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eriss research summary 2012–2013

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2.5 ECOSYSTEM ESTABLISHMENT

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

KKN 2.5.4 Radiation exposure pathways associated with ecosystem re-establishment

2.6 MONITORING

KKN 2.6.1 Monitoring of the rehabilitated landform

Preface

The Environmental Research Institute of the Supervising Scientist (***eriss***) is part of the Supervising Scientist Division (SSD) of the Australian Government's Department of the Environment (DoE). ***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 through 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 2012–13 financial year (1.7.12 to 30.6.13). The report is structured according to the five major topic areas in the KKN framework, noting that this year there are no papers for Nabarlek.

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

Of the 30 or active projects during 2012–13, the majority (>95%) were addressing issues associated with the current operational phase and/or proposed rehabilitation and post-rehabilitation phases of Ranger mine.

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. Research on the toxicity of manganese to local freshwater species was completed, with the results being used to derive a site-specific trigger value (to protect at least 99% of species) for manganese of 75 μg/L. Also, an interim site-specific trigger value (to protect at least 99% of species) for ammonia, of 0.7 mg/L as total ammonia nitrogen, was derived based on the latest international scientific knowledge. Both trigger values are expected to be incorporated into the regulatory framework during 2013–14, while the ammonia value will also be updated in 2015 following the generation of toxicity data for local freshwater species.

Research related to rehabilitation at Ranger remains a key focus at ***eriss***. The trial landform studies continued in 2012–13, with a focus on the processing and reporting of solutes data (up to 2011–12) and continued updating of hydrology and bedload yields. Bedload yields from the study plots continued to decline with time. Similarly, for the solutes, major ion and uranium concentrations have also declined over time. Other key rehabilitation-related research to be progressed include: (i) further refinement and testing of the landform evolution models that are being used to assess the stability of the rehabilitated landform over both short and long-term timeframes; (ii) continued development of water quality closure criteria for solutes and turbidity/suspended sediment in billabongs; (iii) at a broader spatial scale, mapping of vegetation and associated annual variation on the Magela floodplain (from 2010 onwards) using high resolution multispectral satellite data, to inform a baseline from which any landscape scale impacts of mine rehabilitation could be detected; and (iv) commencement of the Ranger rehabilitation and closure ecological risk assessment, in collaboration with ERA.

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. In 2012–13, these data were acting as a baseline prior to the decommissioning and rehabilitation of the interim water management pond at the Jabiluka site during the 2013 dry season. Radiological monitoring and assessment continued at the El Sherana airstrip radiological containment in the South Alligator River Valley, with the primary purpose to assess the containment's performance through time, including whether site radiological conditions are stable. 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.

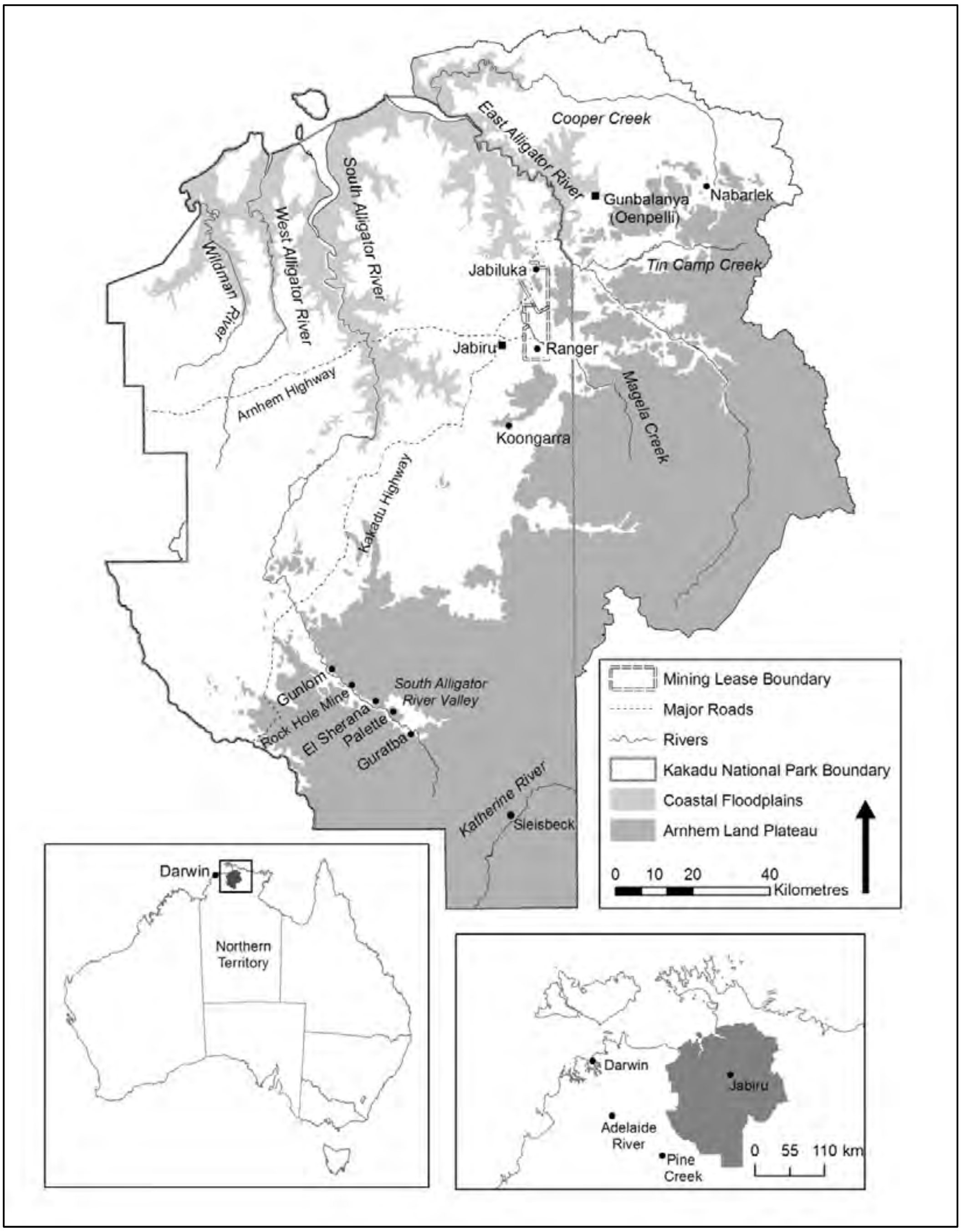
The key non-uranium mining related external activity for 2012–13 was the involvement of several ***eriss*** staff in the current revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Details of the involvement in the Water Quality Guidelines revision are provided in the Supervising Scientist 2012–13 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.

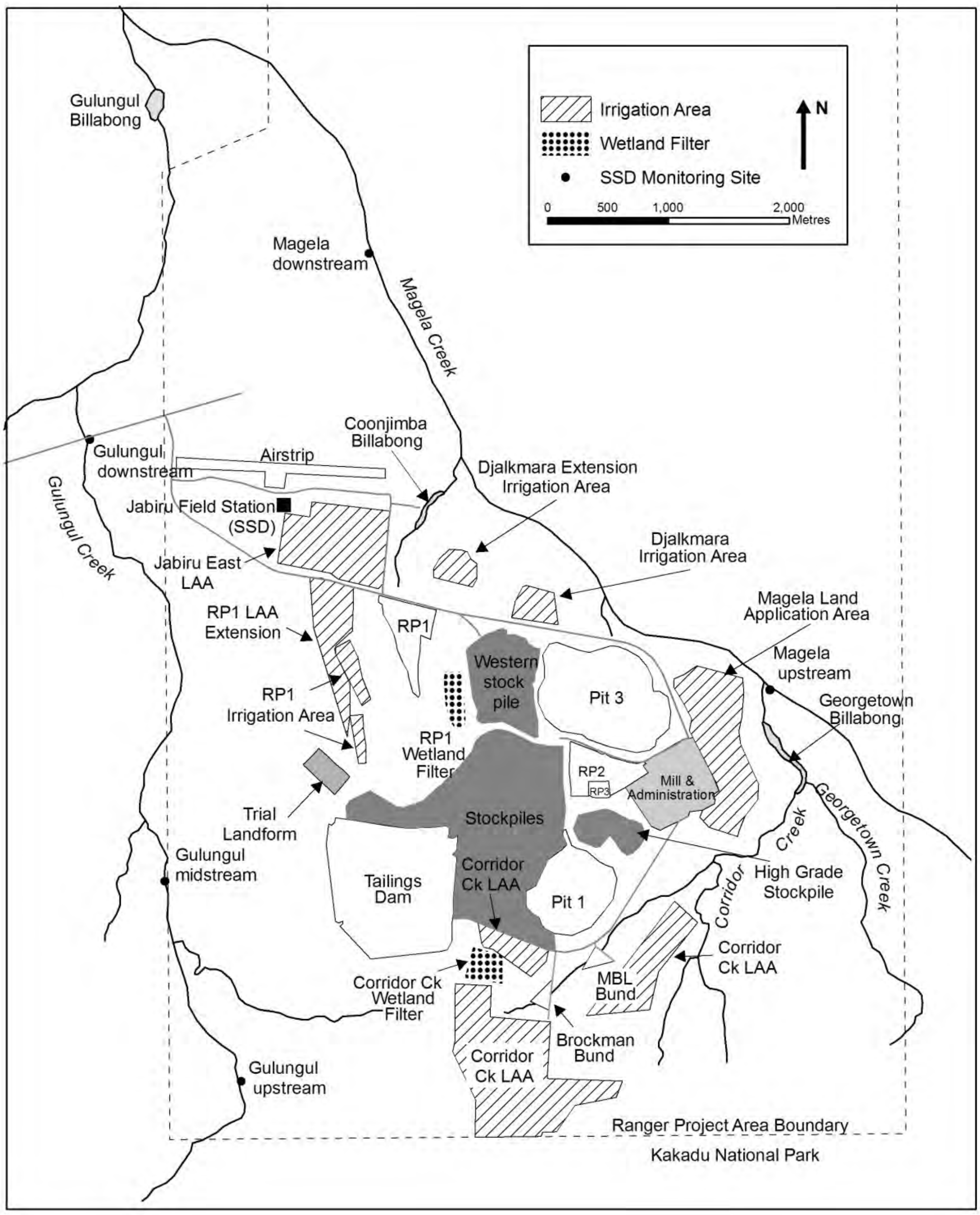
For additional information, readers are referred to the annual publications list (Appendix 1) that details all of the material published, and conference and workshop papers presented, by ***eriss*** staff in 2012–13.

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

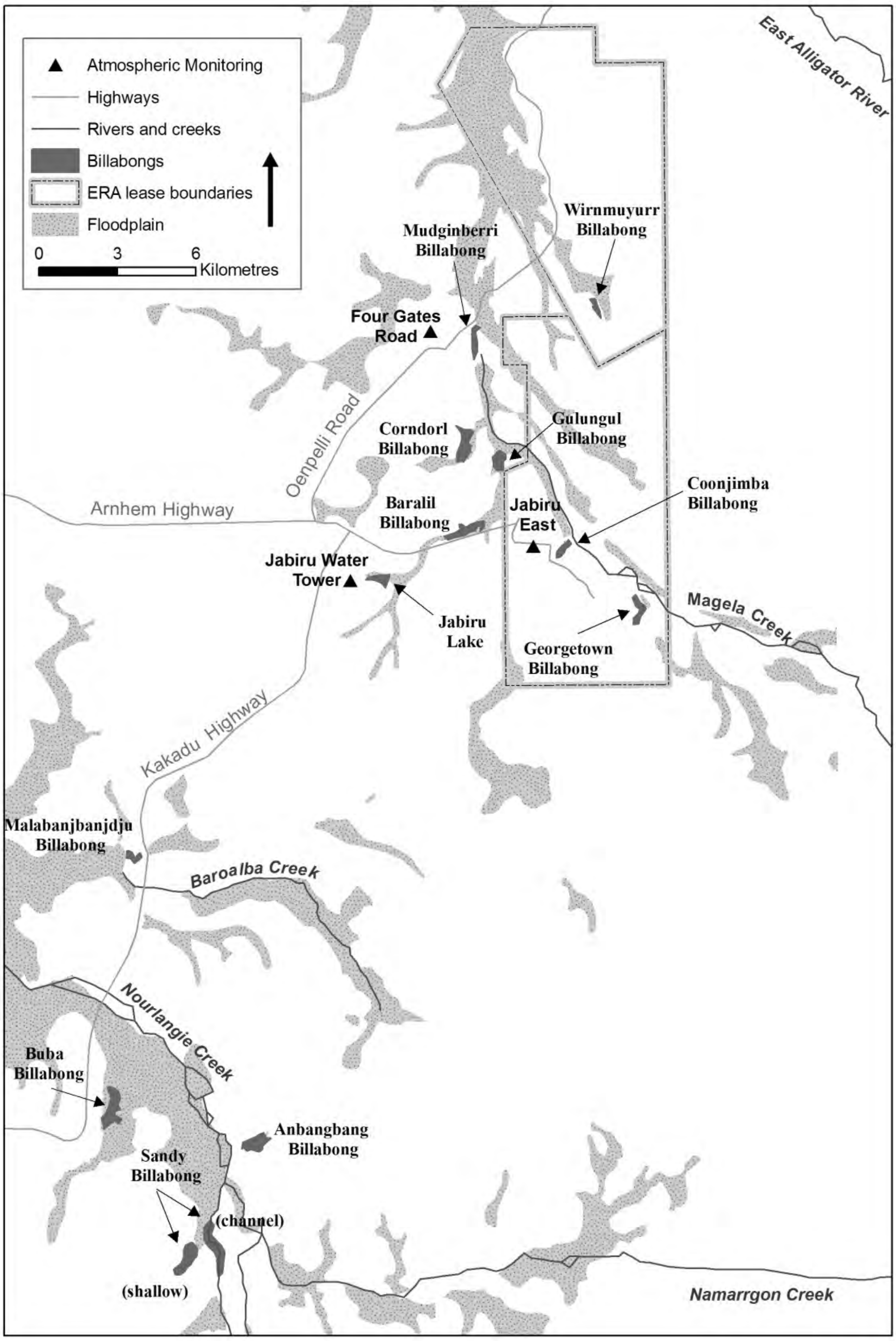
Note: Authors were Supervising Scientist staff at the time of research and/or write-up unless otherwise stated.



Map 1 Alligator Rivers Region



Map 2 Ranger minesite



Map 3 Sampling locations used in SSD's research and monitoring programmes

Part 1: Ranger – current operations

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

AJ Harford, SL Simpson, AA Chariton, RA van Dam & CL Humphrey

Background

There are no reliable toxicity trigger values (TVs) for uranium (U) in sediment. This is a significant issue for both the operational (sediment quality management triggers) and closure (sediment quality closure criteria) aspects of the environmental management of U mines. In the local context, quality sediment U toxicity data are specifically required to determine if any observed differences in populations of benthic biota in billabongs adjacent to the Ranger Uranium Mine are due to U in sediments or to other mining or non-mining related factors. For mine closure, sediment quality criteria are needed for downstream receptor wetlands, as well as for any on-site, re-instated waterbodies, or (temporary) sentinel wetlands which will serve to capture and 'polish' seepage and runoff waters from the rehabilitated mine site.

A project that aims to derive a sediment TV for U commenced in 2009. Following an initial site characterisation during the 2008–09 wet season, two pilot studies were conducted during the 2009–10 and 2010–11 wet seasons (see van Dam et al. 2010, Harford et al. 2011, Harford et al. 2012 for detailed methods and results). Briefly, sediments spiked with U were deployed in an undisturbed billabong (Gulungul) for the duration of the wet season. They were retrieved and sub-sampled for the analysis of bacteria (prokaryotes), microalgae, microinvertebrates and macroinvertebrates (eukaryotes) using a combination of ecogenomic and traditional taxonomic methods. The first pilot study used a sediment spiking method that involved extensive processing, ie sieving to <2 mm followed by U spiking and ~3 months of mixing and equilibration. This process demonstrated that U bound to the sediments quickly and entirely. However, at the end of the wet season the treatment sediments were notably much more compacted than the natural sediments, an artefact created by initial pre-deployment processing.

A different spiking method that involved minimal sediment manipulation was used for a second pilot study with the primary aim of producing softer sediment. This method removed the <2 mm sieving step, and involved pouring U stock solutions twice through the sediment and mixing the sediments 24-h later, after the solution had drained. This method produced sediments with similar proportions of Total Recoverable Metal (TRM) to weak-Acid Extractable Metal (AEM) U compared to the longer, more extensive processing method used in pilot 1. Moreover, compared to pilot 1, these sediments were softer and were considered to be more conducive to colonisation by the resident billabong biota.

The pilot 2 results showed that macroinvertebrate taxa richness was significantly lower in the highest (2000 mg kg^{-1}) treatment (ANOVA, P = 0.042), although multivariate analyses of the community structure data (PERMANOVA, ANOSIM, CAP and DistLM) showed no differences between the treatments and no significant correlations between the abiotic and biotic data. Microinvertebrate communities were not different at

500 mg kg^{-1} U (PERMANOVA, $P = 0.12$) compared to controls but were significantly different at 1000 and 2000 mg kg^{-1} U (PERMANOVA, $P = 0.01$). 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. No reliable results have yet been acquired from the bacteria dataset because the original sequencing of the samples appears to be inaccurate. The analysis of the community assemblage datasets using novel statistical techniques (TITAN, Gradient Forests, NCAP) found thresholds of community changes and also identified the key species correlated with the changes. The main experiment, focusing on macroinvertebrate and microinvertebrate communities, commenced at the beginning of the 2013–14 wet season, with the experimental design revised on the basis of the findings of the two pilot studies.

Methods

The full-concentration range experiment employed a reduced number of replicates (4) per treatment in order to increase, and thereby optimise, the number of U treatments (12), enabling better characterisation of the concentration-response relationship. The concentration range chosen for the final experiment was: control (~5 mg kg-1 U), 50, 100, 200, 400, 600, 800, 1000, 1500, 2000, 3000 and 4000 mg kg-1 (nominal U). The passive absorption method of U spiking described for pilot 2 above was used to spike 2 kg x 4 replicates of each treatment. Deployment, retrieval and processing were the same as described for pilot 2 (Harford et al. 2012).

The sediments were spiked in early December and were deployed in Gulungul Billabong on 29 December 2012. They were retrieved from the billabong on 14 May 2013 with redox measured immediately upon removal. The replicate containers were then each placed in a bucket of billabong water for transport to the Jabiru Field Station where, for each replicate, pH was measured and sub-samples taken for 1) TRM, AEM and porewater metal analyses; 2) DNA sequencing of eukaryotes; and 3) metagenomic analysis of microbial genes. Porewaters were isolated by centrifugation and filtration at 0.45 μm, and samples for DNA analyses were snap-frozen in liquid nitrogen and remained at -196°C during transport. The remaining sediment was sieved and the >500 μm fraction placed in ethanol for macroinvertebrate analysis. Metal analyses by ICP-MS and environmental DNA analyses were conducted by CSIRO (Land and Water, Lucas Heights). Chemical analyses of the sediments included Total Recoverable Metals (TRM), weak-Acid Extractable (AEM) and porewaters, each for a suite of 19 metalloids and major ions. TOC analyses were conducted by ALS laboratories (Brisbane).

The analysis of bacterial community assemblages will not occur at this stage because of uncertainties surrounding the ecological significance of a change in bacterial community structure. Sediment bacteria are factors capable of modifying sediment biogeochemistry and also U speciation and toxicity. 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 biogeochemistry. Therefore, sub-samples of the spiked sediments have been archived and will be used in a metagenome study. Specifically, the abundance of all functional gene sequences will be quantified in order to determine the function of the bacteria in the sediments. This study will be undertaken by a PhD student at the beginning of 2014 and will be a collaboration with CSIRO and Macquarie University.

An extra 2 kg of each treatment prepared for the final experiment was also sent to CSIRO (Land and Water, Adelaide) in order to conduct a temperate chironomid (larval midge) sediment toxicity test. Forty *Chironomus tepperi* larvae were exposed to each U sediment treatment for 10 days. Following the exposure, survival, growth (length, µm) and emergence were measured. Overlying waters were collected and sent to CSIRO (Land and Water, Lucas Heights) for metal analyses.

Results – progress to date

Field experiment

Results of the chemical analyses indicated that the pre-deployed samples had lower than expected concentrations of U, including 69% of the nominal U concentration in the highest 4000 mg kg^{-1} treatment (Table 1). However, the U analysis of post-deployment samples indicated that U concentrations were mostly ±20% of the nominal concentrations (Table 1). (Possible causes of this pre- and post-deployment discrepancy in sediment U concentration are being investigated.) At the beginning of the experiment there were strong positive correlations with TRM U and porewater concentrations of some metals, most notably Al and Mn (data not shown). This could be expected because of the displacement of these metals by U during the spiking process. There was also a correlation of TRM U with porewater U concentrations, with 6275 µg L^{-1} U being present in the highest treatment (Table 1).

Table 1 Uranium analysis of the spiked sediments at the beginning (pre-deployment) and end (post-deployment) of the experiment.

	Pre-deployment			Post-deployment			
Nominal U	TRM [a] U (mg kg^{-1}, mean ±se)	Porewater U (µg L^{-1})	% of nominal	TRM U (mg kg^{-1}, mean ±se)	Porewater U (µg L^{-1})	% of nominal	AEM[b]/TRM ratio
6	6 ± 0	7 ± 1	104	8 ± 1	3 ± 0	138	0.73
50	47 ± 2	16 ± 2	93	64 ± 5	23 ± 0	128	0.78
100	85 ± 4	18 ± 3	85	115 ± 4	48 ± 3	115	0.82
200	180 ± 4	21 ± 1	90	225 ± 11	153 ± 66	113	0.84
400	354 ± 4	25 ± 2	88	386 ± 28	160 ± 43	97	0.83
600	458 ± 14	47 ± 6	76	658 ± 48	314 ± 26	110	0.84
800	641 ± 35	52 ± 11	80	906 ± 41	363 ± 78	113	0.82
1000	780 ± 21	157 ± 55	78	1148 ± 45	507 ± 35	115	0.84
1500	1303 ± 43	458 ± 97	87	1593 ± 55	533 ± 80	106	0.87
2000	1545 ± 23	525 ± 15	77	2010 ± 165	564 ± 86	101	0.89
3000	2300 ± 88	1975 ± 266	77	2880 ± 184	1009 ± 233	96	0.90
4000	2773 ± 121	6275 ± 433	69	3193 ± 320	863 ± 112	80	0.93

[a] TRM = Total Recoverable metals, b AEM = weak-Acid Extractable Metals

This suggests that the washing procedure in the spiking method may have been incomplete in the higher treatments or that weakly bound metal was detaching from the sediments. As expected, porewater concentrations of U in the post-deployment samples were significantly reduced. All other metals in the porewaters of post-deployment samples were at concentrations similar to the controls (data not shown). Ultimately, a suitable gradient of U contamination was measured in the treatments and porewater metals were likely to be at acceptable levels for the majority of the experiment, although it is unknown how quickly they returned to control levels.

