only that the science is robust but also to ensure that the outcomes have the best chance of being practically implemented on site. This can be achieved through ERA and SSD directly collaborating on aquatic ecosystem research, and also through liaising with other studies linked to aquatic ecosystems (ie KKNs illustrated in Table 1 that could contribute to aquatic ecosystem studies).

Table 2 Potential research questions and issues that could be addressed within each of the revised Aquatic Ecosystem KKNs. It should be noted that these questions provide only an introduction of the type of issues that may be addressed. It is not probable that all issues would be incorporated into an Aquatic Ecosystem program. In addition, the issues and questions may change in response to findings over time. The questions are not listed in any particular order.

KKN1 Aquatic Ecosystems – provision of environmental services	• Where does acidification caused by the oxidisation of sulfides in sediments occur and what are its impacts on ecosystem reestablishment • The role of bioturbation on contaminant remobilisation • What effect does remobilisation of contaminants upon rewetting have on ecosystem reestablishment • What is the functional relationship between the degree of sediment consolidation and contaminant attenuation? • What is the rate of contaminant uptake by key aquatic plant species and is this uptake relevant at an ecological process level? • Where do aquatic plants store contaminants that are attenuated • Do contaminants negatively impact propagule development? • At what point does an aquatic ecosystem change/switch from a contaminant sink for attenuation to a contaminant source through concentration, mobilisation and release of contaminants?
KKN2 Aquatic Ecosystems – characterisation and functionality	• A need to re-establish current vegetation baseline through the establishment and monitoring of dynamic analogue sites • What are the tolerances and sensitivities to contaminants of key aquatic vegetation species? • What are the key aquatic vegetation species? • Do tolerances and sensitivities vary in different growth stages? • What are the growth dynamics (and life cycles) of key species? • What is the conceptual understanding of 'self-sustaining' aquatic ecosystems? • How does aquatic vegetation community structure change in response to different impacts? • What are their growth strategies? • Is there an ecological risk associated with 'new' species contributing to natural (uncontrolled/managed) establishment? • What successional stages can be expected within mature and newly constructed/restored wet/dry tropical wetlands? • What conditions are required for seed banks dynamics (germination and establishment of propagules)? • What are the minimal requirements for vegetation establishment? • What species are represented in the current seed-banks?

To initiate the proposed collaborative research on aquatic ecosystems an initial focus on aquatic ecosystem analogues is recommended.

Part 1, identification and selection of aquatic analogue sites, will be delivered through collaboration with the teams addressing KKNs 2.6.1 and 2.6.2, as well as information provided by supporting research in aquatic ecosystems, both current and historical. The SSD SSDI group mapped and identified sites with current vegetation information. They will further work with the authors to populate tables allowing criteria-based selection of aquatic analogue sites. This provides a rigorous approach to selection of aquatic analogue sites across various spatial and temporal scales and also catalogues existing knowledge about aquatic ecosystems. This work links to SSDI projects on updating vegetation distribution across the region. A review of remote sensing data and relevant literature is in progress.

Part 2, standardisation of research objectives/methodologies across analogue and mine-affected sites, will result in the development of a strategic research program to address and standardise aspects of aquatic vegetation research across the identified KKNs. Collaboration will be sought with other researchers invested in aquatic ecosystem aspects to identify potential overlap that could provide cost efficiencies and information sharing opportunities.

Even though two KKNs are suggested, they are strongly related and therefore the development of KKN2 facilitates the development of KKN1. The focus on analogue sites and aquatic vegetation simply reflects the most obvious direction for research development. Analogue site development uses aquatic ecosystem characteristics to identify and develop environmental/ecosystem services required for long-term restoration and rehabilitation.

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Estimating radionuclide transfer to bushfoods and ingestion doses to the public

C Doering & A Bollhöfer

Introduction

The ARR is an area of past and present uranium mining activity. It is also an area where there is customary harvesting of aquatic and terrestrial bushfoods by local Aboriginal people for sustenance. The accumulation of radionuclides in bushfoods and their consumption means that the ingestion pathway should be addressed in member of the public dose assessments for current and future exposure situations. In particular, the ingestion dose from uptake of radionuclides in bushfoods should be assessed for areas impacted by the Ranger uranium mine to provide the evidence base needed to determine the acceptability of current operations and proposed closure and rehabilitation options.

Ingestion dose can be calculated from information on diet and radionuclide activity concentrations in food items and using recommended dose conversion factors for intakes of radionuclides (ICRP 1996). Radionuclide activity concentrations in food items can be determined by direct measurement. They can also be estimated using transfer factors applied to measured or modelled radionuclide activity concentrations in environmental media such as soil or water. The transfer of radionuclides from the environment to food items is commonly parameterised using a concentration ratio (IAEA 2010), which is the ratio of radionuclide activity concentration in the edible portion of the food item (wet or dry) to that in the surrounding environmental media.

eriss has been measuring activity concentrations of natural-series radionuclides in aquatic and terrestrial bushfoods and environmental media from the ARR for around 30 years (Bollhöfer et al 2011, Martin et al 1998, Ryan et al 2005a,b). The data enable derivation of ARR-specific concentration ratios for bushfood items which can be used in ingestion dose assessments. The data also reduce reliance on the use of generic transfer factors, meaning that the estimated ingestion dose should be more representative of the dose received.

The ***eriss*** data on radionuclide activity concentrations in bushfoods and environmental media from the ARR have been consolidated into a consistent, quality controlled and queryable tool. The tool has been named **B**ioaccumulation of **R**adioactive **U**ranium-series **C**onstituents from the **E**nvironment (BRUCE). The intention of the tool is to provide a central data repository and to facilitate the calculation of radionuclide concentration ratios for bushfoods for use in member of the public ingestion dose assessments.

The BRUCE tool

The BRUCE tool is a spreadsheet-based tool for the storage and handling of data on natural-series radionuclide activity concentrations in bushfoods and environmental media from the ARR. Historical data accumulated by ***eriss*** and those reported by the Ranger mine operator have been retrieved from original source files, quality assessed and entered into the tool. Associated metadata such as spatial coordinates, dry-to-wet weight ratios and common names of bushfoods have also been entered. The tool currently contains more than 3500 individual records. Table 1 summarises the number of records currently available for aquatic and terrestrial bushfoods and for environmental media.