Laboratory toxicity test

Exposure of *C. tepperi* larvae to the U spiked sediments resulted in a reduction in growth and emergence (Figure 1). Emergence was the more sensitive of the two endpoints, but there were strong concentration-response relationships for both. The concentrations causing a 10% effect (i.e. EC10s) were 400 and 210 mg kg^{-1} for growth and emergence, respectively. However, measurement of metals in the overlying water (Table 2) showed that there was an increase in waterborne concentrations of U that was correlated with sediment bound concentrations (Table 1). There was also an increasing gradient in waterborne concentrations of Mn, which again, was seen in the sediment porewater analyses (Table 1). Aluminium and Zn also showed elevated concentrations in the overlying water but the increases were not related to increases in U (Table 2). Consequently, the observed effects may have been due to waterborne U exposure, porewater U exposure and/or the sediment U exposure. It is also possible that Mn contributed to the observed toxicity but the toxicity of Mn to this species is unknown.

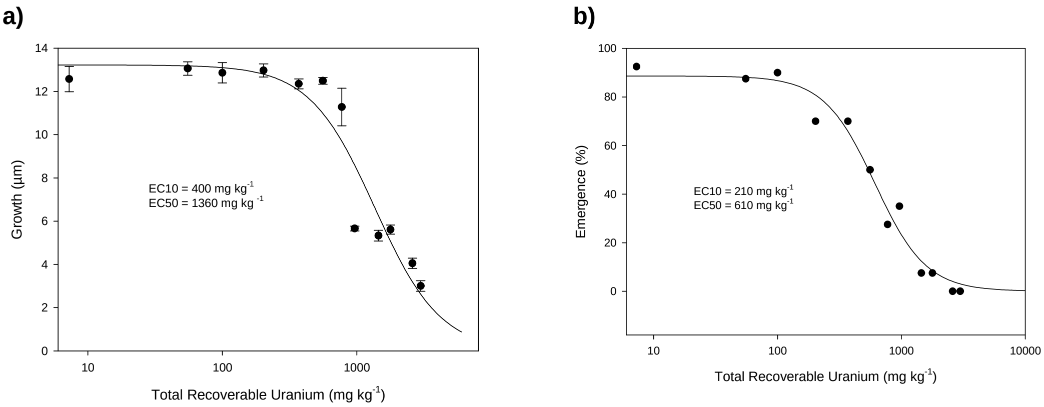


Figure 1 Effect of U spiked sediments on the a) growth and b) emergence of *Chironomus tepperi.* For growth, data points represent the mean ± se of 40 larvae. Emergence is pooled data from 40 midge larvae

Table 2 Metal concentrations in the overlying water following the *Chironomus tepperi* toxicity test

Nominal Uranium (µg L^{-1})	Measured concentration (µg L^{-1})			
	Uranium	**Aluminium**	**Manganese**	**Zinc**
0	1	454	31	11
50	6	356	28	3
100	9	227	26	2
200	16	211	34	3
400	32	201	27	2
600	38	129	63	8
800	37	106	53	8
1000	53	94	82	3
1500	71	102	169	12
2000	64	56	114	6
3000	148	80	223	8
4000	570	312	472	19

Future work for completion

The eukaryote environmental DNA dataset is currently undergoing the final steps in bioinfomatic processing and is yet to be analysed. Due to the low richness and abundance of macroinvertebrates found in pilot 1 and 2, the 500 µm sieved macroinvertebrate samples have been archived. The eukaryote dataset will be analysed using a number of multivariate and univariate statistical analysis in order to determine a sediment quality trigger value for U.

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Ecotoxicological assessment of Manganese

AJ Harford, K Cheng, AC Hogan, M Trenfield & RA van Dam

Background

Manganese (Mn) was recognised as a contaminant of potential ecotoxicological concern at Ranger in the early 2000s following observations of increasing concentrations in shallow groundwater adjacent to Magela Creek (MC20; up to 50 000 µg L^{-1}, ERA 2008) and spikes in Coonjimba Billabong and Corridor Creek (800-1600 µg L^{-1}; previously reported in Harford et al. 2009). The likelihood of Mn in waters being released from Ranger may increase with the commissioning of the brine concentrator plant in mid-2013. A pilot-scale brine concentrator plant trialled in 2011 produced distillate waters containing Mn at concentrations ranging from 130-240 µg L^{-1} (Harford et al. 2012). These Mn concentrations are higher than those currently measured in RP1. Consequently the addition of distillate to on site water bodies may eventually result in Mn concentrations in Magela Creek higher than have previously been measured.

The current site-specific guideline for Mn in Magela Creek downstream of Ranger is 26 µg L^{-1} (Iles 2004). This value was derived from statistical analysis of water quality data from the upstream reference site, and is applicable only when flow in Magela Creek is greater than 5 cumecs. These periods are considered atypical as a whole given the increased contributions from shallow groundwater at these times. Less than 2% of measured concentrations of Mn downstream of Ranger have exceeded the existing guideline, with the majority of exceedances occuring during early wet season flows or end of wet season recessional flows, often when flow was less than 5 cumecs (i.e. when the guideline is not applicable).

Historically, the acute and chronic toxicity of Mn to freshwater biota has been considered to be low (i.e. 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, more recently, a review of the Mn toxicity data 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 & van Dam 2012) indicated that the green hydra, *Hydra viridissima*, is more sensitive to Mn than any other species reported in the literature. No-observed-effect concentrations and 10% inhibition concentrations were reported between 20 and 180 µg L^{-1}, which are well below the historical Mn spikes reported above, and within the range of concentrations of residual Mn measured in process water distillate (Harford et al. 2013). These toxicity data, plus the potential for higher Mn concentrations in Ranger discharges, highlighted the need for a more comprehensive assessment of Mn toxicity.

Currently, insufficient Mn toxicity data exist for local species in Magela Creek water 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 Magela Creek water is predicted to result in higher than expected Mn toxicity based on existing literature. (Peters et al. 2010, 2011).

Methods

The toxicity of Mn was assessed using six Australian tropical freshwater species: the unicellular green alga (*Chlorella* sp.); the duckweed (*Lemna aequinoctialis*); the green hydra (*H. viridissima*); the cladoceran (*Moinodaphnia macleayi*); the aquatic snail (*Amerianna cumingi*) and the Northern trout gudgeon (*Mogurnda mogurnda*)(Riethmuller et al. 2003). The diluent, Magela Creek Water (MCW), was spiked with Mn using a stock solution of manganese sulfate. Dissolved (0.1 μm filtered) concentrations were checked for accuracy before and after the test exposure through ICP-MS analysis.

At least two standard valid toxicity tests were completed for each species and, for some of the toxicity tests, a modified design was used (Table 1). Specifically, the number of concentrations tested was increased by reducing treatment replication. The design has the advantage of being able to better characterise the concentration-response relationship and derive toxicity estimates with increased accuracy. Due to logistical reasons, the modified design was not used for the snail toxicity tests, while the method was abandoned for the *M. macleayi* tests because it was more susceptible to failing QC criteria.

Table 1 Details of the Mn concentration-response tests conducted

Test ID	Date	Species name	Endpoint	Mn concentration range tested (μg L^{-1}) [a]	Comments
1278G	30/04/12	*Chlorella sp.*	Population growth	4.0 – 480000	Modified design [b]
1294G	28/08/12			3.0– 135000	
1276L	30/04/12	*L. aequinoctialis*	Surface area growth rate	3.0 – 44000	Modified design [b]
1279L	23/04/12			2.0 – 19000	
1297L	10/09/12			0.3 – 39000	
1277B	30/05/12	*H. viridissima*	Population growth	0.3 – 840	Modified design [b]
1290B	30/07/12			0.6 – 755	
1310B	19/11/12			1.8 – 1950	
1299D	14/09/12	*M. macleayi*	Reproduction	3.0 – 1150	Modified design for first test only [b]
1345D	1/08/13			2.0 – 4700	
1275S	23/04/12	*A. cumingi*	Reproduction	1.8 – 33500	As per protocol
1307S	29/10/12			2.0 – 10500	
1335S	29/04/13			2.0 – 29500	
1284E	14/06/12	*M. mogurnda*	Survival	2.0 – 46500	Modified design [b]
1293E	23/08/12			4.0 – 295000	
1300E	20/09/12			4.0 - 360000	

[a] Concentration range is based on the mean of start and end Mn values

[b] A modified design of less replicates and more treatments was used.

Due to observed losses of Mn in the *H. viridissima* toxicity tests, the fate of Mn in the hydra test system was, subsequently, comprehensively characterised using three Mn concentrations in MCW (background, 250 and 600 μg L^{-1}). An additional treatment was included for each Mn concentration, whereby the test petri dishes were pre-inoculated with a solution of 250 μg L^{-1} Mn for 24 h prior to the test commencement, i.e. "primed". 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 individual toxicity test response data were expressed as a percentage of their control response, which normalises the result and enables pooling of data from multiple tests. Non-linear regression (3-parameter logistic) analyses were used to determine point estimates of Inhibitory Concentrations (ICs) that reduced endpoint responses by 10% and 50% (i.e. IC10 and IC50) relative to the control responses. The *M. mogurnda* test represents an acute exposure and measures lethality, hence a more conservative 5% effect/lethal concentration was estimated instead of a 10% effect/lethal concentration.

The site-specific Mn TV was derived for the 99% species protection level, from a Species Sensitivity Distribution (SSD) using the Burrlioz 2.0 software. The dataset used to derive the TV included the six local species tested, as well as three non-local species for which Mn toxicity had been assessed in another study under physico-chemical conditions closely aligned with those of Magela Creek. The additional three species were included in order to improve the fit of the distribution and, consequently, the reliability of the TV. Specifically, toxicity estimates from the temperate northern hemisphere species; *Pseudokirchneriella subcapitata* (algae), *Ceriodaphnia dubia* (clodercan) and *Pimephales promelas* (fish) were added to the SSD. These toxicity tests were conducted at 25°C in softwater (Hardness = 12 mg L^{-1} $CaCO_3$, Ca = 4 mg L^{-1}) with a pH of 6.7. The Dissolved Organic Carbon (DOC) was 12 mg L^{-1}, which is 4 times higher than MCW (typically <3 mg L^{-1}). However, DOC has been reported to have less of an influence on Mn toxicity compared to other phyico-chemical parameters (Markich et al. 2000; Peters et al. 2011).

Results

Manganese toxicity varied markedly between the six local tropical freshwater species (Figure 1 and Table 2). Toxicity to *M. mogurnda*, *L. aequinoctialis*, and *Chlorella* sp. was very low, with IC10 values all above 1000 μg L^{-1} (Table 2). The aquatic snail, *A. cumingi*, the cladoceran, *M. macleayi*, and the hydra, *H. viridissima* were markedly more sensitive, with IC10 values lower than 1000 μg L^{-1}. The hydra was the most sensitive species tested, with an IC10 of 140 μg L^{-1} (Table 2). Typically, Mn no/low effect toxicity estimates (e.g. EC/IC10s, no-observed-effect-concentrations) for freshwater species are > 1000 μg L^{-1} (Peters et al. 2010). The order of sensitivity of the six species to Mn was:

> *A. cumingi* > *H. viridissima* >*M. macleayi* >> *L. aequinoctialis* > *Chlorella* sp. >> *M. mogurnda*

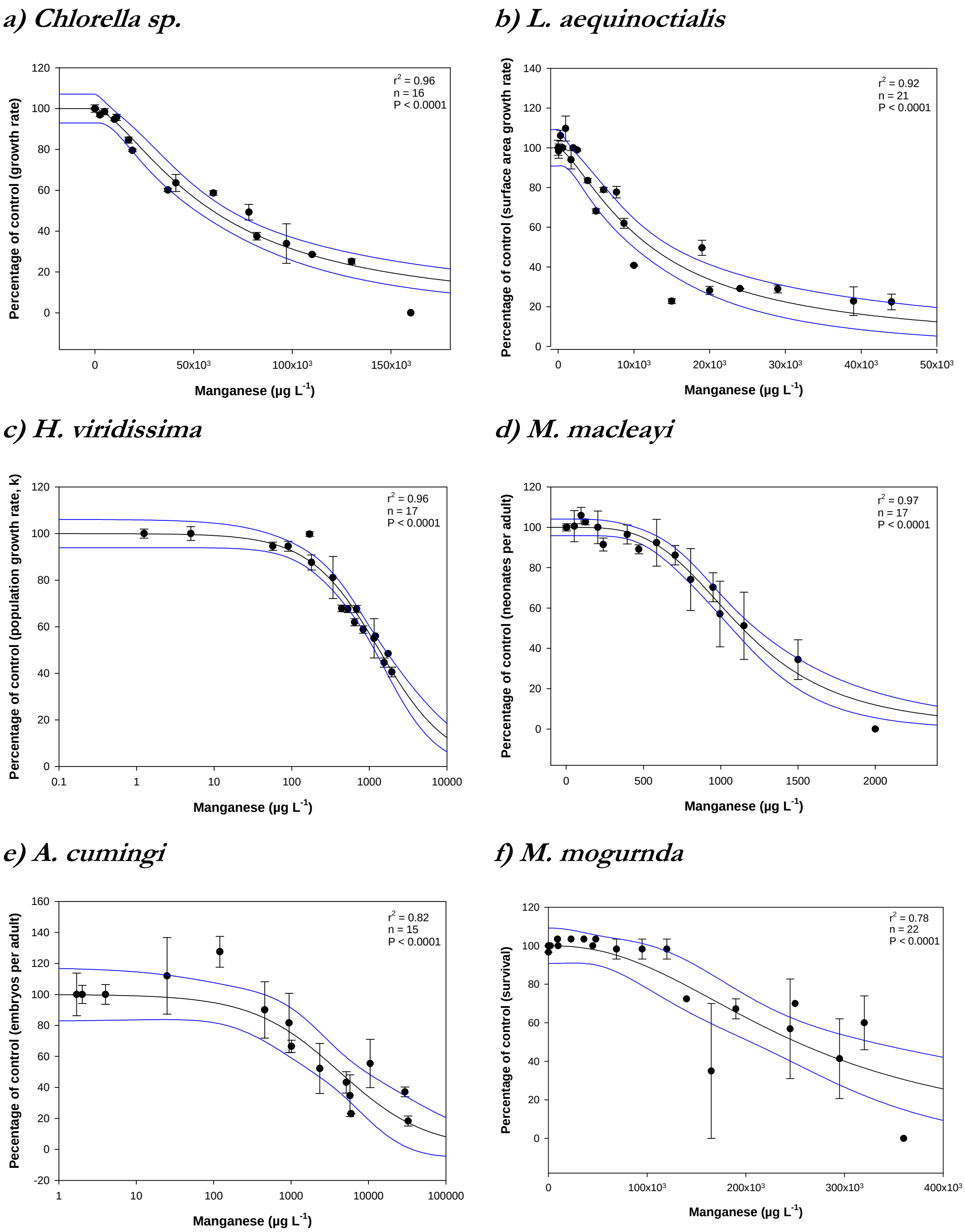

Figure 1 Manganese concentration-response relationships for the six tested species. Data points represent the mean ± standard error of 2–3 replicates, except for *M. macleayi*, where *n* = 5–10 replicates. 3-parameter logistic models were used to determine toxicity estimates for all species

A noteworthy loss of Mn was observed in two of four *H. viridissima* toxicity tests. Due to the initial chemistry sampling design, the Mn loss of ~250 µg L-1 was measured in only half of the treatments in the first *H. viridissima* toxicity test (1277B). Hence, because Mn was not measured in all treatments this test was omitted from the derivation of the toxicity estimate. A similar Mn loss was seen in one other *H. viridissima* toxicity test (1290B). For this test, the concentration of Mn was measured in all treatments at the end of the test and therefore an average Mn concentration could be used for the toxicity estimate. Interestingly, a loss of Mn was not observed in the following two *H. viridissima* toxicity tests and the concentrations of Mn at the end of the test were within 10% of the starting concentrations.

Table 2 Manganese toxicity estimates to six local freshwater species in Maglea Creek Water

Species	IC10 (µg L^{-1})	IC50 (µg L^{-1})
Chlorella sp.	12×10^3 ($10 – 14 \times 10^3$)	60×10^3 ($55 – 70 \times 10^3$)
L. aequinoctialis	2200 (910 - 3400)	11×10^3 ($9 – 13 \times 10^3$)
H. viridissima	140 (100 - 180)	1380 (1200 - 1560)
M. macleayi	610 (500 - 690)	1100 (1030 - 1170)
A. cumingi	340 (830 - 920)	5660 (2830 - 12660)
M. mogurnda	80×10^3 ($40 – 110 \times 10^3$)	240×10^3 ($200 – 320 \times 10^3$)

The fate and rate of the Mn loss in the test system was specifically examined (see below). The 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. The IC10 for *H. viridissima* was two times higher in MCW compared to Narradj Creek Water at 140 µg L^{-1} compared to 70 µg L^{-1} (Harford et al. 2009). Despite this difference, an overlap in the confidence intervals of these toxicity estimates indicates that they are not significantly different.

The 99% species protection TV derived from an SSD based on only the six local species' data was 4.1 (0.7 – 182, 95%CI) µg L^{-1} Mn, which is below the 50th percentile of the concentrations measured at the Magela Creek Upstream Monitoring site (Figure 2). Thus, it cannot be applied as a guideline value. It appears that this low TV is a function of both the wide range of toxicity estimates, and the small sample size, used in the SSD. This issue prompted the inclusion of the additional non-local species toxicity data detailed above. Inclusion of international data from three toxicity tests conducted in relevant physcio-chemical conditions produced a 99% species protection TV of 73 (33 – 466, 95% CI) µg L^{-1} Mn (Figure 3). This TV is two times lower than the IC10 for *H. viridissima* of 140 µg L^{-1} and is three times higher than the current guideline of 26 µg L^{-1}, which is derived from Mn concentrations measured upstream of the mine (Iles 2004). The value of 73 µg L^{-1} has never been exceeded in the creek and is 1.5 times higher than the highest concentration of Mn measured of 50 µg L^{-1} at the downstream monitoring site (Figure 2). In conclusion, the ecotoxicology-based TV of 73 µg L^{-1} is less conservative than the current statistically based 26 µg L^{-1}, but it still appears conservative based on existing biological effects literature, including the proposed PNEC of 123 µg L-1 for Europe (Peters et al. 2010).

In conclusion, the six local species showed a wide range of sensitivities to Mn. The *H. viridissima* toxicity estimate is the lowest reported in the literature to-date, indicating that there is an increased hazard due to the extremely soft waters of the receiving environment. The speciation of Mn in the toxicity tests is complex and would also be the case in the environment. Further studies concerning Mn speciation in the laboratory and in the field would help understand the risk of Mn in the environment. A site-specific TV of 75 µg L^{-1} Mn is recommended for Magela Creek. Subsequent discussions with stakeholders will need to occur in order to incorpotrate this TV into the regulatory framework as a ecotociological-based limit.

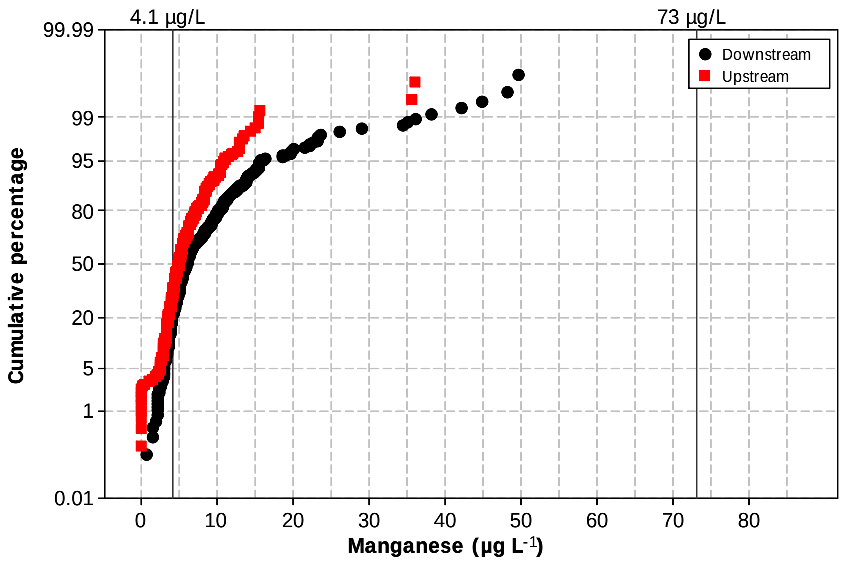

Figure 2 Comparison of the derived trigger values with environmental monitoring data at the upstream and downstream sites.

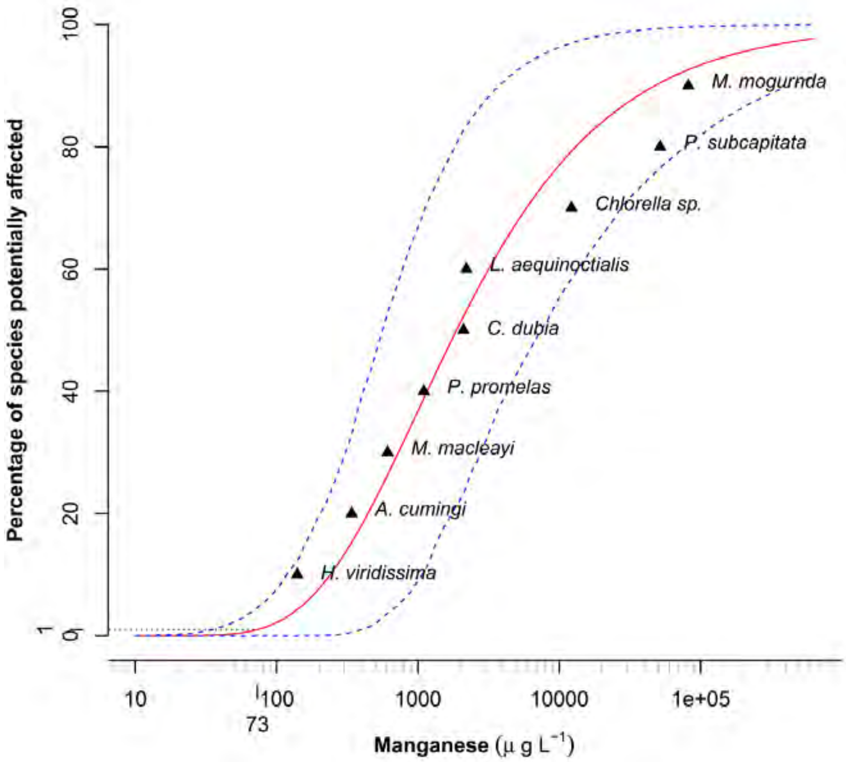

Figure 3 The Species Sensitivity Distribution that included the six local species and three northern hemisphere species. The 99% species protection TV of 75 µg L^{-1} is shown.

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An interim trigger value for ammonia in Magela Creek

AJ Harford & RA van Dam

Background

Although ammonia (NH_3) is present at very high concentrations in Ranger process water (~1000 mg L^{-1} Total Ammonia Nitrogen, TAN aka NH_3-N), to date it has presented negligible environmental risk as process water is not discharged to the off-site environment. However, these risks may increase in the future, through (i) the presence of residual amounts of ammonia in treated process water distillate from the brine concentrator (~1 mg TAN L^{-1}, Harford et al. 2013), which will be discharged to the environment, and (ii) potential seepage of ammonia from in-pit tailings post-closure. Consequently, a site-specific water quality trigger value (TV) for ammonia in Magela Creek is required.

Preliminary toxicity testing of ammonia has been undertaken for at least one local species, the green hydra, *Hydra virdissima*. This species was found to be highly sensitive to ammonia at pH 8, with EC_{10} and EC_{50} (i.e. concentrations causing 10 and 50% effect) values of 0.22 and 1.6 mg TANL^{-1}, respectively (van Dam et al. 2011). However, ammonia toxicity under the typical physico-chemical conditions of Magela Creek water is likely to be lower than this, as its pH typically ranges from 5.4 to 6.4, where the proportion of the more toxic unionised form of ammonia (NH_3) is approximately 30 times lower than at pH 8.0 (Emerson et al. 1975). A project that aims to understand ammonia toxicity under physico-chemical conditions relevant to the off-site surface water environment (i.e. Magela Creek – channel and billabongs) to a range of local freshwater species has recently commenced.

The toxicity test results will be used to derive site-specific TVs that can be incorporated into the regulatory regime; however, there is a more immediate need for an interim TV due to the imminent commissioning of the brine concentrator plant and a need to discharge large volumes of distillate during the 2013–2014 wet-season. Fortunately, there is a large body of literature concerning ammonia toxicity, and Australia and New Zealand, (ANZECC/ARMCANZ 2000) Canada (Environment Canada, 2010), the UK (UKTAG, 2007) and the USA (USEPA, 2013), have all produced ammonia water quality guidelines (WQGs; also variously referred to as TVs, criteria or standards). These WQGs can be adjusted for a specific temperature and pH because in aquatic systems the un-ionised ammonia (NH_3) is in equilibrium with the ionised NH_4^+ ions. Toxicity is predominantly caused by the unionised NH_3 molecule and ammonia speciation is driven by pH and temperature.

Dissociation constants (pK_a) for ammonia have been reported by Emerson et al. (1975). Moreover, the relationship between pH, temperature and ammonia toxicity has been extensively characterised by the USEPA (1999). As a result, the guidance documents provide algorithms that allow adjustment of trigger values for a specific pH and temperature. The USEPA has recently finalised its ambient water quality criteria for ammonia (USEPA 2013), which includes an analysis of the most recent quality-checked toxicity data. Together, this information enables the derivation of an interim site-specific TV for Magela Creek. Subsequently, the laboratory toxicity testing will be used to validate or modify the TV.

Methods

The most up-to-date, quality-checked ammonia toxicity datasets are provided by USEPA (2013) and these were used for derivation of the site-specific interim TV (Table 1). This dataset included the Species Mean Chronic Values (SMCV, i.e. the mean of all toxicity estimates from the same species) and the Genus Mean Chronic Values (GMCV, i.e. mean of all SMCVs for a genus). The chronic measures of effect used by the USEPA included NOEC (No Observed Effect Concentration) and MATC (Maximum Acceptable Toxicant Concentration) but EC_{20} toxicity estimates were the preferred option. Where possible toxicity data were converted to an EC_{20} to determine the SMCV. It should also be noted that phytoplankton and aquatic plants are not included in the USEPA (2013) or the ANZECC/AMRCANZ (2000) WQG derivation (see Results and Discussion).

Table 1 Genus Mean Chronic Values and Species Mean Chronic Values that were converted to Total Ammonia Nitrogen and normalised to pH 7 and 20°C (USEPA, 2013)

Rank	GMCV [a] (mg TAN L^{-1})	Species	SMCV [b] (mg TAN L^{-1})
16	73.74	Stonefly, *Pteronarcella badia*	73.74
15	53.75	Water flea, *Ceriodaphnia acanthina*	64.10
		Water flea, *Ceriodaphnia dubia*	45.08
14	41.46	Water flea, *Daphnia magna*	41.46
13	29.17	Amphipod, *Hyalella azteca*	29.17
12	21.36	Channel catfish, *Ictalurus punctatus*	21.36
11	20.38	Northern pike, *Esox lucius*	20.38
10	16.53	Common carp, *Cyprinus carpio*	16.53
9	12.02	Lahontan cutthroat trout, *Oncorhynchus clarkii henshawi*	25.83
		Rainbow trout, *Oncorhynchus mykiss*	6.66
		Sockeye salmon, *Oncorhynchus nerka*	10.09
8	11.62	White sucker, *Catostomus commersonii*	11.62
7	11.07	Smallmouth bass, *Micropterus dolomieu*	11.07
6	9.187	Fathead minnow, *Pimephales promelas*	9.19
5	7.828	Pebblesnail, *Fluminicola* sp.	7.83
4	7.547	Long fingernail clam, *Musculium transversum*	7.55
3	6.92	Green sunfish, *Lepomis cyanellus*	14.63
		Bluegill, *Lepomis macrochirus*	3.27
2	3.501	Rainbow mussel, *Villosa iris*	3.50
1	2.126	Fatmucket clam, *Lampsilis siliquoidea*	3.21
		Wavy-rayed lamp mussel, *Lampsilis fasciola*	1.41

a Genus Mean Chronic Value (i.e. the average of all toxicity estimates from all the species in a genus). b Species Mean Chronic Value (i.e. the average of all toxicity estimate collected for the same species).

Note that four significant figures are shown for GMCV and SMCVs in order to prevent rounding errors in calculations but this does not reflect their precision.

Where required, the SMCV and GMCVs were converted to TAN and normalised to pH 7 and 20°C. Species Sensitivity Distributions of the pH- and temperature-standardised SMCV and GMCV values were then constructed in order to derive a 99% species protection TV[1] (99% species protection being selected due to the high conservation value of Magela Creek). These values were then modified for a pH of 6.44 and temperature of 31.9°C. The pH and temperature conditions chosen for the site-specific TV were derived from the 90th percentile of data from the surface water chemistry monitoring programme (*n*=76,132). The 90th percentiles were chosen because they represent a worse-case scenario in that ammonia is more toxic at higher pH and temperature. Trigger values were adjusted using the formula provided in USEPA (2013) to derive a "Chronic Criterion" or CCC for Magela Creek.

$$\mathrm{CCC} = \frac{\text{Trigger Value}}{\text{Lowest GMCV}} \times \left(\frac{0.0278}{1+10^{7.688-pH}} + \frac{1.1994}{1+10^{pH-7.688}}\right) \times (\text{Lowest GMCV} \times 10^{0.028\times(20-T)} \quad (1)$$

where T = Temperature in °Celsius; and the Lowest GMCV = 2.126 for *Lampsilis* (Table 1)

In order to "reality check" the interim site-specific TV, four key national water quality guideline documents (UKTAG 2007; ANZECC/ARMCANZ 2000; Environment Canada, 2010; USEPA 2013) were also reviewed, and the default (i.e. in general 95% species protection values) WQGs were converted to equivalent values for comparison. This was required because the various ammonia guidance documents have used different ways to express their ammonia guideline values. Specifically, these values have been expressed in three different ways: 1) un-ionised ammonia; 2) total ammonia; and 3) total ammonia nitrogen (NH_3-N or TAN). Conversion to and from unionised ammonia, total ammonia and TAN was achieved using the following equations.

The percentage of un-ionised ammonia in a solution at a specific pH and temperature is calculated with equations 2 and 3 (ANZECC/ARMCANZ 2000).

$$\mathrm{pKa} = \frac{2729.69}{\mathrm{T}} + 0.1105 - 0.000071\mathrm{T} \quad (2)$$

where T is in °Kelvin, i.e. 273.16 + T °Celsius

$$\%\ \text{un-ionised } NH_3 = \frac{100}{1+10^{(\mathrm{pKa}\,-\,\mathrm{pH})}} \quad (3)$$

Un-ionised ammonia can be converted to total ammonia (ie the sum concentration of NH_3 + NH_4^+ ions) using equation 3 (ANZECC/ARMCANZ 2000).

$$\text{Total ammonia} = [NH_3] + \frac{[NH_3]}{10^{\mathrm{pH}\,-\,\mathrm{pKa}}} \quad (4)$$

[1] The ANZECC/ARMCANZ terminology of "Species Sensitivity Distribution" and "99% species protection TV" are being retained for the results and distributions constructed with Genus Mean Chronic Values.

Total ammonia can be converted to Total Ammonia Nitrogen (TAN) by multiplying by 14/17, or 0.8235 (equation 4, ANZECC/ARMCANZ 2000). This unit of measurement is important because TAN is the value often reported by environmental chemistry laboratories and expressed as NH_3-N.

$$\text{Total Ammonia Nitrogen (TAN, or } NH_3\text{-N)} = \text{Total ammonia} \times 0.8235 \qquad (5)$$

In order to make these default WQGs more comparable to the interim site-specific TV, they were also adjusted to the conditions of pH 6.44 and 31.9°C using equation 1.

Results and Discussion

The method of TV derivation recommended by ANZEEC/ARMCANZ (2000) involves construction of Species Sensitivity Distributions using species, rather than genus, toxicity estimates. However, the 99% species protection TV derived from the USEPA's SMCV dataset resulted in a higher TV of 1.8 mg TAN L^{-1}, compared to the corresponding values based on the GMCV dataset. Therefore, the GMCVs were used in the interim TV derivation in order to ensure the TV was conservative and protective of the environment. A 99% species protection TV of 1.4 mg TAN L^{-1} was derived from the Species Sensitivity Distribution constructed using the USEPA's GMCV data that have been normalised to pH 7 and 20°C (Figure 1). From this, a 99% species protection TV of 0.74 mg TAN L^{-1} was calculated for the site-specific conditions of pH 6.4 and 32°C using equation 1.

Therefore, it is recommended that an interim 99% TV for ammonia of 0.7 mg TAN L^{-1} be applied at MC009.

Future laboratory toxicity testing will be used to modify this TV as necessary. It is also important to note that phytoplankton and aquatic plants are not included in the USEPA dataset because,

> "The data available regarding the toxicity of ammonia to freshwater phytoplankton and vascular plants reported in the 1985 AWQC (Ambient Water Quality Criteria, USEPA 1985) document indicate that aquatic plants appear to be two orders of magnitude less sensitive than the aquatic animals tested, it is assumed that any ammonia criterion appropriate for the protection of freshwater aquatic animals will also be protective of aquatic vegetation"

Hence, the USEPA (2013) WQG does not address issues of phytoplankton stimulation in response to ammonia exposure and the adverse effects this might cause in the environment. Toxicity testing of *Chlorella* sp. in the ERISS laboratory will provide insights into this issue and the stimulation of algal growth may be incorporated into the derivation of a site-specific TV, or may need to be addressed separately (e.g. through field-based or mesocosm data).

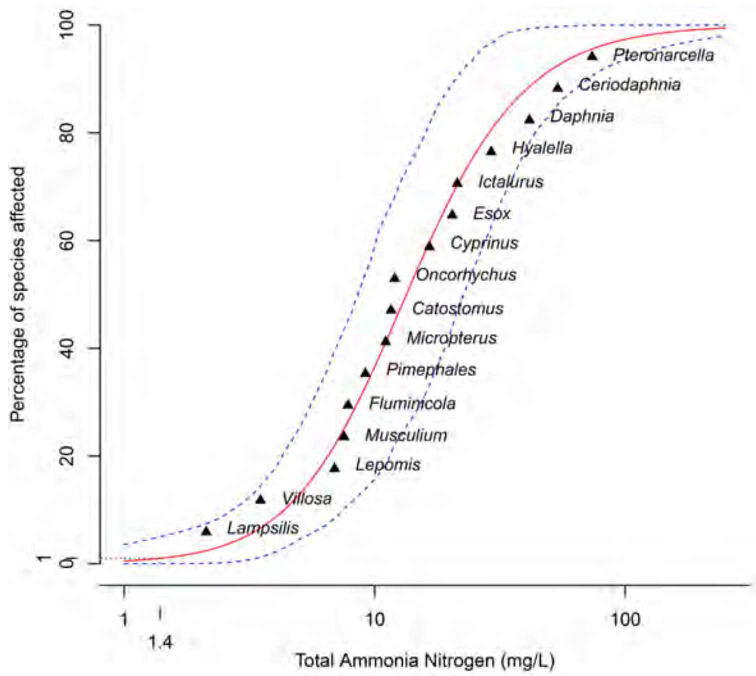

Figure 1 Species Sensitivity Distribution of Genus Mean Chronic Values from USEPA (2013). The value of 1.4 mg TAN L^{-1} represents a 99% species protection Trigger Value (pH 7 and 20°C).

In order to "reality check" the interim TV, it was compared to the default WQGs from ANZECC/ARMCANZ (2000), UKTAG (2007), Environment Canada (2010) and USEPA (2013). However, there are some issues in doing this because the ammonia guidance documents report default WQGs in different concentration units and are also derived using different methods. Specifically, the UK and Canadian key WQGs were reported as un-ionised NH_3, while the Australian/New Zealand and USA WQGs are primarily reported as TAN at specified pH and temperature. Therefore, all WQGs were converted and modified for local conditions of pH 6.4 and 32°C (Table 2).

Table 2 Comparison of the interim ammonia Trigger Value with Water Quality Guidelines reported in federal government documents

		Water Quality Guideline at pH 6.4 and 32°C		
Agency	**Key reported WQG**	**Un-ionised ammonia (mg NH_3 L^{-1})**	**Total ammonia (mg L^{-1} NH_3)**	**Total Ammonia Nitrogen (mg TAN L^{-1})**
USEPA (2013)	1.9 mg TAN L^{-1} at pH 7 and 20°C	0.01	1.2	1.0
Environment Canada (2010)	0.019 mg L^{-1} un-ionised NH_3	0.019	7.5	6.2
UKTAG (2007)	0.0011 mg L^{-1} unionised NH_3	0.001	0.44	0.36
ANZECC/ARMCANZ (2000)	0.9 mg TANL^{-1} at pH 8 and 20°C	0.042	16.6	13.7
Interim 99% site-specific TV	***N/A***	***0.008***	***0.9***	***0.7***

Although the interim TV has been derived using 99% species protection, it is not the most conservative WQG and is only 0.3 mg TAN L^{-1} below the default USEPA (2013) value of 1.1 mg TAN L^{-1}. The most conservative WQG was that of UKTAG (2007). However, it should be noted that this value was not based on a Species Sensitivity Distribution, but derived from a Lowest Observed Effect Concentration of 0.022 mg L^{-1} un-ionised ammonia (i.e. 2.1 mg TAN L^{-1} at pH 7.5 and 15°C, Solbe and Shurben, 1989), which had a conversion (to a NOEC) factor of 2 and a Safety Factor of 10 applied. Consequently, it is an overly-conservative value. The USEPA's WQG was derived using a different approach to the one taken for the interim TV. Specifically, it was derived from a regression analysis of a log-triangular curve using only the four most sensitive GMCVs. Hence, the USEPA (2013) WQG may also have conservatism in the method. The Environment Canada (2010) WQG was six times higher than the recommended interim TV and the USA WQG. However, it appears to be unchanged from an Environment Canada (1999) ammonia guidance document and was therefore unlikely to include the most recent ammonia toxicity data. The Canadian WQG was derived from a similar Species Sensitivity Distribution approach to the one used for the interim TV, except that the WQG is taken from the lower 95% confidence limit of the 5th percentile, which increases its conservatism. The highest WQG value is that of ANZECC/ARMCANZ (2000). It was derived using the same method as the interim TV used here, except that a 95% species protection level was used. This was the least conservative method of all the WQGs, which is reflected in the higher values. It was also derived 13 years ago and does not include the most recent toxicity data.

In conclusion, the recommended interim TV of 0.7 mg TAN L^{-1} considers the most recent international ammonia toxicity data and is more conservative than the ANZEEC/ARMCANZ (2000) value. The interim TV is seven times higher than the highest measured ammonia concentration at MG009 (0.1 mg TAN L^{-1}). Ultimately, the TV will be tested through laboratory toxicity tests.

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Hydra viridissima (Green hydra) recovers from exposure to multiple magnesium pulses

A Prouse, A Hogan, AJ Harford, RA van Dam & D Nugegoda

Background

Magnesium (Mg) has historically been reported to be of low toxicity under most environmental conditions and has, therefore, received little attention as a toxicant (van Dam et al. 2010). Hence, Mg was considered to be a contaminant of lower concern in Ranger Mine discharges. This perception changed in the mid-1990s when Electrical Conductivity (EC) was correlated with changes in benthic community compositions with magnesium sulphate ($MgSO_4$) being the primary contributor to EC (O'Conner et al. 1997). Site-specific ecotoxicological testing confirmed that Mg was more toxic, than in previous Mg studies, in the extremely soft waters surrounding Ranger Mine (van Dam et al. 2010). This work also established that SO_4 was of much lower toxicity than Mg. Consequently, a site-specific Mg Trigger Value (TV; a water quality guideline) based on toxicity data from six local species was developed for Magela Creek (van Dam et al. 2010).

The discharges of Mg to Magela Creek were well understood due to a long term chemical monitoring programme involving weekly grab samples of Mg and the continuous monitoring of EC both up and downstream of the mine (Sinclair et al. 2013). As Mg concentrations could be reliably inferred from EC using a well-established relationship, the continuous EC data provided a highly resolute temporal record of Mg concentrations in the creek (Turner & Jones 2010). This has been examined in order to understand the potential Mg exposure patterns to organisms in the local environment. Generally, increases in Mg concentration have been found to occur as very short 'pulses' lasting minutes to hours (Hogan et al. 2013). In contrast, the data used to derive the site-specific Mg TV was derived from continuous exposure toxicity tests where organisms were exposed for periods of three to six days (van Dam et al. 2010). The literature reported that pulse exposures usually yielded less sensitive responses compared to continuous exposures of equivalent magnitudes (Handy 1994). Hence, it was thought that the Mg TV may have been overly conservative when applied to short-term Mg events in Magela Creek. Therefore, further research investigated the effects of shorter Mg exposures (administered as 4, 8 and 24 h pulses within the standard test protocol) to each of the organisms used in the continuous exposure TV. Shorter pulses were found to be less toxic than a continuous exposure for all organisms tested and the data was used to establish a duration-based TV for Mg in Magela Creek (Hogan et al. 2013). The use of this TV in a regulatory framework was described by Sinclair et al. (2014).

A limitation of the duration-based TV was that it was based on single pulse toxicity data. Hence, it was unknown if the organisms had fully recovered from a pulse exposure or if they carried damage resulting in higher sensitivity to subsequent pulses. It was recognised that multiple pulses did occur in the creek over short time frames and, if organism recovery was slow or incomplete, then the TV could potentially be under protective. This uncertainty was highlighted by Hogan et al. (2013) and it was recommended that organism recovery time and the potential for carry-over toxicity be investigated.

The present study, a collaboration between SSD and RMIT University (Prouse 2013), addressed these questions for one species, the Green hydra (*Hydra viridissima*). *Hydra viridissima* was exposed to a range of multiple pulse scenarios relevant to Magela Creek to allow a comparison of biological responses between combinations of single and multiple Mg pulse with varying pulse and inter-pulse durations. The broader aim of the study was to generate data that informed the use of the site-specific Mg TV framework when assessing multiple, closely-spaced pulses of Mg in Magela Creek.

Methods

Standard *Hydra viridissima* toxicity test

All experiments were based on the standard *H. viridissima* (hydra) test method described by Riethmuller et al. (2003). Hydra cultures were maintained in filtered Magela Creek Water (MCW) and fed newly hatched brine shrimp larvae (*Artemia franciscana*) followed by cleaning every 1 - 2 d. Healthy test hydra, each bearing a bud with early tentacle development, were selected from the culture bowls and transferred into experimental petri dishes containing 30 mL of test solution. Each dish contained ten hydroids with three replicate dishes for each treatment. The test containers were randomly placed in an environmental cabinet for 96 h at a temperature of 27 ± 1°C with a 12:12 h light/dark photoperiod.

Fresh test solution was dispensed each day and allowed to warm in the environmental cabinet. Observations on the general appearance of the hydra (e.g. rigidity, clubbing, colouration) and the number of hydra were recorded daily for 96 h. Each hydra was fed 3–4 freshly hatched brine shrimp and returned to the incubator for 4 h to allow digestion. After this time the test containers were cleaned and the solution renewed.

At the completion of the test (96 h), the number of hydroids in each test container was counted and the population growth rate calculated. From this, concentration-response curves were used to calculate toxicity estimates such as the IC10 and IC50 (concentrations causing a 10% and 50% inhibition of growth). The test was considered acceptable if >30 healthy hydroids were in each control dish at the end of the test period and if the water quality of the test solutions remained with specified criteria.

Analysis of continuous monitoring EC data

An analysis of the continuous monitoring EC data was conducted to determine the frequency, magnitude and duration of pulses and inter-pulse periods that had occurred in Magela Creek between 2005–2012. This information enabled environmentally relevant recovery periods to be tested. The method is described in Hogan et al. (2013) with the present analysis incorporating an additional two years of data and providing the first summary statistics on the inter-pulse periods.

Estimating time to apparent recovery

'Apparent recovery' was considered to have occurred after a single pulse when the growth rate of the pulsed treatments had returned to a similar growth rate as the control treatments. Hogan et al. (2013) reported that recovery appeared to have occurred in all hydra treatments surviving a 4 or 8 h Mg pulse, but that the higher magnitude 24 h pulses may not have recovered, prior to the end of the standard 96 h experiments. An estimate of apparent recovery was needed for the present study to provide guidance on the recovery periods to be tested in the 'true' recovery experiments (see below). As such, a 24 h pulse test was run and the duration extended until apparent recovery was observed (192–240 h depending on treatment).