Table 1 Number of bushfood and environmental media records in the BRUCE tool

Biota/media	Number of records
Aquatic biota	
Fish	319
Mussel	475
Bird	28
Reptile (crocodile, file snake and turtle)	33
Plant	165
Terrestrial biota	
Mammal (bandicoot, buffalo, flying fox, pig and wallaby)	107
Reptile (goanna and snake)	11
Fruits	95
Root vegetables	26
Environmental media	
Soil	329
Water	1952
Sediment	109

A transfer query is available in the BRUCE tool to calculate radionuclide concentration ratios for bushfoods, including the ability to match bushfood and environmental media records on the basis of spatial coordinates and animal home range and on the basis of time of sample collection. The matching of records on the basis of spatial coordinates and animal home range assumes that the animal was collected at the centre of its range. The media samples that lie within the animal home range are then determined by:

$$\sqrt{[(x_1-x_0)^2 + (y_1-y_0)^2]} < \text{Range}$$

where:

x_0,y_0 are the spatial coordinates (easting, northing) of the point of collection of the animal;

x_1,y_1 are the spatial coordinates (easting, northing) of the point of collection of the media; and

Range is the linear home range of the animal.

The matching of bushfood and media records on the basis of time of sample collection is determined by:

$$t_0-a < t < t_0+b$$

where:

t_0 is the time of collection of the bushfood sample

a is a finite time period before the time of collection of the bushfood, which would typically represent the age or life span of the plant or animal; and

b is a finite time period after the time of collection of the bushfood sample to allow for the inclusion of media samples collected shortly after the bushfood.

While media samples are almost always collected at the same location and time as bushfoods, these samples only give an indication of the radionuclide concentrations in the media at a single point in both space and time. By including media samples collected within the home range, as well as those collected before and after the time of collection of the bushfood, a better estimate of the average radionuclide concentration that the plant or animal 'sees' over its life span can be made, meaning that the calculated radionuclide concentration ratio for the bushfood should be more representative.

The transfer query in the BRUCE tool also allows for an onsite/offsite weighting factor to be applied to the radionuclide activity concentration in media samples. This weighting becomes important in situations where the home range of an animal overlaps with the boundary of a mine site. Because of mining-related ground disturbance activities, the onsite media activity concentrations may be very much different to those found offsite in undisturbed environmental areas. Taking a straight average of the media activity concentrations may introduce bias into the concentration ratio calculation. The media weighted concentration ratio in the BRUCE tool is calculated as:

$$CR = [A_{bushfood}\ (d.w.) \times Ratio_{dry\text{-}to\text{-}wet}] / [(M_{onsite} \times F_{onsite} + M_{offsite} \times F_{offsite}) / (F_{onsite} + F_{offsite})]$$

where:

CR is the media weighted concentration ratio;

$A_{bushfood}$ (d.w.) is the dry weight radionuclide activity concentration measured in the bushfood;

$Ratio_{dry\text{-}to\text{-}wet}$ is the dry-to-wet mass ratio of the bushfood;

M_{onsite} is the mean radionuclide activity concentration in media samples collected onsite;

F_{onsite} is the fraction of the range area that is onsite;

$M_{offsite}$ is the mean radionuclide activity concentration in media samples collected offsite; and

$F_{offsite}$ is the fraction of the range area that is offsite.

Application to radiation protection of the environment

International trends in radiation protection indicate the need in certain circumstances to demonstrate that non-human species living in natural habitats are protected against deleterious radiation effects from practices releasing radionuclides to the environment. In particular, this has emerged as a best practice approach for nuclear fuel cycle activities, including uranium mining.

The 2007 Recommendations of the International Commission on Radiological Protection (ICRP 2007) distinguishes environmental protection objectives from human protection objectives. It also establishes a framework for assessing radiation exposures to non-human species from radionuclides released to the environment. Central to the framework is the use of reference organisms as conceptual and numerical models for estimating radiation dose rates to living organisms that are representative of an impacted environment.

The common method for estimating radionuclide transfer to non-human species, necessary for internal dosimetry calculations, is to use concentration ratio (Howard et al in press, IAEA in press). Concentration ratio in this context is the ratio of the average radionuclide activity concentration in the whole organism to that in the surrounding environmental media. This can differ from the concentration ratio for bushfoods, which is generally defined for a specific tissue component of the animal or plant.

The need to determine whole organism concentration ratios for a range of environment and species types has led to an increased data focus, nationally via the Australian Radiation Protection and Nuclear Safety Agency (ARPANSA) and internationally via the International Atomic Energy Agency (IAEA). In particular, ARPANSA has identified that there is a need to collect and assemble concentration ratio data for species typical of Australian environment types to facilitate more robust environmental assessments using existing tools (Doering 2010).

While the data in the BRUCE tool has not been specifically collected for assessing radiation protection of non-human species, there are some measurements of whole organism radionuclide activity concentrations from which concentration ratios can be derived, notably for freshwater mussels and some fish species. Additionally, published values of whole organism to tissue-specific concentration ratios for animals (Yankovich et al 2010) could be used to transform some of the data in the BRUCE tool to the format required for estimating radiation dose rates to biota using tools such as ERICA (Brown et al 2008) or ResRad-Biota.

The whole organism data for freshwater mussels and fish species from the ARR have been provided to Working Group 5 ('Wildlife Transfer Coefficient' Handbook) of the IAEA EMRAS II programme for inclusion in a new IAEA Technical Report Series document, *Handbook of parameter values for the prediction of radionuclide transfer to wildlife* (Howard et al in press), which will provide a summary of worldwide radionuclide transfer data for non-human species.

Future work

Future work with the BRUCE tool will include calculating bushfood-to-media concentration ratios for use in closure and post-rehabilitation dose assessments (ingestion pathway) for the Ranger mine. In this context, the concentration ratios will also be used with information from baseline studies (see KKN 2.2.5 Pre-mining radiological conditions at Ranger mine) to estimate the pre-mining or 'background' dose from bushfood ingestion, as it is only the above background component of the dose received that is amenable to dose limits.