The single 24 h pulse growth rate data from the present study and the 4 h pulse data from Hogan et al. (2013) were incorporated into a control chart. This enabled the determination of when the Mg pulsed treatments had returned to growth rates within the normal range of the control treatments (i.e. within the 95% confidence intervals of the control mean growth rate).

Double pulse experiments to estimate the time to true recovery

'True recovery' was considered to have occurred when hydra exhibited similar sensitivity to the second pulse as they did to the first. Specifically, they had returned to a condition where their ability to withstand a subsequent pulse was equal to their original sensitivity. In order to determine when true recovery occurred, the hydra were exposed to multiple Mg pulses separated by different recovery periods.

Population growth rate relative to the control (i.e. percentage of control response) was used to assess the sensitivity of organisms after a single pulse or a second pulse. Magnesium pulses of 790 mg L^{-1} and 1100 mg L^{-1} for 4 h durations were separated by 2, 10, 18, 24, 48 and 72 h recovery periods.Pulses of 570 mg L^{-1}, 910 mg L^{-1} and 940 mg L^{-1} for 24 h durations were separated by 24, 48 and 168 h recovery periods. All exposures were undertaken at the environmentally relevant magnesium to calcium ratio of 9:1, as previous research had demonstrated the importance of calcium in moderating the toxic effects of Mg. Organisms were returned to the control medium once the pulse had ended.

Figure 1 provides a schematic of the test design for the 24 h multiple pulse exposures. The controls consisted of Magela Creek water that contained 0.9 mg L^{-1} Mg. All initial pulses were conducted in bulk hydra dishes to ensure sufficient organisms at the appropriate growth stage were available for the second pulse experiment. Organisms underwent either an initial or second Mg pulse in petri dishes to enable a comparison of the effects of a single versus a second pulse. All pulse treatments were compared back to their respective control (conducted in parallel) to calculate the 96 h percentage of control growth as an estimate of organism sensitivity to each treatment.

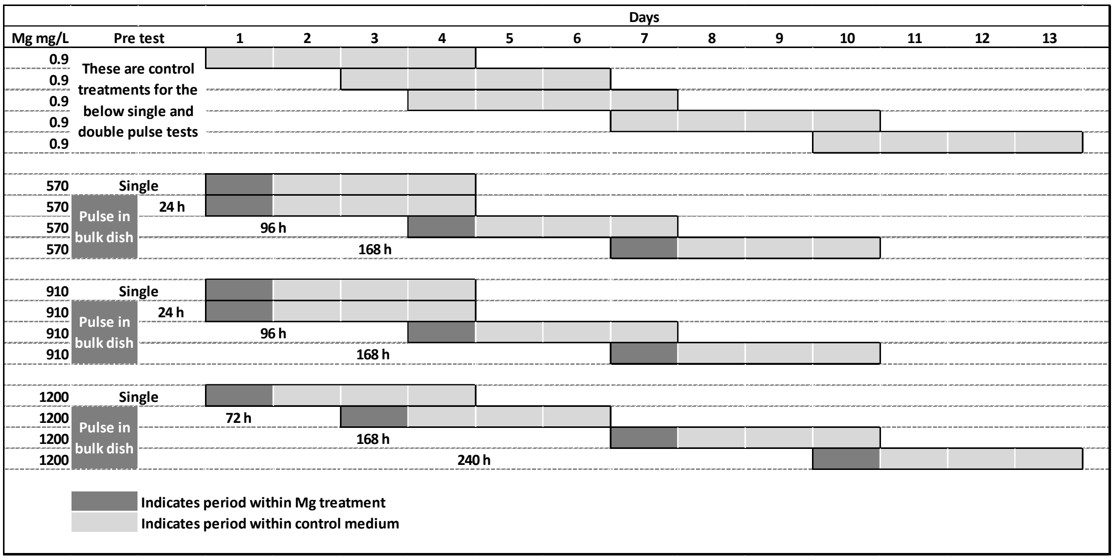


Figure 1 Double pulse test design showing the 24 h pulse experiment as an example. The observation period for each treatment is outlined.

Multiple magnesium pulse scenarios test

This test aimed to compare treatments that could be considered equivalent in terms of total pulse exposure duration and magnitude, but different in terms of the actual pulse and inter-pulse durations. Six pulse and recovery periods were used. Five treatments exposed the hydra to the same time-weighted concentration of Mg, with the other reflecting a potential conservative TV application approach where the maximum concentration is applied to the whole pulse and inter-pulse period.

The first four treatments exposed the hydra to 850 mg L^{-1} Mg for an overall exposure of 24-h but under different pulse scenarios (i.e. 1 × 24 h pulse, 6 × 4 h pulses, 3 × 8 h pulses and 4 × 6 h pulses; Figure 2). The 68-h continuous exposure to 300 mg L^{-1} Mg was a time-weighted average comparison to the 6 × 4-h pulse exposure scenario, while the 68-h pulse exposure to 850 mg L^{-1} provided a treatment that mimicked a potential conservative application of the TV where the duration over which a series of Mg events has occurred in the creek is considered one long event.

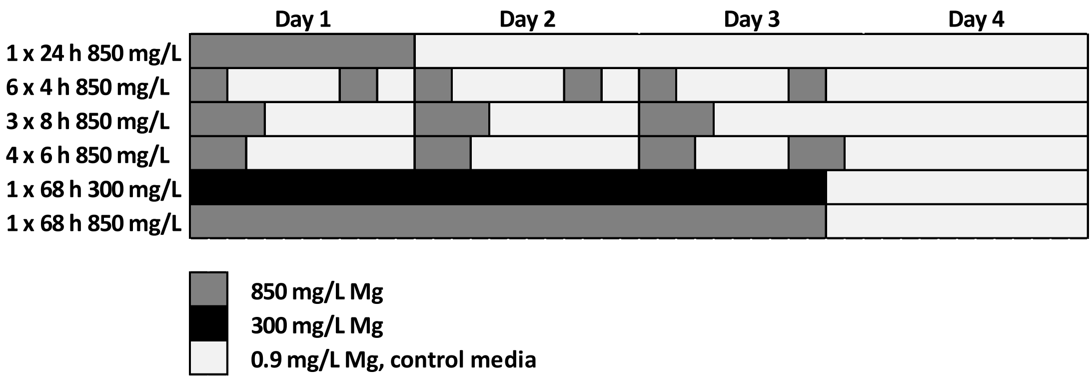


Figure 2 Multiple magnesium pulse and inter-pulse durations test design. Controls were also undertaken for each variation to quantify handling effects but are not shown in this figure.

Results

Analysis of continuous monitoring EC data

Seventy one Mg pulses were identified from the 2005–2006 wet season through to the 2011–2012 wet season, with 95% of the pulse durations being < 24 h and only 2 pulses with durations considered as chronic (> 96 h). A large proportion of the pulses (60%) were within the 4–24 h duration range assessed in this study. Thirty six percent of pulses were shorter than 4 h and the remaining 4% were greater than 24 h in duration.

Inter-pulse periods were often short with the shortest lasting only 1.3 h. Multiple pulses often occurred within a short timeframe, with 30 % of inter-pulse periods being < 24 h duration, 50 % < 48 h duration and 60% < 96 h duration. This further emphasised the need for this study and provided guidance on the recovery periods to be tested.

Estimating time to apparent recovery

All hydra treatments recovered provided that they survived the pulse period, but recovery took longer the higher the Mg concentration. Based on analysis of data from tests undertaken by Hogan et al. (2013), recovery from a 4 h Mg exposure to 212, 422, 651, 821, 842 and 1030 mg L^{-1} Mg was very rapid, occurring prior to the first observation at 24 h (Figure 3). Recovery of hydra undergoing a 4 h Mg exposure to the highest surviving treatment of 1300 mg L^{-1} Mg was apparent by 48 h. The estimated times to apparent recovery after a single 24 h pulse were between 24 h and 48 h for 210, 420 and 630 mg L^{-1} Mg; between 72 h and 96 h for 860 mg L^{-1} Mg; and 192 h for 1100 mg L^{-1} Mg (Figure 3).

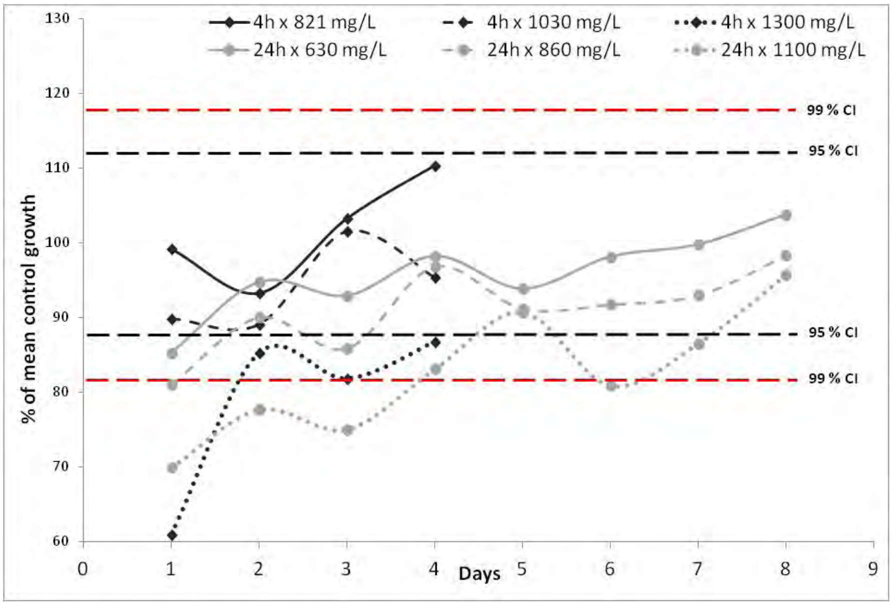


Figure 3 The time to apparent recovery for *Hydra viridissima*. The 99 % CI represent the expected range of typical control growth.

Double pulse experiments to estimate the time to true recovery

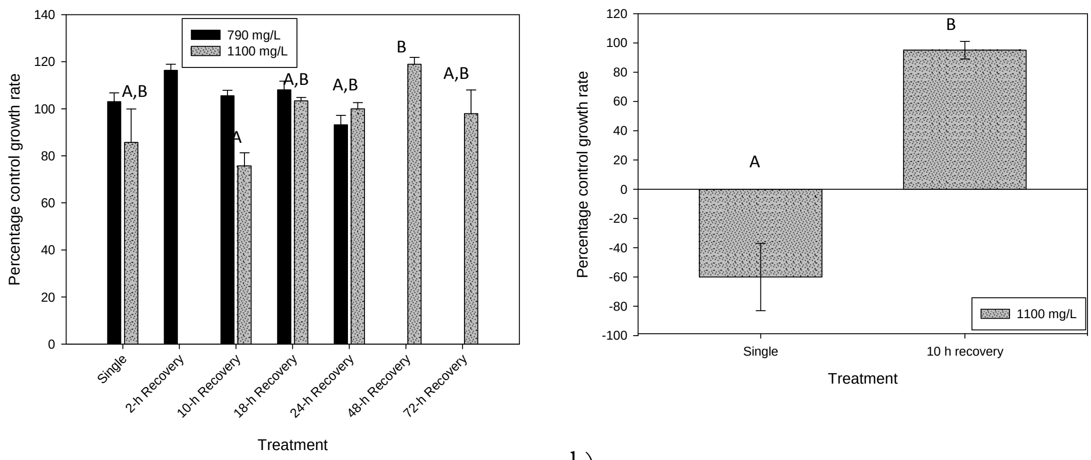
All organisms showed a statistically similar or reduced sensitivity to the second pulse when compared to the single pulse sensitivity. This indicated that full recovery occurred prior to the exposure of the second pulse for the recovery periods tested.

The percentage of control growth rate and standard error (SE) of a single 4-h 1100 mg L^{-1} pulse was 86% ± 14% (Figure 4a). Percentage control growth rates ± SE, for the double pulse treatments were 76 ± 6%, 103 ± 1%, 100 ± 3%, 119 ± 3% and 98 ± 10% for 10, 18, 24, 48 and 72 h recovery periods, respectively. Given the higher than ideal variability in the single pulse treatment, and that the 10 h recovery treatment had a lower (albeit non-significant P = 0.930) growth rate, it was decided to repeat these two treatments to confirm that recovery had occurred by 10 h (Figure 4b and corresponding text below).

Organisms exposed to a second 1100 mg L^{-1} pulse after a 48 h recovery showed a statistically significant higher percentage control growth than the single pulse 1100 mg L^{-1} treatment (P = 0.027, Figure 4a), indicating an increased capacity for the hydra to withstand the second pulse.

A single 4 h Mg pulse of 710 mg L^{-1} was less toxic than expected and did not elicit any effect on the hydra (Figure 4a). Accordingly, recovery could not be assessed for this treatment.

A higher than expected level of sensitivity to a single 1100 mg L^{-1} Mg pulse was observed in the repeat experiment with a percentage control growth rate of -60 ± 23% being observed. This occasionally occurs in hydra testing because Mg effects occur over a narrow concentration range for this species. As such, a small change in organism sensitivity can result in a response that is considerably larger or smaller than expected. Despite this, the data showed that the hydra were significantly less sensitive to a second pulse administered after 10-h recovery period (percentage control response ± SE = 95 ± 6%, P = 0.002, Figure 4b). This indicated that a 10 h inter-pulse period was sufficient for the hydra to recover from a 4 h pulse of 1100 mg L^{-1} Mg, thus supporting the results of other double pulse scenarios.



a) b)

Figure 4 *Hydra viridissima* mean percentage of control growth rate after a single 4-h magnesium pulse and a second 4-h magnesium pulse after the nominated recovery periods. a) Full original experiment b) repeated treatments. Significant differences between treatments (Tukey's, Dunnett's and or Kruskall Wallis tests) are denoted by different letters.

For the 24 h pulse exposures, all 570 mg L^{-1} treatments exposed to single and second pulse resulted in similar percentage control growth rates (Figure 5a; P = 0.402). The percentage control growth rate (±SE) of a single pulse was at 78 ± 7%, while the double pulse treatments were 87 ± 6%, 96 ± 3% and 88 ± 10% for 24, 96 and 168 h recovery periods, respectively.

No statistically significant differences were found between the single 24 h pulse 910 mg L^{-1} treatment and the 910 mg L^{-1} second 24 h pulse treatments, indicating that true recovery had occurred within 24 h of the initial exposure (Figure 5). The percentage of control growth rates ±SE of a single pulse was 65 ± 2%, while for the double pulse treatments were 43 ± 9%, 81 ± 6% and 68 ± 9% for 24, 96 and 168 h recovery periods, respectively, indicating that full recovery had occurred for all recovery periods tested. The lower percentage control growth rate for the 24 h recovery treatment of 43 ± 9% was thought to be due to an issue with the availability of hydra at the appropriate life stage for test commencement. This was confirmed in a separate test (data not shown) but a repeat of the double pulse experiment for this treatment was considered necessary for confirmation (see Figure 5b and text below).

The 24 h pulse 940 mg L^{-1} single pulse and second pulse after 24 h recovery had percentage of control growth rates of 54 ± 7% and 102 ± 1% respectively (Figure 5b). The second pulse treatment had statistically higher percentage of control growth than the single pulse treatment (P = 0.002, Figure 5b), confirming that full recovery from a 24 h pulse occurred after a 24 h recovery period.

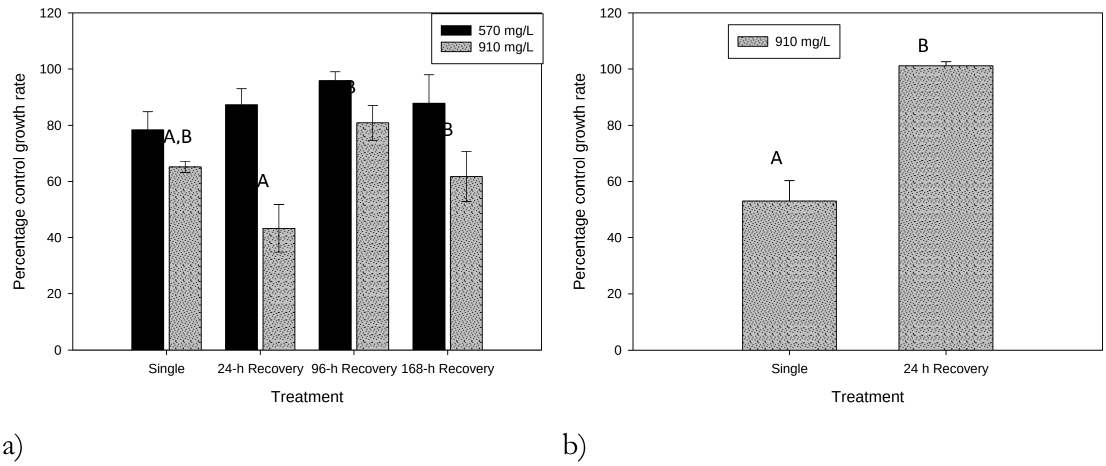


Figure 5 Mean percentage of control growth rates of *Hydra viridissima* after a single 24 h magnesium pulse and a second 24 h magnesium pulse after the nominated recovery periods. a) Full original experiment b) repeated treatments. Letters denote statistically similar treatments (Tukey's or Kruskall Wallis tests)

The recovery periods required to achieve a full recovery were all shorter than the apparent recovery period estimates used to select appropriate test durations. This indicates that apparent recovery from a single pulse provides a conservative indication that true recovery has occurred for this species.

Multiple magnesium pulse scenarios test

Five variations of equivalent time-weighted average concentrations were used to compare sensitivity of organisms after multiple pulses within a standard 96-h toxicity test. The sensitivity of the organisms to the multiple pulses generally corresponded with the time-weighted average response although hydra growth was statistically higher in two multiple pulse scenarios indicating that this species benefited from recovery periods (Figure 6).

The growth of two of the three multiple pulse treatments was significantly higher (P = <0.001) with percentage control growth rates ± SE of 84 ± 6% and 82 ± 7% for the 3 × 8 h and 4 × 6 h pulses compared to only 51 ± 10% for the 1 × 24 h pulse. The 6 × 4 h pulse was statistically similar (P = 0.148) to the 24 h pulse despite the higher percentage of control growth rate of 78 ± 7%. The single 68 h exposure to 300 mg L^{-1} treatment showed statistically higher growth at 90 ± 2% than the 24 h exposure to 850 mg L^{-1} treatment, and was similar to all the multiple pulse treatments (P = <0.001, Figure 6). The potential regulatory approach of applying a total additive time, or the 68 h exposure to 850 mg L^{-1}, showed statistically lower percentage control growth at 5 ± 5% than all other treatments indicating that, based on the response of *H. viridissima* alone, this would be an overly-conservative approach.

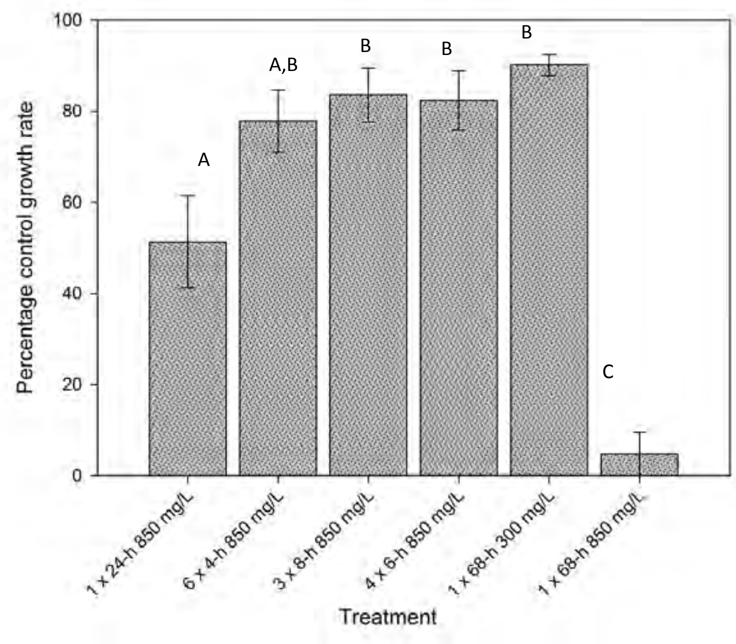


Figure 6 Hydra viridissima mean 96-h percentage of control growth rate after the pulse exposure patterns described. Significant differences between treatments are shown by differing letters (Tukey's post-hoc test).

Conclusions

Full recovery of *H. viridissima* was observed for all Mg concentrations, pulse durations and recovery periods tested. This indicates that this species fully recovered from sub-lethal Mg pulses and do not carry over any toxicity to subsequent pulses.

Full recovery occurred more rapidly than expected based on observations of apparent recovery from previous single Mg pulse tests, demonstrating that apparent recovery is a conservative indicator of full recovery for this species.

While additional organisms should be tested, this study suggests that Mg pulses more than 24 h apart may be considered as independent events when applying the TV in a regulatory framework.

Hydra viridissima appear to be slightly less sensitive to multiple short pulses than one longer pulse of equivalent exposure. Although further species need to be tested, this indicates that time weighted averaging may be a suitable approach when assessing the risk of multiple Mg pulses with inter-pulse periods < 24 h apart.

Further work

A manuscript is currently being prepared for this research to be submitted to an international journal.

As this study focussed on one organism, further toxicity testing on several species would need to be undertaken prior to recommending any changes to the site-specific water quality guideline. Given the high sensitivity and slow apparent recovery from single pulse experiments using *Moinodaphnia macleayi,* this species is considered a priority test organism.

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Revision of the site-specific trigger values for uranium in Magela Creek

RA van Dam, A Hogan & AJ Harford

Introduction

The site-specific water quality trigger framework for uranium (U) in Magela Creek was set in 2004 (Iles 2004), and comprises a U Limit, and its associated Action and Focus levels. The three values were derived from a species sensitivity distribution (effectively, a cumulative probability distribution of species' sensitivity to U) based on chronic toxicity data for five local species from four taxonomic groups, as per methods described by ANZECC/ARMCANZ (2000) (Hogan et al. 2005). The Limit of 6 $\mu g\ L^{-1}$ represented the 1st percentile of the species sensitivity distribution, or the concentration predicted to protect at least 99% of species, while the Action and Focus values of 0.9 and 0.3 $\mu g\ L^{-1}$, respectively, represented the lower 80% and 95% confidence intervals of the Limit value, respectively.

Since then, new data and increased knowledge have been acquired, including the following:

- U toxicity data have been published for an additional three local species, *L. aequinoctialis*, *Ceratophylum demersum* and *A. cumingi* (Hogan et al. 2010; Markich 2013);
- Additional U toxicity data have been published for at least three species for which data already existed (Trenfield et al. 2011; van Dam et al. 2012);
- An algorithm has been developed, largely based on local species data, to modify U trigger values based on environmental dissolved organic carbon (DOC) concentrations (van Dam et al. 2012).

Given the new data and knowledge, it is timely for the Magela Creek site-specific trigger framework to be revised.

Methods

All existing chronic toxicity data based on ecologically relevant endpoints (e.g. reproduction, growth, survival) for freshwater species local to the ARR were compiled and quality-checked using the quality assessment procedure proposed for the current revision of ANZECC/ARMCANZ (2000). Data were included from toxicity tests undertaken using natural or synthetic Magela Creek water. All data that passed the standard quality assessment procedure were subjected to further scrutiny (e.g. closer inspection of concentration-response relationships and resultant toxicity estimates) to ensure the data were robust and reliable for use in deriving TVs.

As there were often multiple data available for a single species, specific rules have been used to guide selection or calculation of the most relevant value for use in TV derivation. These rules were consistent with the toxicant TV derivation method proposed for the current revision of ANZECC/ARMCANZ (2000). Given the work undertaken to characterise the effects of DOC on U toxicity (Trenfield et al. 2011, van Dam et al.

2012), it will be possible to incorporate a DOC correction factor into the site-specific U TV derivation process (i.e. the ability to adjust the U TV based on the DOC concentration in the Magela Creek channel and billabongs). To do this, all accepted U toxicity data were standardised to a DOC concentration of 1 mg L^{-1} using the approach developed by van Dam et al. (2012).

The site-specific U TV (at 1 mg L^{-1} DOC) will be derived for the 99% species protection level, from a species sensitivity distribution of the final toxicity values for the seven species, using the Burrlioz 2.0 software. Burrlioz 2.0 has been developed for deriving toxicant trigger values for Australia and New Zealand, and fits a 2 parameter log-logistic distribution to datasets where $n < 8$. The resultant U TV will be adjustable based on the DOC concentration in Magela Creek, using the approach developed by van Dam et al. (2012).

Progress to date

All data have been compiled, quality checked and assessed in order to arrive at a single toxicity value for each species. The final toxicity dataset contains data for seven local species from six taxonomic groups (*Chlorella* sp., *L. aequinoctialis*, *C. demersum*, *M. macleayi*, *A. cumingi*, *H. viridissima* and *M. mogurnda*).

Steps for completion

A revised site-specific U TV will shortly be derived. The implications of the revised TV being adopted as a revised Limit for U in Magela Creek will be assessed. This will include comparisons of the revised site-specific U TV with (i) the existing U Limit, (ii) measured U concentrations in Magela Creek downstream of Ranger, and (iii) other recent or current revisions of (national and international) U water quality trigger values/guidelines. Similarly, the implications of revised Focus and Action levels, calculated using the same method as that for the existing Limit, will be assessed. Subsequently, comprehensive stakeholder consultation will be undertaken to discuss and agree on any changes to the existing U Limit.

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Reference toxicity testing for routine toxicity test species

C Pease, K Cheng, AJ Harford, A Hogan, M Trenfield & RA van Dam

Background

Reference toxicity testing has been a routine part of the ***eriss*** Ecotoxicology Programme'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 2012–13 was to continue with the established reference toxicity testing programme, 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 *L. aequinoctialis* protocol in Riethmuller et al. (2003). Specifically, the diluent used has been modified to use 1% modified Hoaglands E (Cleland & Briggs 1969) and K medium (Maeng & Khudari 1973) and growth rate calculated from total surface area change has been included as an endpoint. Details of these changes have been documented in an Internal Report.

Progress

In total, 16 reference toxicants tests (*Chlorella* sp. – 3; *H. viridisima* – 3; *M. macleayi* – 5; *M. mogurnda* – 3; and *L. aequinoctialis* – 2) were completed during 2012–13. Of these tests, 15 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 Figure 1-6. The Ecotoxicology laboratory aims to complete four tests per species per annum. However, due to issues and method development with the *Chlorella* sp. and *M. macleayi* (see below), this was not achieved for four species (*Chlorella* sp., *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 and method development undertaken in 2012–13 for each component of the reference toxicity test programme is provided below.

Chlorella sp. (green alga)

Of the three *Chlorella* sp. tests all three were valid for this reporting period. For all tests, control growth rates were within the acceptability criterion of 1.4 ± 0.3 doublings/day. The running mean EC_{50} is 40 µg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 7 and 73 µg L^{-1} U, respectively (Figure 1). The current algal density used in reference toxicity tests is higher than would be found in natural systems. As such, we are undertaking method development to reduce the starting density of our tests using a flow cytometer which has the capability to detect lower cell densities compared to the previously used coulter counter. In past ARRTC reports we discussed the potential for residual nutrients in the culture medium ameliorating U toxicity. We are also currently undertaking method development working on reducing the nutrients used during tests.

H. viridissima (green hydra)

All three reference toxicity tests for *H. viridissima* were valid. There were no issues associated with this protocol. The running mean EC_{50} is 84 μg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 44 and 124 μg L^{-1} U, respectively (Figure 2).

M. macleayi (cladoceran)

Of the five reference toxicity tests for *M. macleayi* four were valid. The running mean EC_{50} is 91 μg L^{-1} with all but one test within the lower and upper warning limits (± 2 standard deviations) of -69 and 251 μg L^{-1} (Figure 3.). Test 1339Ib resulted in an EC_{50} of 266 μg L^{-1} which exceeds the upper warning limit of 251 μg L^{-1}. The failed test 1339Ia was due to the temperature conditions of the test not meeting our QA/QC checks.

The reduction in *M. macleayi* sensitivity to U continued into the 2012–13 period, however the sensitivity dropped back down to previous levels in the last test performed (1346I). As previously reported, differences in the sensitivity of *M. macleayi* have been attributed to different batches of fermented food with vitamins (FFV) with varying organic components that may result in increased binding of U to these components. There is currently a project underway dedicated to resolving the inconsistencies associated with *M. macleayi* diet which will help to resolve this issue.

M. mogurnda (fish)

The three toxicity tests for *M. mogurnda* were valid with the EC_{50} values within the lower and upper warning limits of 941 and 2049 μg L^{-1} U, respectively. There were no problems associated with this protocol. The current running mean EC_{50} is 1495 μg L^{-1} U (Figure 4).

L. aequinoctialis (duckweed)

The reference toxicant test method for *L. aequinoctialis* was finalised, and an internal report for the method development is about to be published. We are currently estimating the toxicity of U using two different methods; growth rates are measured based on frond number (Figure 5) as well as the change in total surface area (Figure 6). The sensitivities of these two different methods are similar (Figure 7), with the frond number method having a running mean EC_{50} of 11,141 μg L^{-1} U. All results are within the lower and upper warning limits of 3664 and 18,618 μg L^{-1} U, respectively (Figure 5). The surface area method current running mean EC_{50} is 8496 μg L^{-1}, with all results within the lower and upper warning limits of 5671 and 11321 μg L^{-1} (Figure 6).

Table 1 Summary of the results from reference toxicity tests

Species & endpoint	Test Code	EC_{50} (µg L^{-1})	Valid?	Comments
Chlorella sp (72-h cell division growth rate)	1313G	47 (39, 54)	Yes	
	1328G	46 (39, 55)	Yes	
	1339G	62 (51, 75)	Yes	
Hydra viridissima (96-h population growth)	1311B	101 (93, 107)	Yes	
	1327B	99 (89, 107)	Yes	
	1351B	84 (74, 100)	Yes	
Moinodaphnia macleayi (48-h immobilisation)	1309I	201 (197, 203)	Yes	
	1314I	171 (156, 184)	Yes	
	1339Ia	NC[a]	No	Incubator temp. dropped below 20°C
	1339Ib	266 (153, NC[a])	Yes	
	1346I	32 (17, 53)	Yes	
Mogurnda mogurnda (96-h sac fry survival)	1308E	1275 (1185, 1371)	Yes	
	1326E	1675 (1584, 1770)	Yes	
	1350E	1212 (1146, 1282)	Yes	
Lemna aquinoctialis (96-h growth rate)	1315L	7360 (6959, 7804)	Yes	
	1330L	10830 (10330, 11830)	Yes	

Values in parentheses represent 95 % confidence limits

a Not calculable

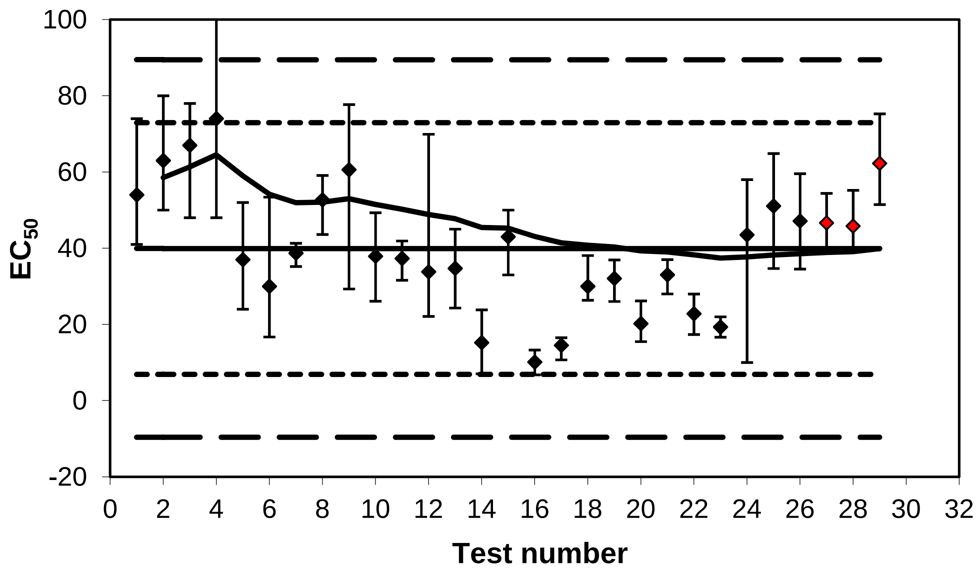

Figure 1 Reference toxicant control charts for *Chlorella* sp., as of October 2013. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) 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.

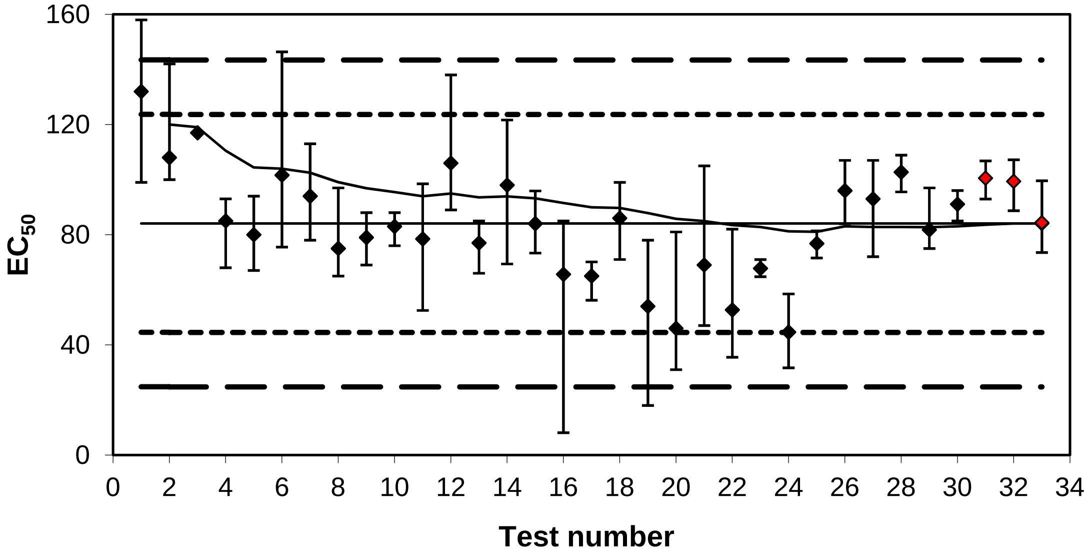

Figure 2 Reference toxicant control charts for *H. viridissima* as of October 2013. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) 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.

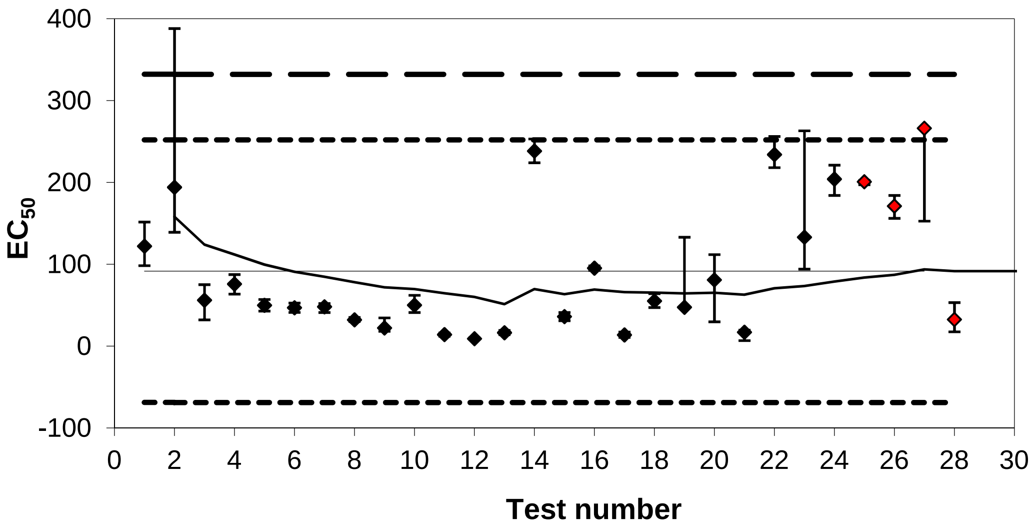

Figure 3 Reference toxicant control charts for *M. macleayi* as of October 2013. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of 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.

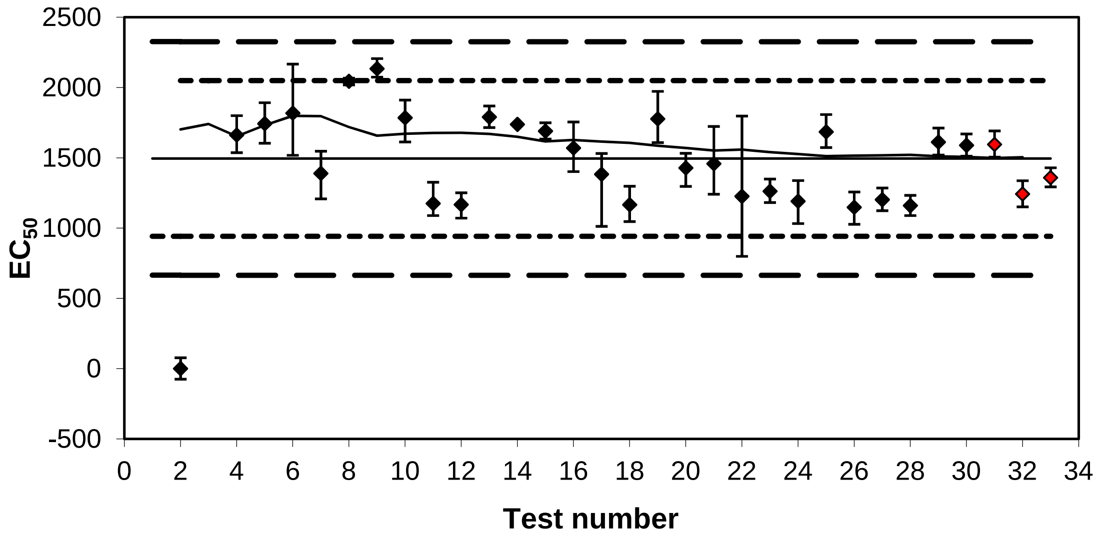

Figure 4 Reference toxicant control charts for *M. mogurnda* as of October 2013. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines lower and upper 99% confidence limits (± 3 standard deviations) of the 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.

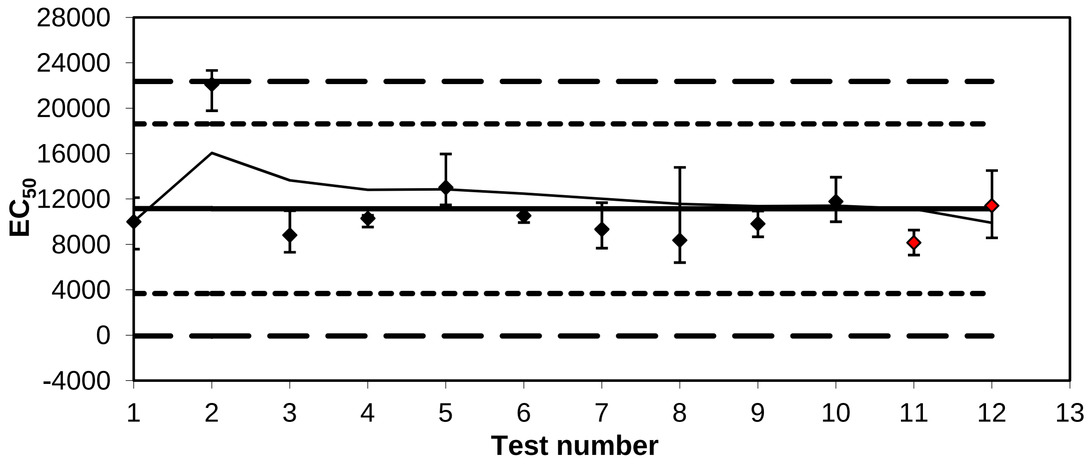

Figure 5 Reference toxicant control chart for *L. aequinoctialis* based on frond number as of October 2013. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 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.

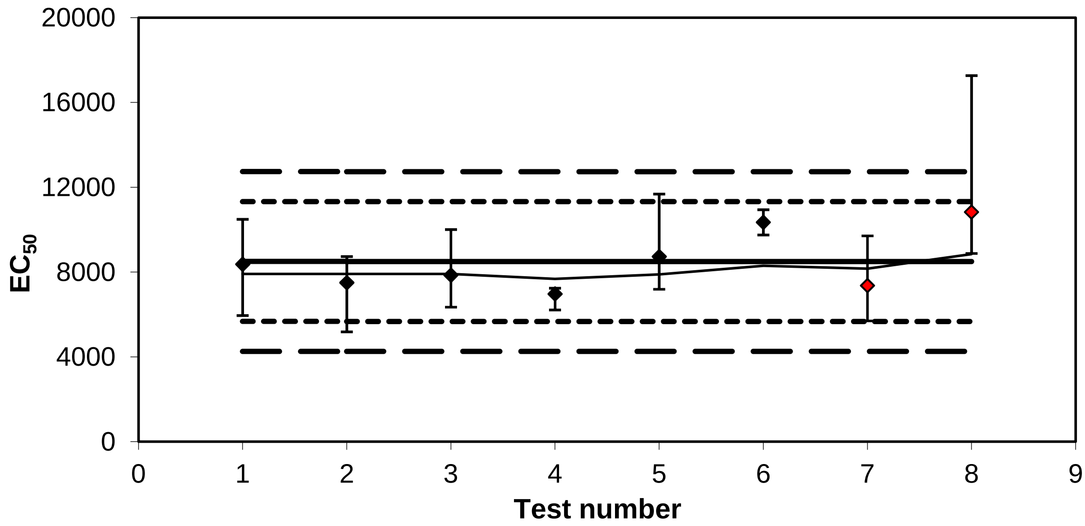

Figure 6 Reference toxicant control chart for *L. aequinoctialis* based on total surface area as of October 2013. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 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.

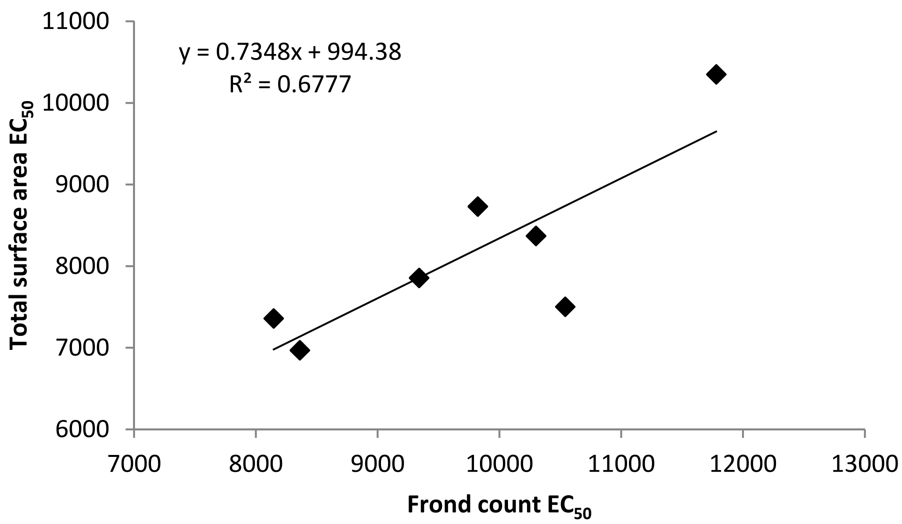

Figure 7 Comparison of U sensitivity between the two *L. aequinoctialis* reference toxicant test methods. Each data point represents the EC50s derived from the two different methods (frond number and total surface area) for 7 valid tests.

Planned testing in 2013–14

The reference toxicity testing programmes for all five species will continue in 2013–14, with the aim of completing at least four tests per species. Investigations will continue into the reduced sensitivity of *M. macleayi* to U and reducing the nutrients used in the algal test.

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Ecotoxicological assessment of distillate from the Ranger brine concentrator plant

AJ Harford, A Hogan, C Pease, M Trenfield & RA van Dam

Background

The increasing process water inventory at the Ranger uranium mine has become a major operational issue for Energy Resources of Australia Ltd (ERA). Following an assessment of potential technology options, ERA decided that brine concentration was the most viable route to reduce the inventory. A brine concentrator would produce large volumes of a purified water product (distillate) and a waste stream containing the salts present in the process water (brine concentrate). The distillate will be released (following approval) into the environment via a yet-to-be determined method, while the brine concentrate will be returned to the tailings storage facility (TSF) or, eventually, directly injected into the bottom of Pit 3. In 2011, Rio Tinto – Technology and Innovation (RT-TI, Bundoora, Victoria) were engaged by ERA to conduct trials on a pilot-scale brine concentrator plant. Two key aims of the RT-TI trial were to (i) demonstrate that the distillate does not pose risks to operator health or the environment, and (ii) provide data to assist with designing water management and disposal systems. To assist with addressing the aquatic environment protection aspect, ***eriss*** undertook a comprehensive toxicity testing programme of the pilot plant distillate (Harford et al. 2013). The aims of the toxicity test work were to: (i) detect and quantify any residual toxicity of the pilot distillate and, (ii) in the event that effects were observed, to identify the toxic constituent(s) of the distillate.