Radiation protection of the environment is another area where work with the BRUCE tool is planned. This includes calculating organism-to-media concentration ratios for use in biota dose rate assessments for the ARR, contributing to the ARPANSA-coordinated project on 'concentration ratios in non-human biota inhabiting Australian uranium mining environments' and contributing to the environmentally themed working groups of the IAEA Modelling and Data for Radiological Impact Assessments (MODARIA) programme.

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Vegetation mapping of the Magela floodplain using high resolution satellite data

T Whiteside & RE Bartolo

Introduction

The significance of the wetlands of the Magela Creek floodplain and their biodiversity has been recognised through their listing by the Ramsar Convention on Wetlands. Since vegetation within the wetland is spatially and temporally variable, a robust methodology for mapping wetland vegetation at scales that can detect this variability is required. Mapping of the vegetation through time can assist with determining the driver(s) of this variability and identify if the change is naturally induced, or the result of human activities (eg burning, mining activities) and/or other stressors (eg feral animals, weeds). Multispectral high spatial resolution (pixels < 5 m) satellite imagery, such as WorldView-2, provides data of sufficient resolution for spatially and spectrally detailed analysis of such landscapes. However, the increased spatial heterogeneity associated with the finer resolution of the data requires data aggregation to assist with classification. The project described here uses GEographic Object-Based Image Analysis (GEOBIA) to classify the Magela floodplain vegetation.

GEOBIA involves the partitioning of remotely sensed images into meaningful image-objects. This involves assessing object characteristics through spatial, spectral and temporal scales. The process generates GIS-ready information to obtain new geo-intelligence (ie geospatial content in context, in this case floodplain communities). Object-based methods have been shown to cope better with the increase in spatial heterogeneity associated with high resolution multi-spectral satellite imagery, compared with traditional pixel-based (based on spectra) classification techniques. One of the main advantages of the GEOBIA approach is the ability to use contextual information to assist with image classification.

Apart from an unpublished map produced by ***eriss*** in 2004 there has been no updated Magela Creek floodplain vegetation community map produced for over 20 years.

Methods

The systematic remote sensing captures that were started in May 2010 continued through 2011 and 2012. Project work in 2011–2012 focused on producing a high resolution map of vegetation for the Magela Creek floodplain, downstream from the Ranger uranium mine, and processing of World-View 2 (WV2) image captures and associated ground surveys through that period.

The multispectral sensors onboard the WV2 satellite collect data in 8 spectral bands (Table 1) with a spatial resolution of 2 m. The raw data collected by the sensors were geometrically corrected to map projection and radiometrically calibrated to ground reflectance. The methods reported here focus on image classification since the image pre-processing methods used for the May 2010 World-View 2 imagery have been reported previously. However, the radiometric correction of the data for this project was undertaken using the FLAASH atmospheric correction model and not the Empirical Line (EL) method as previously undertaken. There were a number of reasons for using FLAASH over EL; chiefly no ground spectra were collected for northern third of the floodplain at the time and the image for this region was captured at a differing illumination and view angles from the other images and as such displayed different reflectance characteristics. In addition, due to the nature of commercial satellite imagery acquisition and the variability of early dry season weather, a major effort is required to collect field spectra necessary for EL that is costly and resource intensive.

Table 3 WorldView-2 Spectral band specifications

Spectral band	Wavelength centre (nm)	Wavelength min – max (nm)
Coastal	427	400–450
Blue	478	450–510
Green	546	510–580
Yellow	608	585–625
Red	659	630–690
Red Edge	724	705–745
Near Infrared 1	831	770–895
Near Infrared 2	908	860–1040

The first steps of the classification process created masks (in the form of objects) to extract the regions of non-image pixels and non-target land covers (eg the township and mine site) and eliminate these regions from the remaining processing. The next step involved extracting the floodplain from the surrounding terrestrial savannah landscape. To be able to successfully delineate the floodplain boundary using a semi-automated approach, a threshold segmentation set at 6 m and below was applied to the 30 m digital elevation model (DEM) derived from the Shuttle RADAR Topography Mission. An automated delineation would have not been possible based purely on spectral information due to similarities in the spectral response from some floodplain and non-floodplain vegetation communities (eg *Melaleuca* and *Eucalyptus* open woodlands). Once the floodplain boundary was delineated, objects for the open water class were extracted by segmenting the floodplain based on a threshold of the Near Infrared 1 band of the imagery (NIR1). Pixels within the floodplain with a NIR1 value less than 100 were assigned to open water objects. Similarly, cloud objects were extracted based on a ratio using the Near Infrared 2 (NIR2), Red Edge and Blue bands. Pixels below the threshold value of -850 for this ratio were assigned to the cloud class.

Objects representing floodplain vegetation community classes were created using a series of rules based upon a number of spectral indices (Table 2). A series of iterations involving segmentation, classification and reshaping were undertaken to create the class objects. The steps typically involved segmentation where objects were iteratively partitioned into smaller objects until an object threshold was met (based either on size or an index or ratio value). Class rules were applied using a step-wise method and adjacent objects of the same class merged to form larger objects. If objects were still spectrally variable (containing measurable heterogeneous pixel cover) they were considered to contain two or more classes and subsequently re-segmented with new rules applied to separate the classes. Otherwise spurious objects deemed too small (generally < 5 pixels), and completely enclosed by a larger object of a different class, were dissolved into (that is, deemed to be part of) the larger object. Processes were iterated until satisfactory class separability was achieved.

The thematic accuracy of version 1 of the floodplain vegetation map consisted of the validation or 'ground truthing' of the map against the two available reference data sets: an aerial survey collecting 100 reference sites across the floodplain, undertaken 29 May 2010, and an airboat survey collecting 28 reference sites, undertaken 17–20 May 2010. The validation involved site-specific analysis of the classes in the map and the classes determined at the locations within the reference data. This comparison was conducted using a confusion matrix. From the matrix, User's and Producer's accuracies were calculated for each class along with the overall classification accuracy.