Five tropical freshwater species (*Chlorella* sp. (green algae), *Lemna aequinoctialis* (duckweed), *Hydra viridissima* (green hydra), *Moinodaphnia macleayi* (cladoceran) and *Mogurnda mogurnda* (fish) were exposed to a limited concentration range of the distillate (0, 25, 50 and 100%). The distillate was toxic to only *H. viridissima* (50-100% effect when exposed to 100% distillate). A series of experiments demonstrated that the effect was not due to residual ammonia (~1 mg L^{-1} N) or trace organics, and unlikely to be due to elevated manganese (Mn; 130 – 230 µg L^{-1}). In contrast, addition of calcium (Ca), sodium (Na) and potassium (K) (at 0.5, 1.0 and 0.4 mg L^{-1}, respectively) resulted in 100% recovery of *H. viridissima* population growth rate. This indicated that ion deficiency must be considered as a potential stressor in risk/impact assessments of the discharge of purified waste waters, and that such waters may need to be supplemented with the deficient ions to reduce environmental impacts (Harford et al. 2013). Further assessment on the likelihood of Mn toxicity indicated that the residual Mn concentrations in the distillate were at levels that could inhibit the growth of *H. viridissima*, and further data are needed to assess the risk of Mn in low pH, soft waters (see paper Ecotoxicological assessment of Manganese page 7).

The full-scale brine concentrator plant at Ranger was commissioned in September 2013 and the electrical conductivity of the distillate stabilised in early October 2013. The aims of the present study were to assess the toxicity of a distillate from the full-scale brine concentrator plant, and to identify the cause/s of any observed effects.

Methods

On 7 October 2013, following the stabilisation of distillate water quality, samples of the distillate were collected in glass and plastic containers, including samples for Volatile and semi-Volatile Organic Compound analysis (VOCs and sVOCs measured by Gas Chromatograph-Mass Spectrometry, GC-MS). The samples were transported to the Darwin laboratory and immediately measured for dissolved oxygen, pH and electrical conductivity (EC). The plastic containers were sub-sampled for a full-suite of metals and major ions.

Five tropical freshwater species were used to test the toxicity of the distillate, using the standard protocols described in Reithmuller et al. (2003; Table 1). The exposure regimes differed for the five species, thus:

1. *Hydra viridissima* and *M. macleayi*, were exposed to a limited concentration range of the distillate diluted in Magela Creek Water (MCW; 0, 25, 50 and 100%). Additionally, an undiluted sample of distillate was amended by adding Ca, Na and K at 0.5, 1.0 and 0.4 mg L^{-1}, respectively (termed “100% amended”). The concentrations of the added Ca, Na and K are representative of those measured in Magela Creek and were added to determine whether the adverse effects observed in the pilot-plant study would now be eliminated. Magnesium was not added because the distillate from the pilot-plant contained residual Mg that was similar to concentrations measured in the creek.
2. The *Chlorella* sp., *L. aequinoctialis* and *M. mogurnda*, were all initially exposed to just 0 and 100% distillate because these species tolerated the pilot-plant distillate and effects were not expected. However, toxicity observed in the 100% distillate treatment resulted in repetition of the toxicity tests for *L. aequinoctialis* (using 0, 25, 50 and 100%) and *M. mogurnda* (0, 50 and 100%). Both of the repeated toxicity tests included a 100% amended treatment, as used for the other three test species.

Table 1 Details of each of the toxicity tests undertaken to assess the distillate from the brine concentrator.

Test organism	Acute/ Chronic	Test code	Date	Treatments tested (% Distillate)
Chlorella sp. (unicellular alga)	Chronic	1356G	08/10/13	0, 100
Lemna aeqinoctialis (duckweed)	Chronic	1355L	08/10/13	0, 100
	Chronic	1362L	14/10/13	0, 25, 50, 100, 100 (amended)[1]
Hydra viridissima (green hydra)	Chronic	1352B	08/10/13	0, 25, 50, 100, 100 (amended)
Moinodaphnia macleayi (cladoceran)	Chronic	1353D	10/10/13	0, 25, 50, 100, 100 (amended)
Mogurnda mogurnda (fish)	Acute	1354E	25/10/13	0, 100
	Acute	1364E	1/11/13	0, 50, 100, 100 (amended)

1 Amended: undiluted distillate with addition of 0.5, 1.0 and 0.4 mg L^{-1} (nominal concentrations) Ca, Na and K, respectively.

Results

The chemical analyses showed that the distillate sample from the full-scale plant was a highly-purified water and contained less metals and major ions compared to the sample from the pilot plant (Table 2). The EC of the distillate was 3 µS cm^{-1}, all major ions were below detection limits and the ammonia concentration was 0.25 mg L^{-1}. Manganese, which was at 130 - 230 µg L^{-1} in the pilot-plant sample, was measured at 7 µg L^{-1} and U was 0.05 µg L^{-1}. The only other measured inorganic elements of note were Al and B, which were 3 and 13 μg L^{-1}, respectively (Table 2). All sVOC and VOCs were below detection limits (data not shown).

The effects of the distillate on the five freshwater species are shown in Figure 1. The full-scale-plant distillate was higher in toxicity compared to the pilot-plant distillate, but this was most likely due to the lower concentrations of major ions in the former distillate. All species displayed some degree of adverse effects in 100% distillate. *M. mogurnda* was the most tolerant species with an effect of 7.4% that was not statistically different from the control. However, the effect of the 100% distillate was not as apparent in the repeated test due to a lower 90% survival of the controls in this test. The first test produced a statistically significant effect of 17%. Addition of the major ions improved the survival of the fish to a rate comparable with the controls. The growth rate of *Chlorella* sp. was reduced by a statistically significant 8%. *M. macleayi* and *L. aequinoctialis* were equally sensitive to the 100% distillate with 71 and 73% reductions in reproduction and growth, respectively. However, *L. aequinoctialis* responded better to the addition of the major ions returning to a growth rate similar to controls. The reproduction of *M. macleayi* improved with the major ion addition but was still 40% lower than the controls. As seen for the pilot-plant distillate, *H. viridissima*, did not grow in the 100% distillate and all organisms exposed to the water died in 96 h. The addition of the major ions resulted in 82% recovery (Figure 1), but this was lower than the for the pilot-plant study where 100% recovery was observed (Harford et al. 2013).

Table 2 Selected measured chemicals in the distillate

Analyte	Detection limit	Pilot-plant 1st sample	Pilot-plant 2nd sample	Full-scale plant
pH	0.1	5.8	6.7	6.05
Electrical conductivity (μS cm^{-1})	1.0	17	12	3
DOC (mg L^{-1})	0.1	0.6	NM [a]	0.6
Calcium (mg L^{-1})	0.1	0.11	<0.1	<0.1
Magnesium (mg L^{-1})	0.1	0.6	0.4	<0.1
Sodium (mg L^{-1})	0.1	<0.1	<0.1	<0.1
Potassium (mg L^{-1})	0.1	<0.1	<0.1	<0.1
Biocarbonate (mg L^{-1} $CaCO_3$)	1	7	6	<1
Ammonia (mg L^{-1} NH_3-N)	0.005	0.7	0.8	0.25
Aluminium (μg L^{-1})	0.1	18	23	3
Manganese (μg L^{-1})	0.005	230	130	7
Boron (μg L^{-1})	0.5	100	88	13
Uranium (μg L^{-1})	0.001	1.1	1.5	0.05

a NM: Not measured

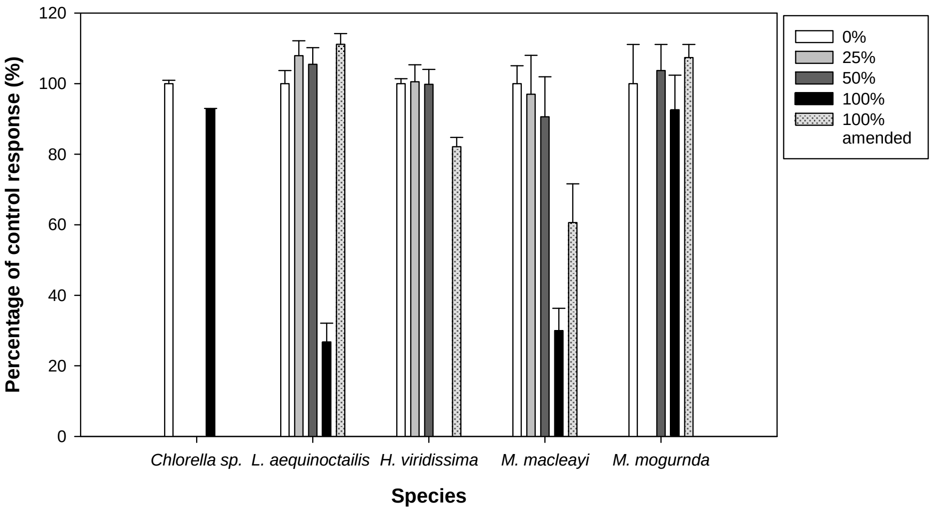

Figure 1 Toxicity of the full-scale-plant distillate to five local freshwater species. Where a repeated test was conducted, the first test result is not shown. Control responses were (mean ± se); 1.7 ± 0.02 dbl/d for *Chlorella* sp.; 0.32 ± 0.0 cm^2 d^{-1} for *L. aequinoctialis*; 0.31 ± 0.0 for *H. viridissima*; 29.7 ± 1.5 neonates/adult for *M. macleayi*; and 90 ± 1.0% survival for *M. mogurnda*.

Conclusion

The toxicity of the full-scale-plant distillate was higher than that of the pilot-plant product but the results were not unexpected due to the higher purity of the former water. It is possible the remaining effects in the 100% amended waters were due to the concentration of Mg not being sufficient for hydra and cladoceran growth and reproduction. The major ion concentrations in the amended waters were similar to MCW except for Mg, which was not added to the treatments because this was not required during the pilot-plant study (Table 3). Nonetheless, adding Ca, Na and K resulted in markedly improved performance of the organisms and indicates that a major ion deficiency is the primary cause of effects in the distillate. In light of these results, discussions need to be held with stakeholders regarding the management of the distillate dishcharges, so that ion deficiency impacts do not lead to adverse environmental consequences.

Table 3 Comparison of major ion composition in Magela Creek water and amended distillate treatments

Major ion	MCW	Hydra/Lemna	Cladoceran
Calcium	0.2	0.4	0.4
Sodium	1.3	1	1
Potassium	0.2	0.4	0.4
Magnesium	1.1	<0.1	<0.1

References

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Calculating annual solute loads in Magela and Gulungul creeks

K. Turner, A Mackay & W Erskine

Background

The Supervising Scientist Division (SSD) undertakes comprehensive water quality monitoring to ensure the protection of the Ramsar-listed Magela Creek wetlands and the people living semi-traditional livelihoods, downstream of the Ranger uranium mine (RUM). Starting in the 2010–11 wet season, continuous monitoring, incorporating event-based collection of water samples (fully automated using automatic sampling units), replaced grab sampling as the primary method for collection of water samples for chemistry analysis in Magela and Gulungul creeks. As well as providing primary water quality data (electrical conductivity, turbidity and pH) for impact assessment and community assurance purposes, the continuous monitoring data, along with associated chemistry and stream discharge data, have been used to derive annual solute loads for Magela and Gulungul creeks. This enables temporal and spatial comparison of annual solute loads transported by each creek, allowing an assessment to be made of the annual performance of the site's mine water management system.

The potential sources of mine-derived solutes to Magela Creek include: i) managed and passive surface water discharge via the Coonjimba (monitored at ERA's RP1 site) and Corridor Creek (monitored at ERA's GC2 site) catchments; and/or ii) remobilisation of any residual solutes in the Land Application Areas (LAAs) that lie between the mine site and Magela 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). Potential sources 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. While solute loads have been presented over a number of years for Magela Creek (Turner et al. 2012) the Gulungul Creek loads were presented for the first time by Mackay & Erskine (2013).

The calculation of a robust and internally consistent solute budget for Magela Creek has been attempted in previous years, however this depends on reliable measurement of discharge at sites on Magela and Gulungul creeks, upstream and downstream of the mine, as well as at sites on each of the small, point-source tributaries including ERA's RP1 and GC2 sites. Development of robust rating curves (and hence reliable discharge measurement) for RP1 and GC2 has proven to be difficult in the past. This was due to the multiple release methods from RP1 (passive flow over the weir, manual pumping/syphoning over the weir or manual pumping via a pipeline to ERA's Magela01 site) and the unconstrained nature of the cross section (much of the flow bypasses the control structure) and backwater effects at GC2. During the 2013–14 wet season ERA's telemetry team will be investigating a method of in situ discharge measurement using permanently deployed acoustic Dopplers. SSD will assist ERA with this investigation and with all other aspects of stream discharge measurement, as required.

Methods

Continuous in situ EC and water level data have been measured in both Magela and Gulungul creeks since the 2005–06 wet season. Surface water grab samples have also been collected routinely from both creeks at sites upstream and downstream of the mine. These samples were analysed for a range of solutes including uranium (U), manganese (Mn), magnesium (Mg) and sulphate (SO_4). 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 and Gulungul creeks, where significant relationships between EC and Mg (dissociated from $MgSO_4$) concentration have been reported (Turner et al. 2012; Mackay & Erskine 2013). These relationships have been used to estimate continuous Mg concentrations from the continuous EC record measured in each creek.

The annual solute loads are calculated by combining the estimated Mg concentration data with discharge data for each creek 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 wet season can be calculated.

Progress to date

The most recent updates of the Magela and Gulungul creeks Mg loads are shown in Tables 1 and 2.. The data illustrates that the mine-related inputs of Mg (inferred by the difference between upstream and downstream sites) into Magela Creek are much higher than the diffuse inputs to Gulungul Creek.

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

Water year	Gauging stations		Difference^
	Upstream	Downstream	
2005–06	168	377	209
2006–07[#]	135	510	375
2007–08	100	364	264
2008–09	75	173	98
2009–10	130	277	147
2010–11	180	399	219
2011–12	160	268	108
Mean	135	338	203

^ Difference is calculated by subtracting the upstream load from the downstream load.
[#] Data uncertainty is high due to instrument damage during the March flood.

Table 2 Estimated Mg loads (t/yr) in Gulungul Creek

Water year	Gauging stations		Difference^
	Upstream	Downstream	
2005–06	32	53	21
2006–07	29	72	43
2007–08	19	35	16
2008–09	10	17	7
2009–10	17	36	19
2010–11	37	67	30
2011–12	30	41	11
Mean	25	46	21

^ Difference is calculated by subtracting the GCUS load from the GCDS load.

The solute loads for Magela and Gulungul creeks have not been calculated for the 2012–13 wet season. Key staff from the Hydrological, Geomorphological and Chemical Processes (HGCP) group and the Jabiru Field Station have been concentrating their effort on the Ranger Trial Landform (TLF) project for most of the 2013 dry season, which is when the loads are usually calculated and reported. These resources were redistributed to ensure progression of the TLF data analysis and interpretation, as well as to carry out a technical review of the TLF monitoring stations, subsequent associated field work and upgrades. Effort was also prioritised to ensure ongoing management and preparation of the routine water chemistry monitoring programme for the 2013–14 wet season, including an upgrade to the Ngarradj Creek monitoring site (enabling automated event-based sampling and improving satellite communication) as well as an extensive review of quality control data collected over the last four years.

Steps for completion

It is anticipated that the creek discharge data, which is required for the load calculations, will be validated by the end of 2013 and on completion of this, the solute loads will be calculated and reported. The review of the quality control data will be published in an internal report in early 2014 and this will include recommendations for sample collection method refinement, a review of the relationship between dissolved and total U in the creeks and a new method for management of SSD's water chemistry monitoring data by utilising the new Envirosys database. The Magela catchment solute budget work will remain on hold until ERA complete their investigation into the use of new acoustic Doppler technology at their RP1 and GC2 discharge monitoring sites.

References

Mackay A & Erskine WD 2013. Calculating annual solute loads in Gulungul Creek. In ***eriss*** *research summary 2011–2012*, eds van Dam RA, Webb A & Parker SM, Supervising Scientist Report 204, Supervising Scientist, Darwin NT, 49–54.

Turner K, Jones DR & Erskine WD 2012. Recent developments in Magela Creek solute loads. In: ***eriss*** *research summary 2010–2011*. eds Jones DR & Webb A, Supervising Scientist Report 203, Supervising Scientist, Darwin NT, 53–58.

Results of the stream monitoring program in Magela Creek and Gulungul Creek catchments, 2012–13

CL Humphrey & A Bollhöfer

Progress under this KKN for the stream monitoring programme in the Magela Creek and Gulungul Creek catchments is reported by way of (i) results of the routine monitoring programme conducted for the 2012–13 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 programmes 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 programme incorporates chemical, physical and biological components.

The techniques and 'indicators' used in the monitoring programme 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 programme has evolved through time as technologies (e.g. continuous physicochemical monitoring using data sondes and telemetry) have evolved, and improved methodologies for biological assessment (e.g. *in situ* monitoring using snails) have been developed under the SSD research programme.

The results from the stream chemical and biological monitoring programme for 2012–13 are summarised below.

Chemical and physical monitoring of Magela Creek

A Sinclair

Flow was first recorded at the Magela Creek upstream and downstream monitoring stations on 7 January 2013. Due to the below average rainfall this wet season (Figure 1) the Magela Creek flow remained very low (<25 m3/s) with the exception of a flood event occurring on the 31 March 2013.

On 17 January 2013, a localised high intensity rainfall event in the vicinity of the downstream monitoring station (81.8 mm at Jabiru airport and 40.4 mm at MG009) resulted in a rapid increase in flow at MCDW peaking at 14 m3/sec, in turn decreasing EC (Figure 2) and significantly increasing turbidity (Figure 3). This rainfall event was located primarily downstream of Pit 3 and the Pit 3 levee as ERA reported only 14 mm of rainfall was recorded at Pit 3. Thus the source of the turbidity is likely to be localised surface run-off and not erosion of the Pit 3 levee. Water quality parameters returned to previous levels within a few hours. The effects of this rainfall event were also observed at both the upstream and downstream monitoring sites in Gulungul Creek (see below).

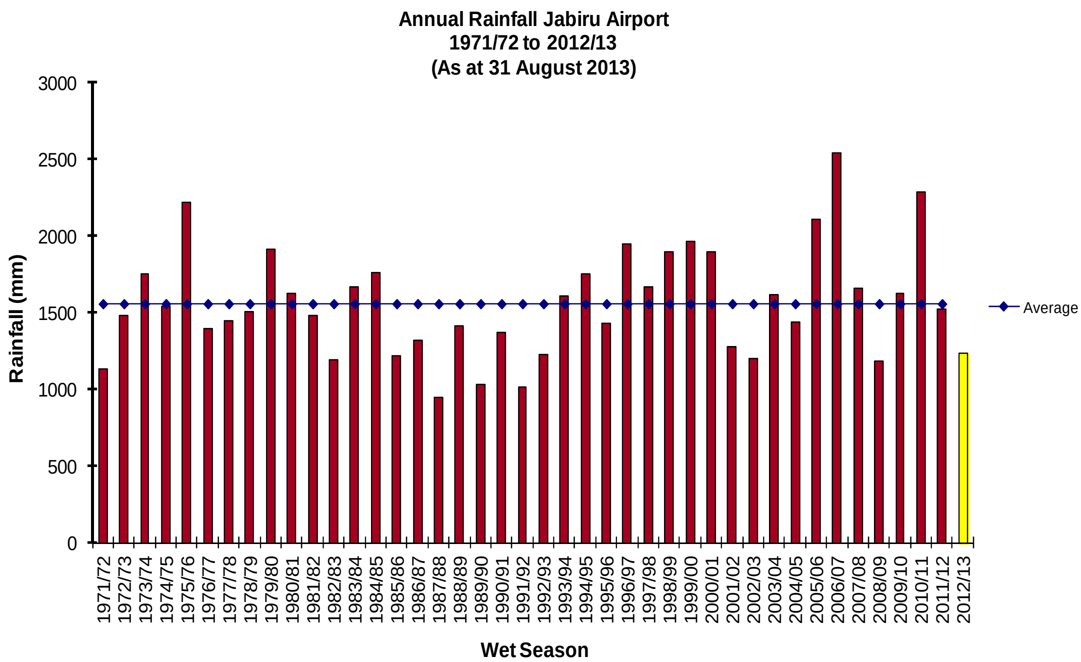

Figure 1 Annual rainfall Jabiru Airport 1971–72 to 2012–13 (data from Bureau of Meteorology).

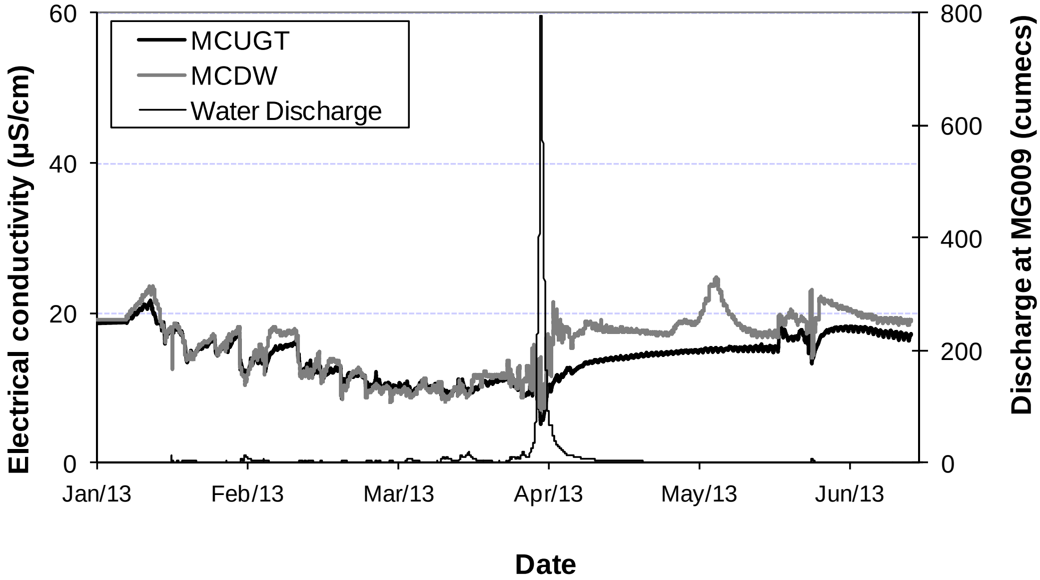

Figure 2 Continuous electrical conductivity and discharge in Magela Creek between January and June 2013.