Table 2 Spectral indices used within segmentation and classification algorithms

Index	Equation	What the index is sensitive to
NDVI2	NIR2-Red/NIR2+Red	The Normalised Difference Vegetation Index (NDVI) is strongly related to photosynthetic material. The index enabled discrimination between actively photosynthesising vegetation, senescent vegetation such as *Oryza,* and open water.
FDI2	NIR2-(Red Edge + Blue)	The Forest Discrimination Index (FDI) enables the separation of photosynthetic vegetation from bare soil and non-PS vegetation. In particular it separates woody canopy from understorey and ground cover.
NREB	NIR2+Red Edge-Blue	This Near infrared/Red edge/Blue index (NREB) distinguished non-PS vegetation that is highly reflective; in this case communities dominated by *Nelumbo*, *Leersia*, and *Salvinia.*
EVI*	G x (NIR2-Red) / (NIR2+(C1xRed)+(C2xBlue)+L)	The Enhanced Vegetation Index (EVI) is strongly correlated to evapotranspiration.

* G=2.5, C1=6, C2=7.5 and L=1

The Producer's accuracy was calculated by dividing the number of correctly identified sample units for a given class by the total number of sample units for a given class according to the reference data. The User's accuracy was calculated by dividing the number of correctly identified sample units for a given class by the total number of sample units for given class based on the classification. The overall classification accuracy is given by dividing the total number of correctly identified sample units for all classes by the total number of sample units for all classes.

Results

The final vegetation community map that was produced (Figure 1) consisted of 13 vegetation classes labelled based on the dominant *Genera* for the community: *Eleocharis*, *Hymenachne*, *Hymenachne*/Para Grass, *Leersia*, Mangrove, *Melaleuca* open forest, *Melaleuca* woodland, *Nelumbo*, *Oryza*, Para grass, *Pseudoraphis*, *Pseudoraphis/Hymenachne* and *Salvinia*. These class types are consistent with the classes identified and mapped previously in 1989 and 2004. Also displayed within the map are classes for open water, cloud and cloud shadow. Based on the reference data, the overall accuracy of the map was over 72%. User's and Producer's accuracies are displayed in Table 3.

The results indicate the vegetation classification process developed for this work easily distinguished between the spectrally and structurally distinct vegetation communities within the floodplain. The major source of confusion was between the *Eleocharis* and *Hymenachne* classes. The use of multiple indices and ratios were able to differentiate between classes that otherwise appeared spectrally (that is, optically) similar. Likewise, several contextual rules were able to be developed using structural (physical morphology features within the objects), differences to differentiate between spectrally similar classes. In addition, the high spatial resolution of the WV-2 imagery also enables the creation of maps of greater detail than past maps (Finlayson et al 1989) and smaller areas of vegetation can be readily identified and mapped (Figure 2).

Conclusions and future work

This project provides a good example of the application of the GEOBIA technique to mapping floodplain vegetation using high spatial resolution imagery. The rule-set implemented a number of well-known spectral indices and sensor band specific ratios to: (1) segment and classify major landscape units (MLUs) and mask out non-floodplain land covers, and (2) extract objects representative of the vegetation communities within the floodplain from the MLUs. The results indicate the rule set was able to distinguish the majority of floodplain classes. The availability of a digital elevation model with sufficient resolution and the use of contextual physical features, such as size and relationships to neighbouring objects, assisted in identifying and removing spurious objects and in facilitating the separation of objects of different cover type but with otherwise similar spectral characteristics.

Table 3 Producer's and User's accuracies for the classes of Magela Creek floodplain vegetation

Class name	Producer's accuracy	User's accuracy
Hymenachne	76.9%	57.1%
Melaleuca woodland	63.6%	58.3%
Eleocharis	46.9%	83.3%
Oryza	85.7%	77.4%
Para grass	80.0%	80.0%
Pseudoraphis	77.8%	87.5%
Pseudoraphis/*Hymenachne*	100%	56.5%
Salvinia	62.5%	83.3%
Melaleuca open forest	60.0%	60.0%
Nelumbo	80.0%	80.0%
Leersia	50.0%	100%
Hymenachne/ Para grass	75.0%	100%
Mangrove	100%	57.1%
Water	66.7%	100%
Cloud shadow	100%	100%
Cloud	100%	100%

This classification method will now be applied to mapping the floodplain vegetation using the images acquired in May 2011 and in June 2012. To facilitate this, the 2011 and 2012 imagery are now being geometrically registered to the 2010 imagery and radiometrically calibrated using the FLAASH model. The time series of maps will enable the annual variation between communities to be tracked, and facilitate identification of the key contributors to the changes that are occurring. It is anticipated the rule set will be transferable from one year to the next (with minor threshold adjustments associated with radiometric differences such as sun and view angle differences). Further work will involve adjustment of class rules to improve the thematic accuracy of the map.

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2.6.2 Off-site monitoring during and following rehabilitation

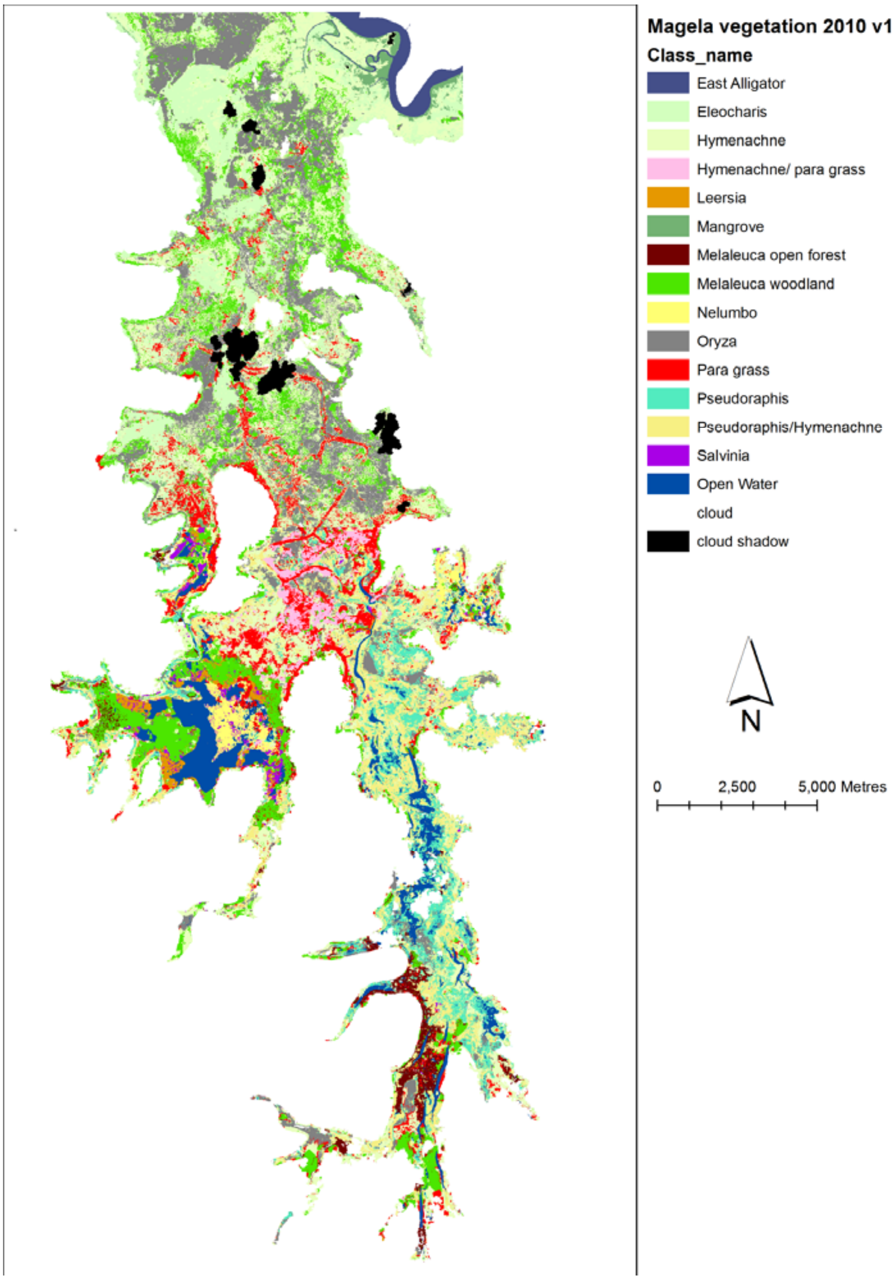

Figure 1 Version 1 of the 2010 Magela Creek floodplain map

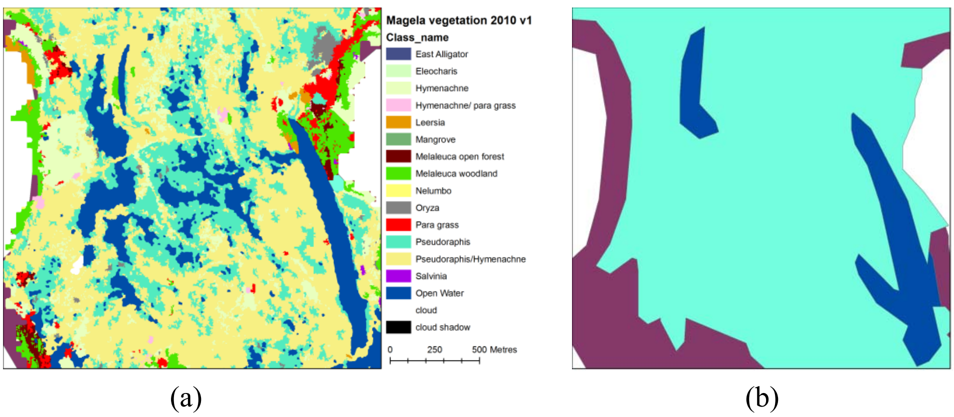

Figure 2 Sample area showing increased detail evident in 2010 vegetation map compared to previous published vegetation map (Finlayson et al 1989)

Part 3: Jabiluka

There are no research papers this year in the Jabiluka key knowledge needs theme.

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

Radiological assessment of the El Sherana containment

A Bollhöfer & C Doering

Introduction

The upper South Alligator River valley (SARV) in the south of Kakadu National Park is a popular tourist destination and a region of past uranium exploration, mining and milling activities. Mining in the area started soon after the discovery of the Coronation Hill deposit in 1953, and continued through to 1964. During that time, approximately 1000 tonnes of U_3O_8 were produced from 13 small scale uranium mines (Waggitt 2004) with an ore tonnage between 500 and 40,000 tonnes. When mining ceased, no substantial effort was made to clean up and rehabilitate the mine and mill areas or camps. Figure 1 shows a map of the general area with the location of several former mines marked.

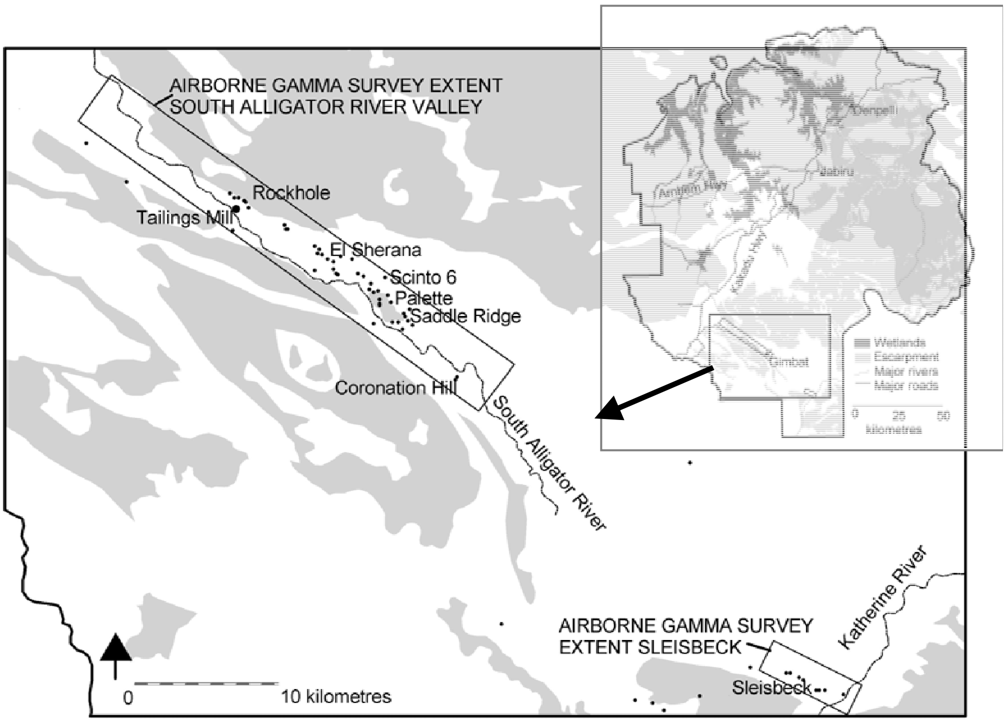

Figure 1 Alligator Rivers Region, with a detailed excerpt of the southern area showing the extent of two airborne gamma surveys conducted in 2000 and 2002 and the location of known uranium anomalies (from MODAT database, black dots). Some historic mining and milling areas are labelled (Supervising Scientist,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