On 30 and 31 March 2013, a low pressure system resulted in heavy rainfall over the Arnhem Land region. Jabiru Airport recorded a total rainfall of 240 mm over the two days and Magela Creek flow quickly rose and peaked at 800 m^3/s. During this period, water from RP1 with an EC of approximately 300 µS/cm was released into Coonjimba Billabong. This resulted in slightly increased EC levels at MCDW during the latter part of the flood event, as the creek flow receded and the water in the billabong discharged into the creek. However, due to the large dilution rates during this rainfall event, the EC did not exceed the focus trigger level of 21 µS/cm.

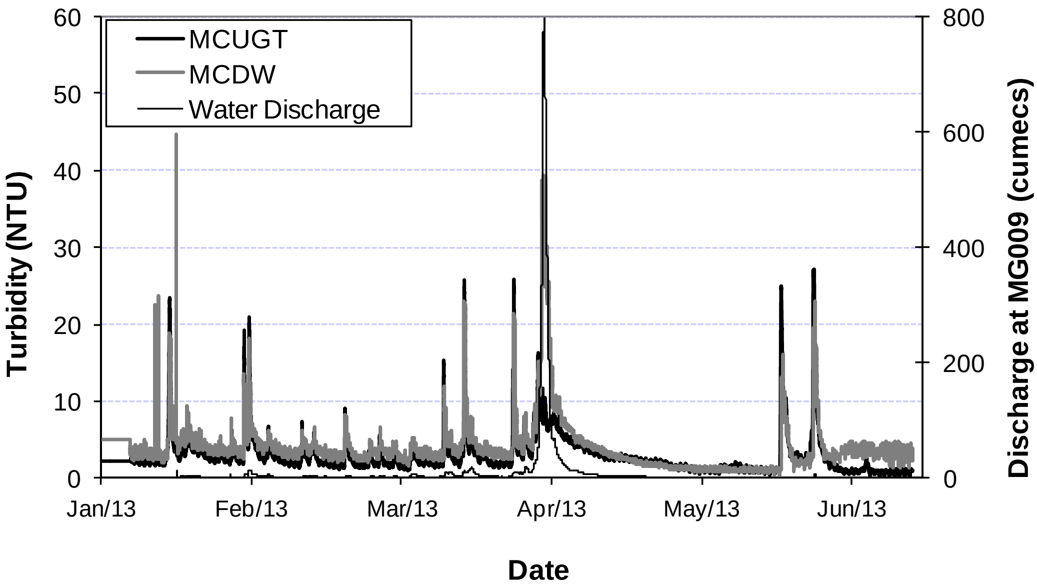

Figure 3 Continuous turbidity and discharge in Magela Creek between January and June 2013.

During early May the minesite released pond water treatment plant permeate (good quality water) into the Corridor Creek system. This input resulted in the outflow of water from Georgetown Billabong into Magela Creek. As the billabong waters typically have a higher EC than the creek waters during the dry season due to solute inputs from Corridor Creek and evapoconcentration, the outflow from Georgetown Billabong was detected at MCDW via an increase in EC to 25 µS/cm on 5 May 2013. This EC was just above the Focus trigger value of 21 µS/cm. The minesite stopped the discharge of permeate into Corridor Creek and in turn Georgetown Billabong ceased flowing into Magela Creek and a corresponding decrease in EC at MCDW was observed. A number of EC fluctuations were observed during May and June as a result of discrete rainfall events combined with very low flow in the creek.

During the wet season, the maximum total uranium concentration measured downstream from the Ranger mine was 0.05 µg/L. This value is approximately 1% of the local ecotoxicologically-derived limit of 6 µg/L for protection of aquatic ecosystems, and approximately 0.25% of the 20 µg/L guideline for potable water (Figure 4).

The maximum manganese concentration of 9.9 µg/L occurred in early April following the large rainfall event (Figure 5). This value is below the action trigger value of 11 µg/L for manganese. However, surface water flows on this date were less than 1 cumecs, noting that the manganese guideline trigger value only applies when surface water flows are greater than 5 cumecs.

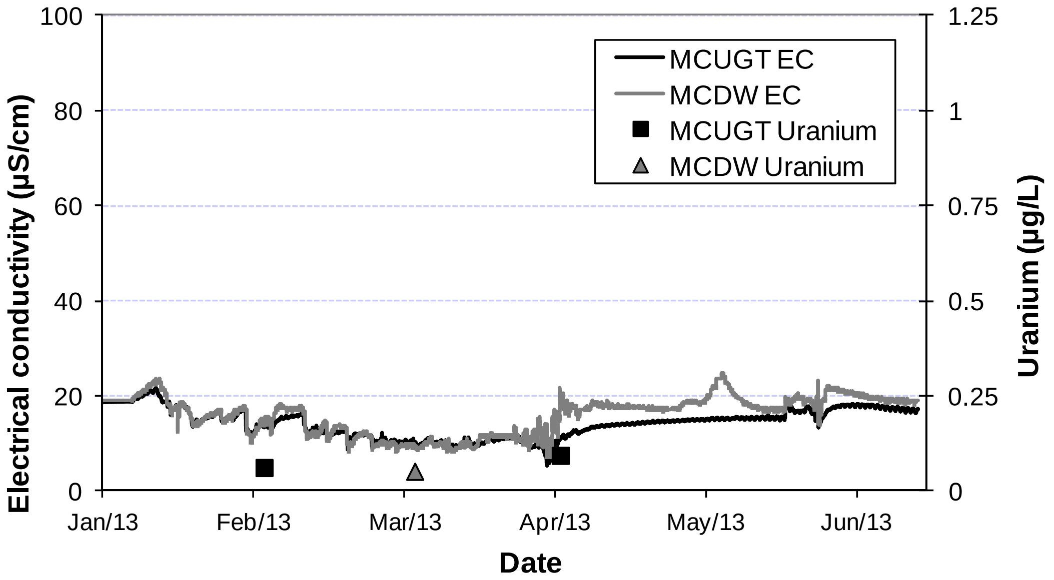

Figure 4 Total uranium concentrations in triggered samples and continuous electrical conductivity in Magela Creek between January and June 2013.

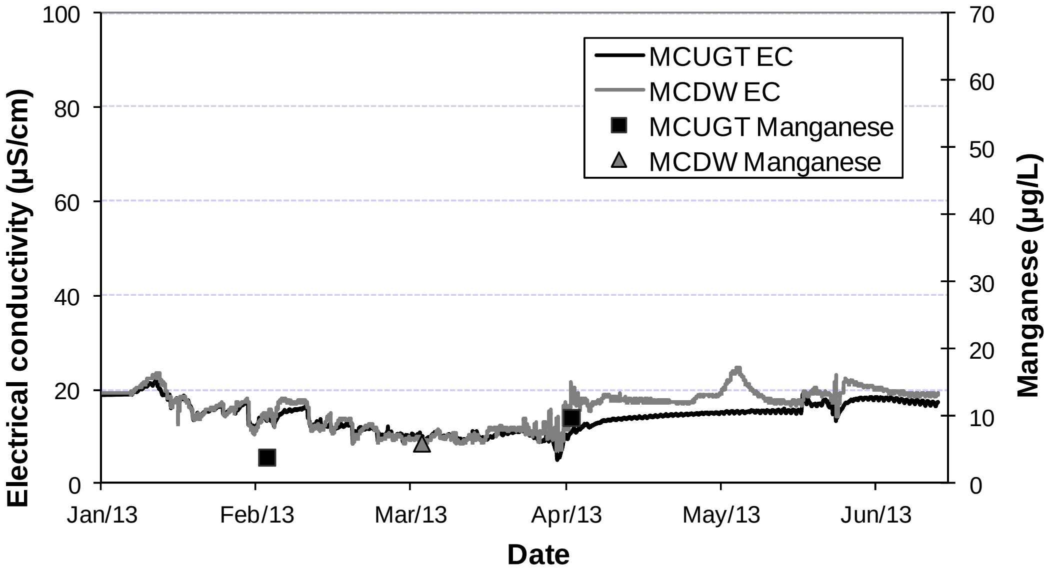

Figure 5 Total manganese concentrations in triggered samples and continuous electrical conductivity in Magela Creek between January and June 2013.

Magnesium and sulfate concentrations measured during 2012–13 were low reflecting the low mine site inputs to the creek this season (Figures 6 & 7). Automatic samples were not triggered for any EC peaks during the 2012–13 wet season as the EC did not exceed the 42 µS/cm (corresponding to 3 mg/L magnesium) guideline.

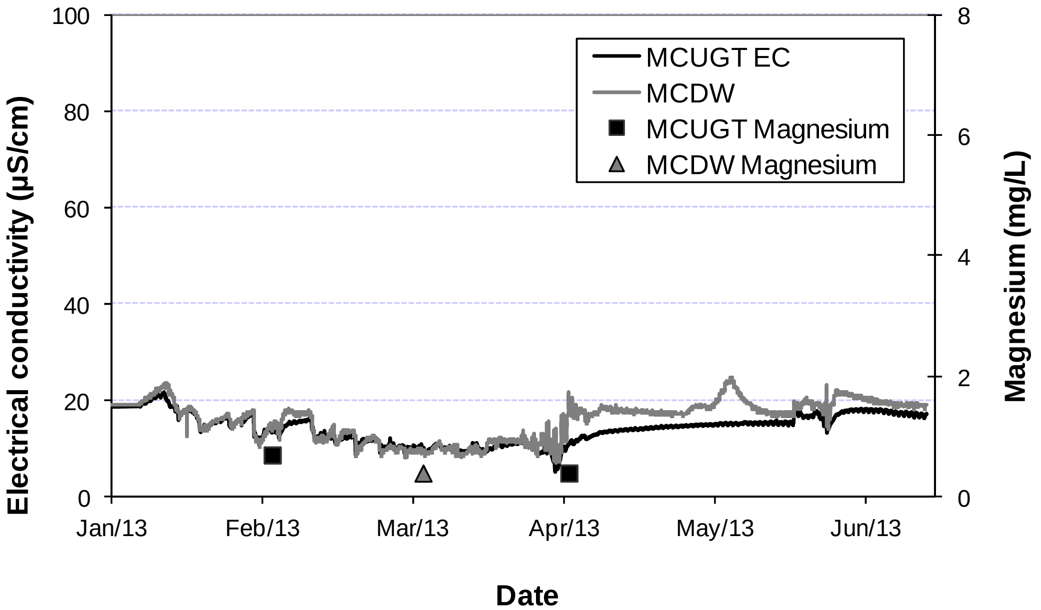

Figure 6 Total magnesium concentrations in triggered samples and continuous electrical conductivity in Magela Creek between January and June 2013.

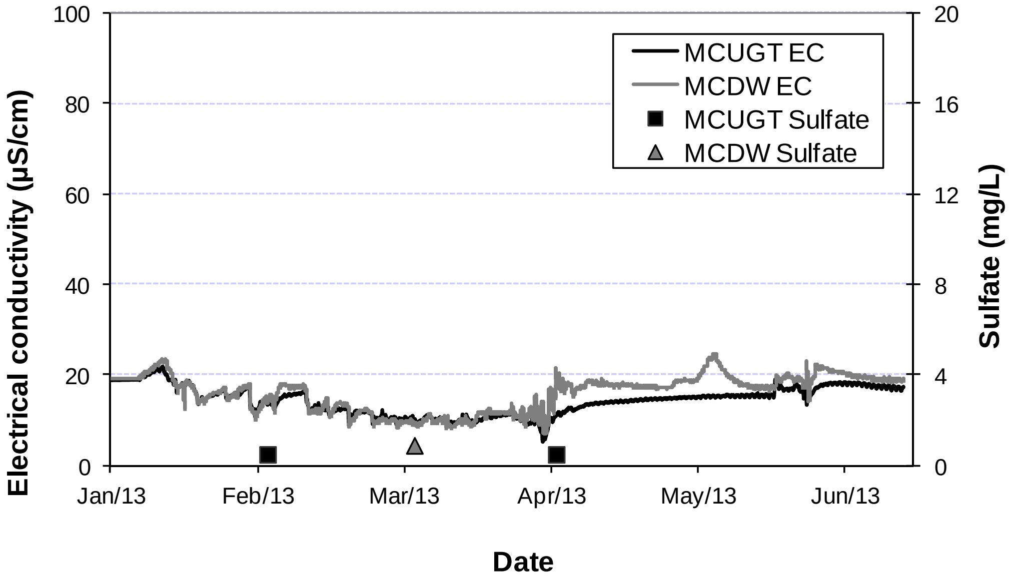

Figure 7 Total sulfate concentrations in triggered samples and continuous electrical conductivity in Magela Creek between January and June 2013.

Continuous monitoring continued until 12 June 2013 when the multi-probes were no longer submersed and could not be lowered any further. Cease to flow in Magela Creek was agreed by stakeholders on 1 July 2013.

Overall, the water quality measured in Magela Creek for the 2012–13 wet season showed lower EC at the downstream monitoring site compared to previous wet seasons due to very low rainfall, which resulted in reduced surface run-off and low creek flow conditions plus reduced volume of managed released waters from the minesite. The results indicate that, based on water quality, the aquatic environment in the creek has remained protected from mining activities (Figure 8).

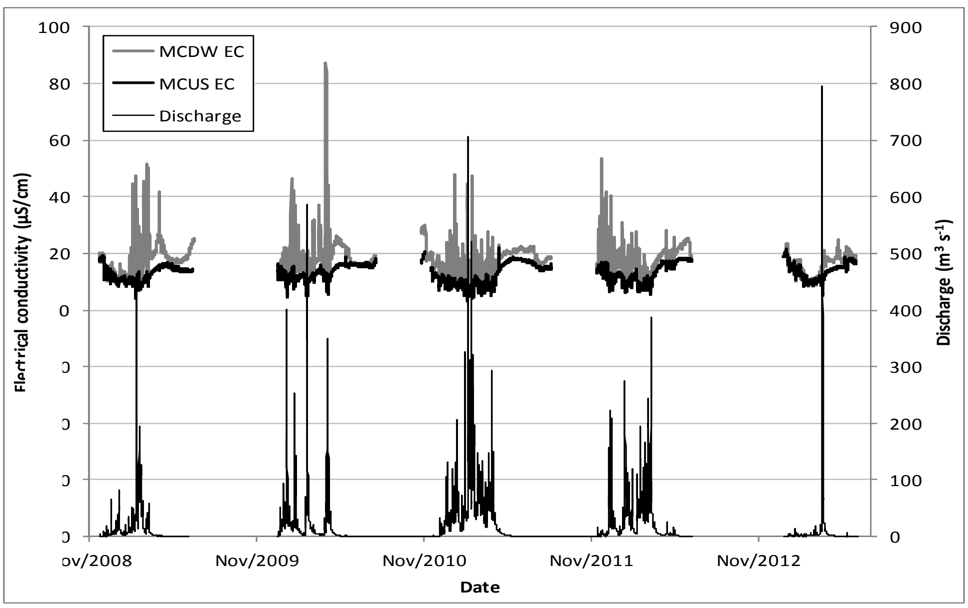

Figure 8 Continuous electrical conductivity and discharge (lower trace) in Magela Creek for each wet season between November 2008 and June 2013 (values averaged over a 90 minute period of measurement).

Radium in Magela Creek

Surface water samples are collected fortnightly from Magela Creek upstream and downstream of the Ranger mine. The fortnightly samples are combined to give monthly composite samples. Total radium-226 (^{226}Ra) is measured in these samples and results for the 2012–13 wet season can be compared with previous data ranging back to the 2001–02 wet season (Figure 9).

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. This has previously been discussed in the 2004–05 Supervising Scientist Annual Report.

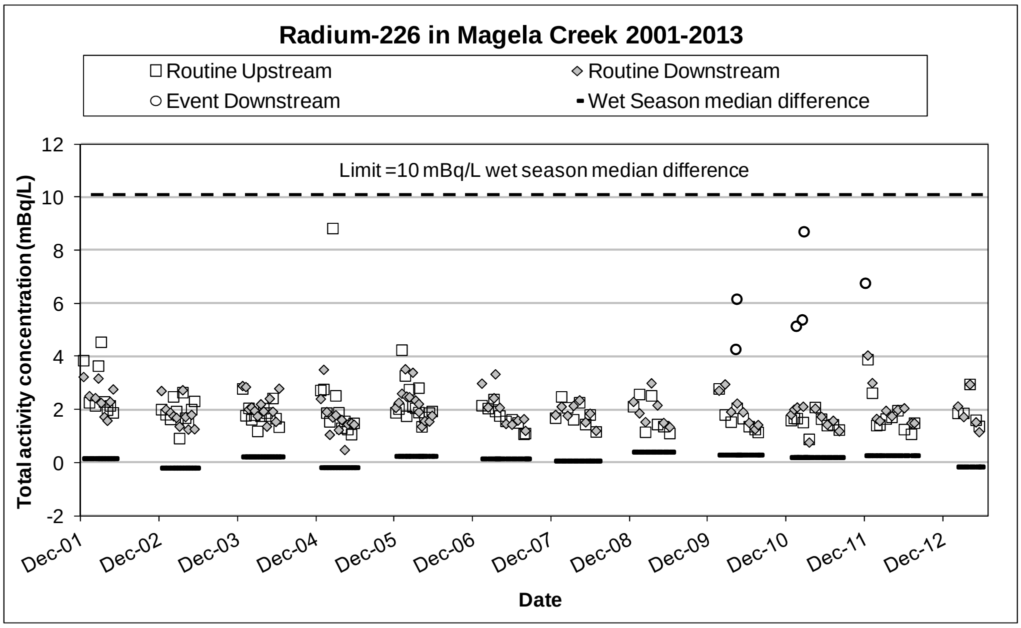

Figure 9 Radium-226 in Magela Creek 2001–2013.

The limit value for total ^{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 over one entire wet season. The median of the upstream ^{226}Ra data collected over the current wet season is subtracted from the median of the downstream data. This difference value, called the 'wet season median difference', quantifies any increase at the downstream site, and should not exceed 10 mBq/L.

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. Wet season median differences (shown by the horizontal lines in Figure 9) from 2001 to 2013 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 (2001–2013) is only 0.1 mBq/L.

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.I, together with the results from the routine radium analyses. The EC-triggered event data are not included in the calculation of the wet season median difference, because these EC events are short-lived and their impact on seasonal ^{226}Ra loads is very small.

There were no EC-triggered event samples collected during the 2012–13 wet season. The wet season median difference for the 2012–13 wet season is -0.1 mBq/L indicating a greater median value for the upstream monitoring site than for the downstream monitoring site.

Chemical and physical monitoring of Gulungul Creek

A Sinclair

Flow was first recorded at the Gulungul Creek upstream and downstream monitoring stations on 23 December 2012. Flow remained very low due to below average rainfall this wet season.

On 17 January 2013 Jabiru airport recorded a rainfall event of 81.8 mm. This appeared to have been a localised rainfall event resulting in increased EC and increased turbidity at both the upstream and downstream monitoring sites (Figures 1. and 2.). Water quality parameters returned to previous levels within a few hours.

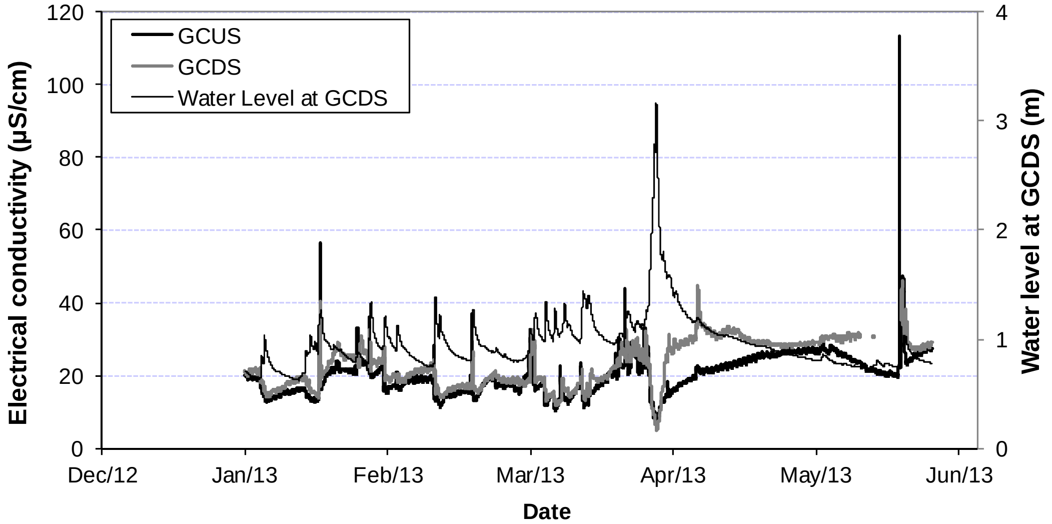

Figure 1 Electrical conductivity and water level in Gulungul Creek between December 2012 and June 2013.

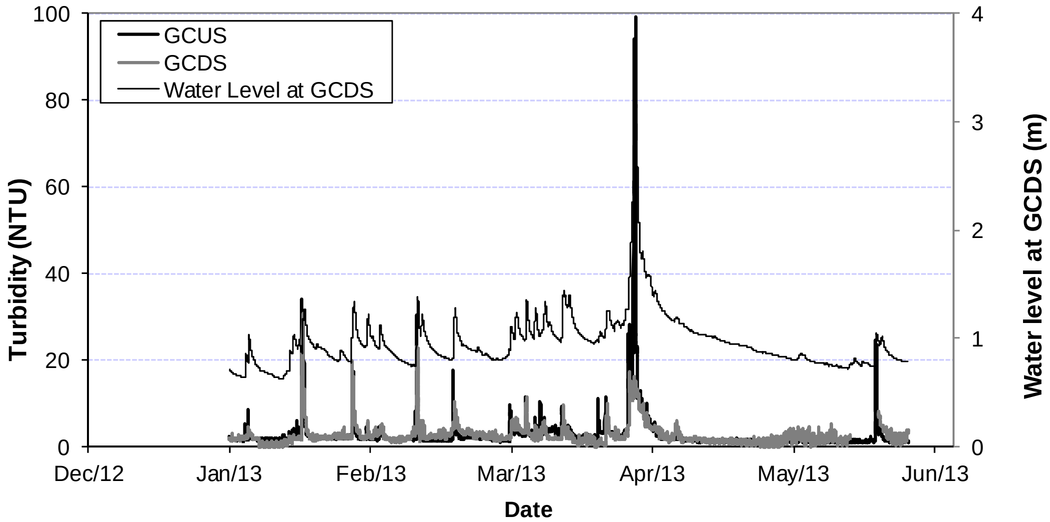

Figure 2 Turbidity and water level in Gulungul Creek between December 2012 and June 2013.

Continued low rainfall conditions resulted in low flow levels in the creek through most of February. The frequency of rainfall events increased in March which increased the flow levels in Gulungul Creek.