In the mid 1980s radioactive tailings were discovered by Supervising Scientist Division (SSD) staff next to the road to the Gunlom waterfall, and close to the South Alligator River. In 1985-86 most off the tailings were removed by *Pacific Goldmines* to extract gold at the Moline mill, and during subsequent rehabilitation works in 1990–92 most of remainder of the tailings were removed. In 1999 some residual tailings were found on erodible terrain adjacent to the river (Tims et al 2000). These tailings 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.

The 1996 lease agreement between the Gunlom Aboriginal Land Trust and the Director of National Parks required the Director of National Parks to develop and fully implement a plan to restore mine-impacted areas in the SARV to near to natural environmental status by the end of 2015. Parks Australia was allocated $7.3 million over four years in the 2006–07 budget, specifically for this remediation programme (Director of National Parks 2006). SSD provided specialist support with regard to radiological assessment of the historic uranium mining and milling sites, including airborne (Pfitzner et al 2001) and ground-based (Bollhöfer et al 2007, Bollhöfer et al 2009) surveys, which helped delineate the extent and magnitude of radioactively contaminated areas. Works conducted by Parks Australia in 2009 included the bulk removal of mining wastes from various sites contaminated with naturally occurring radioactive material (NORM) and their placement into a near-surface disposal facility (the containment) constructed at the site of the disused El Sherana airstrip (Fawcett Mine Rehabilitation Services 2009). Engineering details of the containment are given in Doering & Bollhöfer (2012).

The Environmental Radioactivity program of SSD measured gamma dose rates and radon flux densities at the El Sherana containment prior to construction and again one year after closure (in 2010). The results of these surveys were published in an SSD internal report (Doering et al 2011). The paper presented here summarises the results of a follow up survey conducted in 2012.

Methods

A gamma survey was conducted at the El Sherana airstrip 3-4 September 2012 to measure the external gamma radiation. Environmental monitors equipped with compensated Geiger-Müller tubes were used. Measurements were made of the total counts per 100 s in air at a height of 1 m above the ground surface. These measurements were converted to absorbed dose rate using a calibration equation that related the count rate to the air kerma rate (absorbed dose rate in air) for the instruments. A total of 274 readings were taken and the locations were recorded using a handheld GPS.

Charcoal-loaded canisters were used for radon exhalation measurements. The general methods of sampling, radioactivity analysis and determination of radon exhalation flux density were similar to those described in Bollhöfer et al (2006). 43 canisters were deployed on 3 September 2012 and recovered two days later on 5 September 2012. Two additional canisters were carried into the field but remained sealed at all times. These canisters were controls, used to determine the background activity of the charcoal in the canisters.

Soil samples from 0-10 cm depth under each charcoal canister deployed on the containment were collected during the survey in September 2012. The radionuclide activity concentration in these samples will be analysed using the ***eriss*** HPGe gamma spectrometers.

Results

External gamma dose rates

Figure 2 shows the results from the 2012 gamma dose rate survey at the containment overlaid on a geo-referenced aerial photograph from 2007, showing the former El Sherana airstrip. The outer rectangle indicates the dimensions of the fenced area around the containment and the inner rectangle shows the approximate extent of the containment. It appears that dose rates measured north (upslope) of the containment, outside of the fenced area in a relatively undisturbed area, are slightly higher than those measured within, or south of, the fenced area.

Figure 3 shows the distribution of the external gamma dose rate data measured within the fenced area. The average gamma dose rate measured in 2012 was slightly higher at 0.15 ± 0.01 $\mu Gy \cdot hr^{-1}$ compared with 0.10 ± 0.01 $\mu Gy \cdot hr^{-1}$ in 2010 (Doering et al 2011). These values compare to pre-construction baseline values measured in 2009 of 0.12 ± 0.01 $\mu Gy \cdot hr^{-1}$. The measured values for each survey are low and are similar to typical background values between 0.1-0.15 $\mu Gy \cdot hr^{-1}$ measured elsewhere in the Alligator Rivers Region.

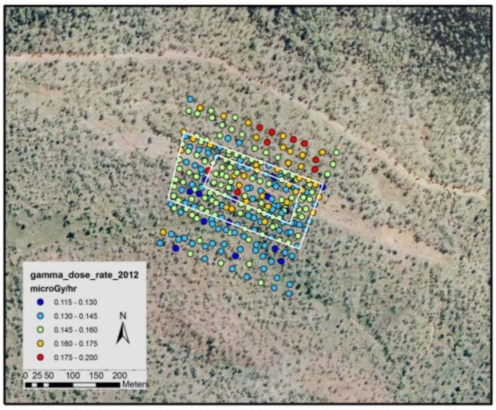

Figure 2 Gamma dose rates measured at the containment in 2012

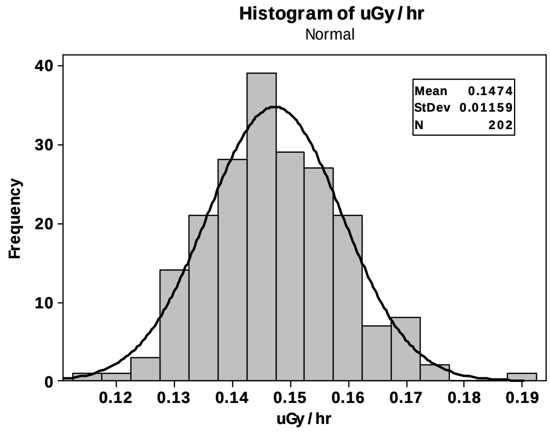

Figure 3 Distribution of gamma dose rates measured within the fenced area in 2012

Radon flux densities

Radon flux densities were determined at sites within and south of the fenced area. A total of 43 sites were investigated. Figure 4 shows the radon flux densities measured in 2012, and a comparison with results from 2010 (Doering et al 2011). Table 1 gives a statistical summary of post construction radon exhalation flux densities measured within the fenced area of the containment and the baseline radon exhalation (adapted from Doering & Bollhöfer 2012).

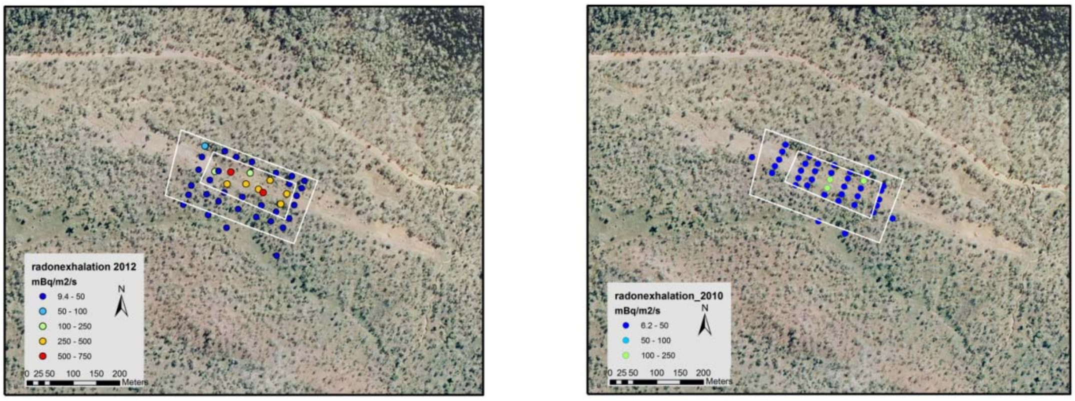

Figure 4 Radon flux densities measured at the containment in September 2012 and 2010

Table 1 Statistical summary of baseline and post-construction radon exhalation flux densities within the fenced area at the El Sherana containment (in mBq m^{-2} s^{-1})

Statistic	Baseline value	Post-construction (2010)	Post-construction (2012)
Arithmetic mean	14	27	124
Standard deviation	6	37	182
Median	12	15	33
Geometric mean	13	18	56
Minimum	5	6	18
Maximum	25	166	745
Count [n]	21	39	38

Erosion gullies on the containment

Erosion gullies visible in September 2012 at the south eastern end of the containment indicate that the containment is not yet geomorphologically stable. Gamma dose rate and radon flux density measurements conducted in, or in the vicinity of these erosion gullies (Figure 5) however, do not show higher values than those measured elsewhere on the containment. This shows that no radioactive material buried in the containment has been exposed.

Figure 5 Measurement of the gamma dose rate in an erosion gully at the containment

Discussion

The geometric mean radon flux density within the fenced area increased threefold between 2010 and 2012, from about 18 mBq·m^2·s^{-1} to 56 mBq·m^2·s^{-1} (Table 1). However, there has been little change in radon flux densities outside the fenced area at the control sites. The geometric mean measured in 2010 outside the fenced area was 16 $mBq \cdot m^2 \cdot s^{-1}$, with values ranging from 10–36 $mBq \cdot m^2 \cdot s^{-1}$. In 2012, the geometric mean outside the fenced area was 21 $mBq \cdot m^2 \cdot s^{-1}$, with a range of 9–30 $mBq \cdot m^2 \cdot s^{-1}$. It is thus unlikely that meteorological conditions or soil moisture are the cause of the observed increase in radon flux densities, as there has been no similar change at the control sites.

Other factors that may influence radon exhalation from the ground at the El Sherana containment include soil ^{226}Ra activity concentration, soil porosity and grain size. Although it is unlikely that the threefold increase in radon exhalation is due to higher soil ^{226}Ra activity concentration in the surface layer at the sites investigated (as indicated by the measured gamma dose rates) soil samples were collected during the latest survey and analysis for ^{226}Ra via gamma spectrometry is currently underway. This will allow SSD to determine the ratio of the ^{222}Rn flux density to the soil ^{226}Ra activity concentration and highlight areas where variability in ^{222}Rn flux is unrelated to changes in soil ^{226}Ra activity concentrations.

The low level radioactive material buried in the containment is covered by a compacted clay layer (0.5 m thick), which is covered by 2.5–3.5 m of soil. As the typical diffusion length of radon in soil is about 1.5 m (Porstendörfer 1994) radon originating from deeper layers usually decays within the soil profile before reaching the surface. It is thus unlikely that the radon exhaled at the surface of the containment originates from the buried waste. It is more likely the soil cover, a growth medium to facilitate re-vegetation of the area, is the source of the exhaled radon. This soil is a different substrate again to the two types of covers investigated at the Ranger trial landform (see ‘Radon exhalation from a rehabilitated landform’, this volume). Measurement of soil ^{226}Ra activity concentrations associated with the ^{222}Rn flux density measurement sites at the El Sherana containment will thus provide further data for the prediction of radon exhalation from a remediated landform.

When radon flux density measurements were performed in 2010, after construction of the containment, vegetation on the containment was not established (Doering et al 2011) and the surface of the containment was relatively compact. In September 2012 however, the vegetation cover of the area was denser (see Figure 5). The establishment of roots from vegetation growing on the containment may potentially create preferential radon transport pathways, allowing radon to more easily diffuse to the surface and thereby enhancing radon exhalation. Weathering of the substrate used to cover the containment may also be a cause of the increasing ^{222}Rn flux density, similar to what has been observed at the Ranger trial landform.

Further work

Ongoing environmental radioactivity monitoring of the surface radiological conditions at the containment and area adjacent to the site will allow continuing assessment of changes in radiological conditions, and determine when the system has reached a steady state with stable radiological conditions. It will also facilitate the assessment of radiological impact to members of the public and the environment.

SSD intends to deploy an ERDM (Environmental Radon Daughter Monitor) at the containment weather station in November 2012 to determine the potential alpha energy concentration (PAEC) in the air above the containment. This will help to determine potential radiation doses from the inhalation of radon decay products in the air at the site, and will allow a comparison to be made with typical background PAEC values measured elsewhere in the Alligator Rivers Region.

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1 Presentations to conferences and symposia that have been externally published in 2011–12 are included in section 'Conference papers – published'.

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Further information about SSD publications

SSD publications on the web
http://www.environment.gov.au/ssd/publications/index.html

SSD annual reports
http://www.environment.gov.au/ssd/publications/index.html#annual

Supervising Scientist Report series
http://www.environment.gov.au/ssd/publications/ssr/index.html

Consolidated list of publications 1978 – 30 June 2012
http://www.environment.gov.au/ssd/publications/ssd-bibliography.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:

- seven independent scientific members (including the Chair) nominated by the Federation of Australian Scientists and Technological Societies (FASTS) (now known as Science and Technology Australia) with expertise in the following disciplines:
  - Hydrogeology
  - Radiation protection and health physics
  - Plant ecology of minesite revegetation
  - Freshwater ecology
  - Ecotoxicology
  - Geomorphology
  - Ecological risk assessment
- seven members representing key stakeholder interests.

ARRTC functions

The primary functions of ARRTC are:

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

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

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

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

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

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

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

As a result of the extension in mine life and the conduct of a further three years of research since the original key knowledge needs (KKNs) and timeline priorities were established, it was judged by the Alligator Rivers Region Technical Committee (ARRTC) that a revision of the KKNs should be conducted in 2007–08. This was done and a revised list of KKNs approved by the committee. For comparison, both the original (2004–2006) KKNs and the new (2008–2010) KKNs were listed in Appendix 3 of the 2007–2008 ***eriss*** research summary (SSR200). The appendix in this volume contains the 2008–2010 KKNs. Although the 2008–2010 KKNs are still being reported against, given their date range they are currently under review again. The revision will be largely informed by an ecological risk assessment of the rehabilitation and closure phases of Ranger, to take place in 2013. It is expected that a revised list of KKNs will be finalised by the end of 2013.

Overall objective

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

Background

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

mining of uranium at Ranger is expected to cease in about 2012. This will be followed by milling until about 2020 and final rehabilitation expected to be completed by about 2026;

Nabarlek is decommissioned but has not reached a status where the NT Government will agree to issue a Revegetation Certificate to the mine operator. Assessment of the success of rehabilitation at Nabarlek is ongoing and may provide valuable data for consideration in the design and implementation of rehabilitation at Ranger;

Jabiluka will remain in a care and maintenance condition for some years. ERA, the project owner, has stated that further mining will not occur without the agreement of the traditional owners; and

grant of an exploration title at Koongarra is required under the terms of the Aboriginal Land Rights (Northern Territory) Act 1976 before the mining company can apply for a mining title. As such, any future activity at Koongarra is subject to the agreement of the traditional owners and the Northern Land Council.

This scenario is considered to be a reasonable basis on which to base plans for research and monitoring, but such plans may need to be amended if mining plans change in the future.

ARRTC will ensure the research and monitoring strategy is flexible enough to accommodate any new knowledge needs.

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

Specifically, under the Ranger Environmental Requirements (ERs):

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

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

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

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

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

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

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

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

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

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

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

1 Ranger – Current operations

1.1 Reassess existing threats

1.1.1 Surface water transport of radionuclides

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

1.1.2 Atmospheric transport of radionuclides

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

1.2 Ongoing operational issues

1.2.1 Ecological risks via the surface water pathway

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

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

1.2.2 Land irrigation

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

1.2.3 Wetland filters

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

1.2.4 Ecotoxicology

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

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

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

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

1.2.5 Mass balances and annual load limits

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

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

1.3 Monitoring

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

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

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

2 Ranger – Rehabilitation

2.1 Reference state and baseline data

2.1.1 Defining the reference state and baseline data

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

2.2 Landform

2.2.1 Landform design

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

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

2.2.2 Development and agreement of closure criteria from the landform perspective

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

2.2.3 Water quality in seepage and runoff from the final landform

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

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

2.2.4 Geomorphic behaviour and evolution of the landscape

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

2.2.5 Radiological characteristics of the final landform

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

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

2.3 Groundwater dispersion

2.3.1 Containment of tailings and other mine wastes

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

2.3.2 Geochemical characterisation of source terms

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

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

2.3.3 Aquifer characterisation and whole-of-site model

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

2.3.4 Hydrological/hydrogeochemical modelling

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

2.4 Water treatment

2.4.1 Active treatment technologies for specific mine waters

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

2.4.2 Passive treatment of waters from the rehabilitated landform

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

2.5 Ecosystem establishment

2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective

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

2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites

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

2.5.3 Establishment and sustainability of ecosystems on mine landform

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

2.5.4 Radiation exposure pathways associated with ecosystem re-establishment

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

2.6 Monitoring

2.6.1 Monitoring of the rehabilitated landform

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

2.6.2 Off-site monitoring during and following rehabilitation

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

2.7 Risk assessment

2.7.1 Ecological risk assessments of the rehabilitation and post rehabilitation phases

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

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

2.8 Stewardship

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

3 Jabiluka

3.1 Monitoring

3.1.1 Monitoring during the care and maintenance phase

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

3.2 Research

3.2.1 Research required prior to any development

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

4 Nabarlek

4.1 Success of revegetation

4.1.1 Revegetation assessment

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

4.1.2 Development of revegetation monitoring method

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

4.2 Assessment of radiological, chemical and geomorphic success of rehabilitation

4.2.1 Overall assessment of rehabilitation success at Nabarlek

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

5 General Alligator Rivers Region

5.1 Landscape scale analysis of impact

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

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

5.2 South Alligator River valley rehabilitation

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

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

5.3 Develop monitoring program related to West Arnhem Land exploration activities

5.3.1 Baseline studies for biological assessment in West Arnhem Land

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

5.4 Koongarra

5.4.1 Baseline monitoring program for Koongarra

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