On 30 and 31 March 2013 a low pressure system resulted in significant rainfall over Arnhem Land. Jabiru Airport recorded total rainfall of 240 mm over the two days. Flow within Gulungul Creek quickly rose, with a corresponding decline in EC due to dilution with low salinity rainfall. EC increased in early April as flow levels decreased.

On 9 April 2013 an increase in EC coincided with a localised rainfall event. The EC at the downstream monitoring site peaked at 45 µS/cm and remained above 42 µS/cm for a period of 5 hours. Under the SSD ecotoxicology electrical conductivity-magnesium pulse framework, a 5 hour pulse duration would have an EC limit value of 581 µS/cm, thus with a peak of 45 µS/cm the downstream aquatic ecosystem was very unlikely to have been impacted by this event.

From mid April, recessional flow conditions became established within Gulungul Creek with sustained low flows and slowly rising and converging EC at upstream and downstream monitoring sites. During early to mid May, the monitoring multi-probe sensor at GCDS was no longer submersed due to very low flow levels.

A localised rainfall event in the upper Gulungul Creek catchment on 22 May 2013 flushed solutes into the creek resulting in a non-mine derived EC peak of 113 µS/cm at the upstream monitoring site, GCUS. These solutes were washed down the creek and progressively diluted, producing EC peaks of 76 and 46 µS/cm at GCMID and GCDS monitoring sites (Figure 3). The EC at the downstream monitoring site remained above 42 µS/cm for a period of 3.3 hours. Under the SSD ecotoxicology electrical conductivity-magnesium pulse framework, a 3.3 hour pulse duration would have an EC limit value of 1140 µS/cm, thus with a peak of 46 µS/cm, the downstream aquatic ecosystem was very unlikely to have been impacted by this natural event.

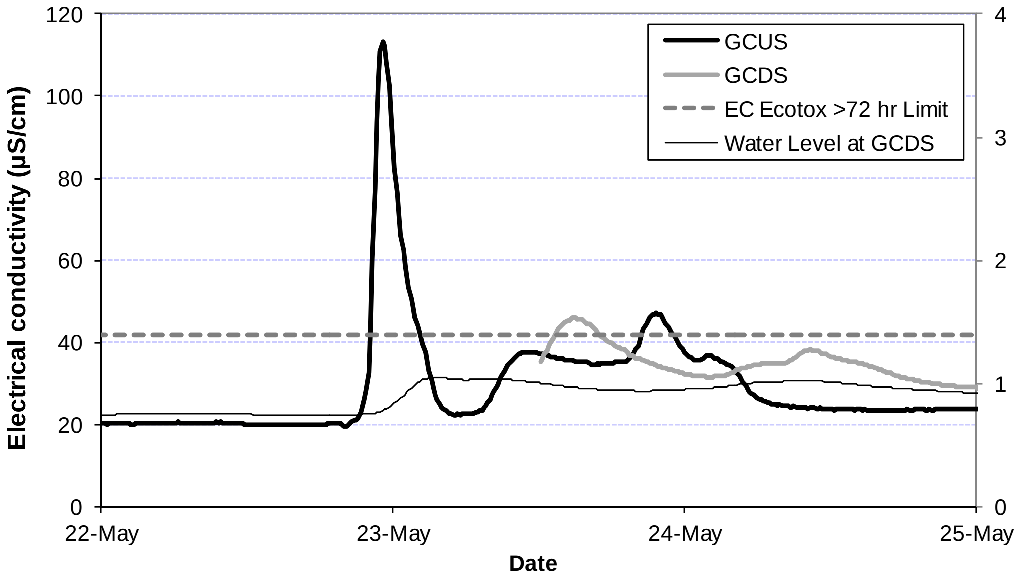

Figure 3 Electrical conductivity and water level in Gulungul Creek during 22–25 May 2013.

Continuous monitoring continued until 29 May 2013 when the multi-probes were no longer submersed and could not be lowered any further. Cease to flow in Gulungul Creek was agreed by stakeholders on 18 June 2013.

During the 2012–13 wet season, the maximum total uranium concentration of 0.45 µg/L (Figure 4) was measured at GCUS upstream from the Ranger mine. Manganese concentrations were also greatest at the upstream monitoring site (GCUS) with a maximum concentration of 22 µg/L recorded on 11 February 2013 (Figure 5).

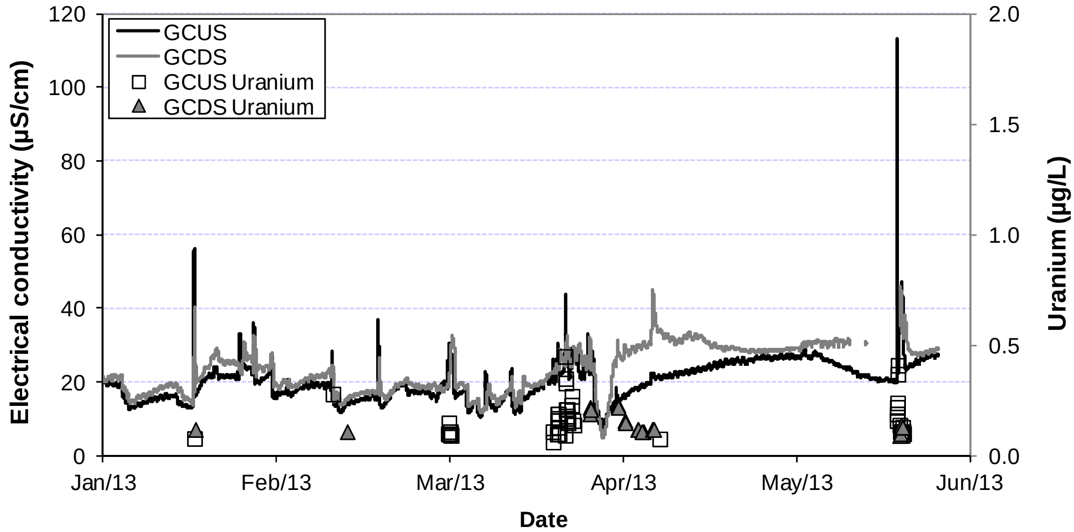

Figure 4 Electrical conductivity and total uranium concentrations in Gulungul Creek between December 2012 and June 2013.

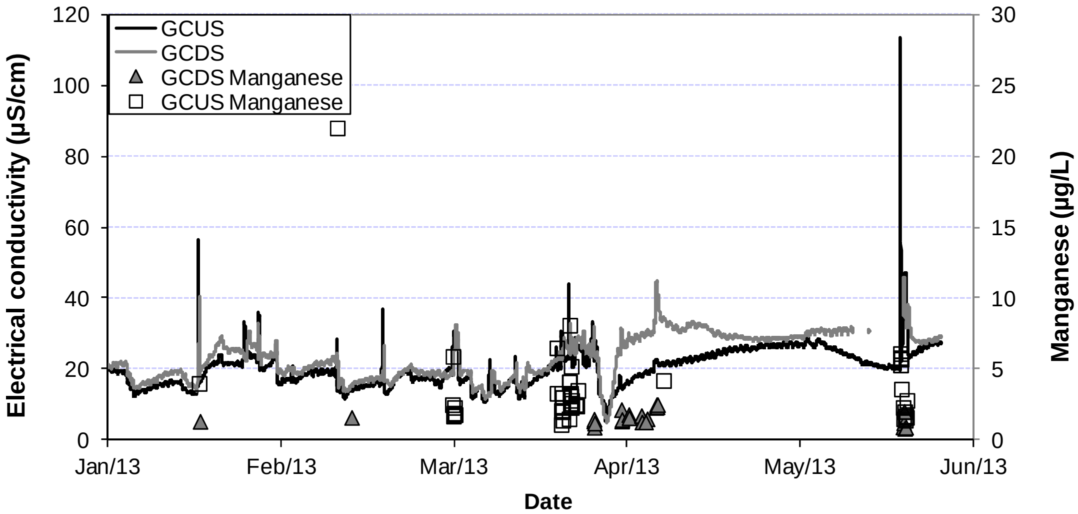

Figure 5 Electrical conductivity and total manganese concentrations in Gulungul Creek between December 2012 and June 2013.

Magnesium concentrations closely followed the EC continuous monitoring trace (Figure 6). The highest recorded concentrations occurred at GCUS monitoring site, upstream of the minesite with a peak of 9.3 mg/L on 22 May 2013. At the downstream monitoring site the concentrations of magnesium remained below 2.6 mg/L throughout the wet season.

Sulfate concentrations were generally low. The highest concentrations of 5 mg/L were observed at GCDS on 9 April 2013 (Figure 7).

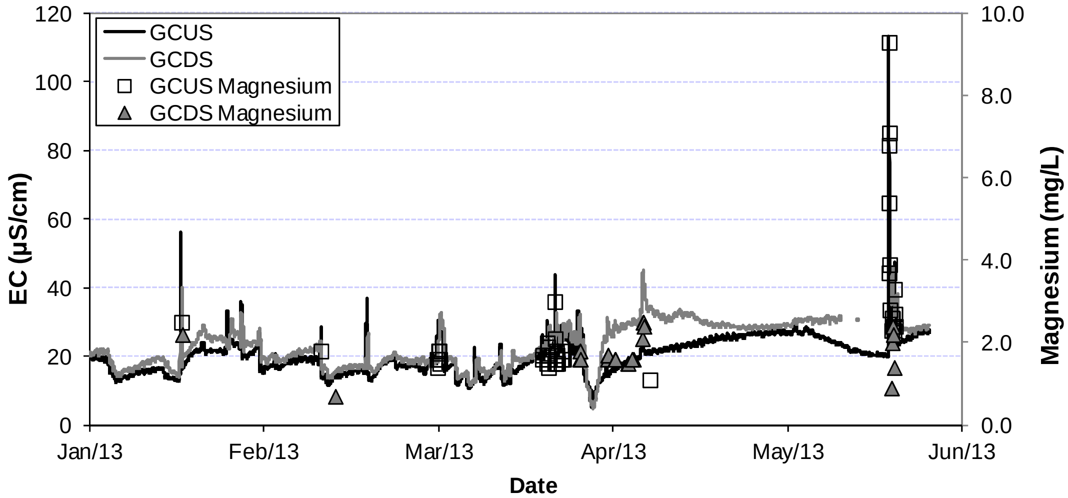

Figure 6 Electrical conductivity and total magnesium concentrations in Gulungul Creek between December 2012 and June 2013.

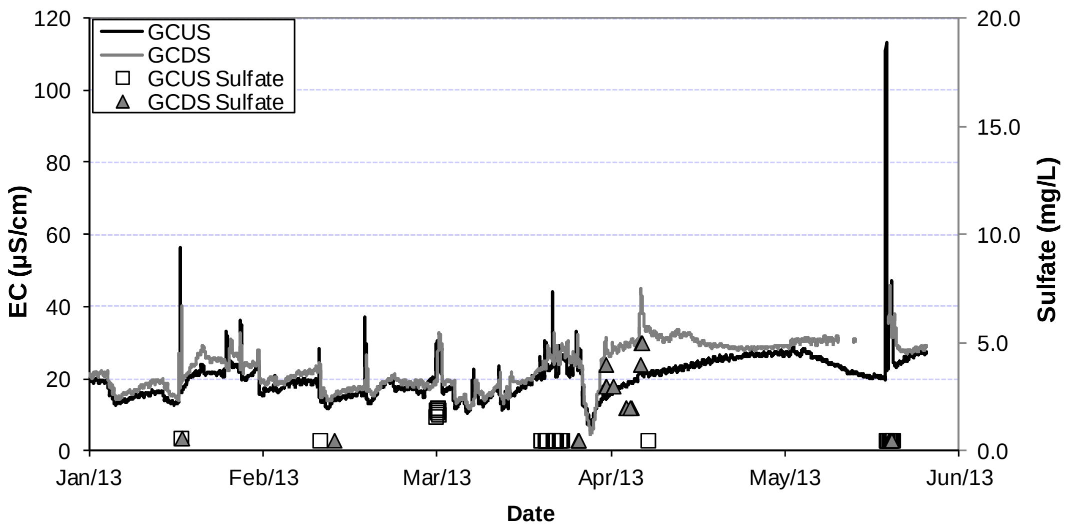

Figure 7 Electrical conductivity and total sulfate concentrations in Gulungul Creek between December 2012 and June 2013.

Overall, the water quality measured in Gulungul Creek for the 2012–13 wet season showed greater fluctuation in EC at the upstream monitoring site compared to previous wet seasons. This was due to very low overall rainfall and the effects from small scale rain events within the Gulungul catchment area, which resulted in localised surface runoff and influx of solutes into the creek. The results for the downstream monitoring site are comparable to previous years and indicate that, based on water quality, the aquatic environment in the creek has remained protected from mining activities (Figure 8).

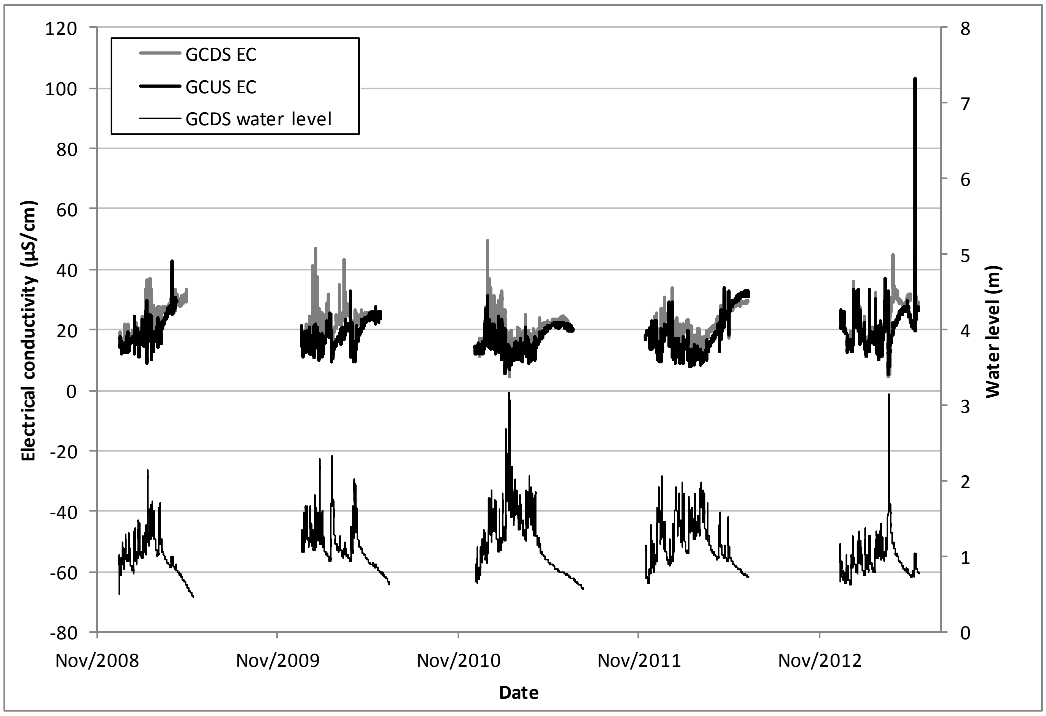

Figure 8 Electrical conductivity measurements and discharge (lower trace) in Gulungul Creek between November 2008 and July 2013 (values averaged over a 1 hour period of measurement).

Toxicity monitoring in Magela and Gulungul Creeks

CL Humphrey, M Ellis & J Hanley

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 h) exposure period. A full protocol for the testing procedure is provided in Supervising Scientist Division (2011). 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 (Figure 1A). 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 2006–07 wet season, an in situ testing method commenced, in which test animals are placed in floating (flow-through) containers located in the creek itself (see Supervising Scientist (2008), section 3.2, for details). Thus for the 2006–07 and 2007–08 wet seasons, creekside and in situ testing were conducted in parallel, to evaluate the effectiveness of the in situ method. For current data analyses, creekside data up to and including the 2005–06 wet season and in situ data from the 2006–07 wet season onward (Figure 1) are combined. The most recent refinement to this programme has been the extension of toxicity monitoring to Gulungul Creek, with testing commencing in the 2009–10 wet season (Figure 1B).

The first of eight tests in Magela Creek commenced on 10 January 2013, six days after the establishment of continuous flow in the creek. The following week, on 17 January 2013 the first of the seven Gulungul tests commenced. A combined total of 15 tests were completed over 15 weeks, alternating weekly between both creeks. The final test was completed in Magela Creek on 22 April 2013. For the fifth and sixth Gulungul Creek tests, data arising from one of the two duplicate floating containers deployed at the upstream site were deemed invalid due to container misalignment and subsequent restricted through-flow of creek waters. As such, results from the remaining duplicate only were used for subsequent statistical analyses for these two tests. Upstream and downstream egg production and difference values for both creeks are displayed in Figure 1B.

A marked increase in the mean number of eggs produced during each test for both sites of Magela and Gulungul Creeks was observed over the 2012–13 wet season (Figure 1B). Similar high egg production had previously only been observed in the first three tests of the 2006–07 wet season (Figure 1B). A significant factor contributing to this increase in egg production appears to be a new and more effective culturing regime for the snails at the laboratory aquaculture facility. The potential influence of snail husbandry on egg production and wet season toxicity monitoring results is discussed below and in more detail in Humphrey & Ellis (2014), an accompanying paper in this research summary.

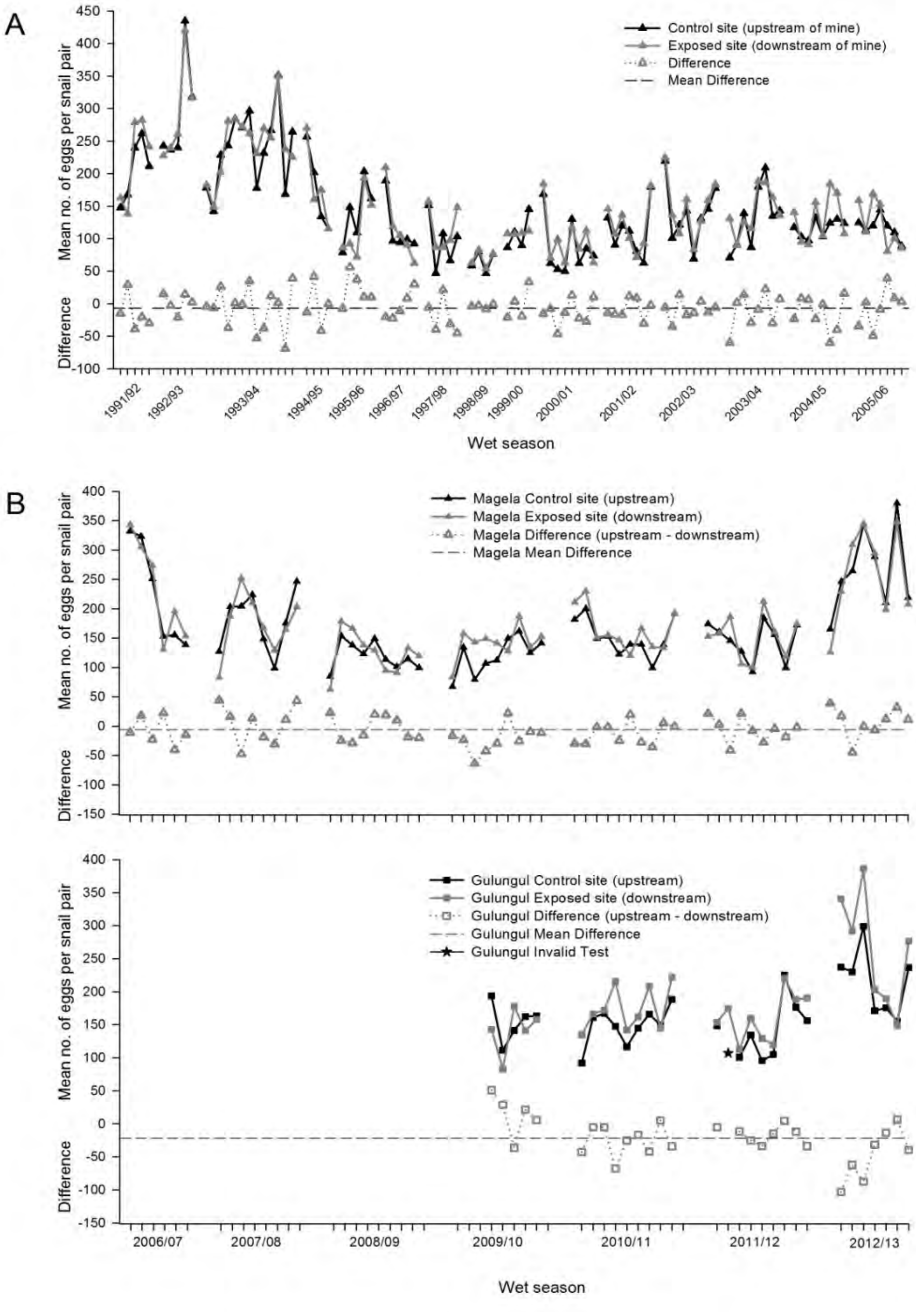

Figure 1 Time-series of snail egg production data from toxicity monitoring tests conducted in Magela Creek using A: creekside tests, and B: in situ tests with Gulungul tests commencing in 2009–10.

Analysis of 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. This Before-After Control-Impact Paired (BACIP) design, with Analysis of Variance (ANOVA) testing, is described further in Supervising Scientist Division (2011).

Magela Creek

The positive mean difference value for the eight 2012–13 wet season tests, of 7.5, indicates higher upstream egg production than downstream, contrasts with the historical trend of greater downstream egg production (mean difference value across all wet seasons of -8.0). The only previous period for which a positive mean difference value was observed was for the 1995–96 wet season. ANOVA results for the 2012–13 wet season, together with results from previous wet season scenarios, are displayed in Table 1. The significant differences observed in previous years, associated with particularly high egg production at the downstream site relative to the upstream site in the 2009–10 wet season, and to a lesser extent the 2010–11 wet season (Figure 1), are discussed in the respective Supervising Scientist annual reports. No significant difference was observed between the difference values derived from the 2012–13 wet season and those from previous wet seasons, though the low ANOVA significance value (near the 5% level, $p = 0.083$) highlights the unusually higher upstream egg production compared to downstream observed in this year (Table 1).

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.

Before	After	Probability value (*P*)	Significance
All previous seasons	2009–10	0.043	at 5% level
All previous and following seasons	2009–10	0.040	at 5% level
All previous seasons	2010–11	0.436	NS
All previous seasons	2010–11 + 2009–10	0.043	at 5% level
All previous and following seasons	2010–11 + 2009–10	0.025	at 5% level
All previous seasons	2011–12	0.916	NS
All previous seasons	2012–13	0.083	NS

NS = not significant

Gulungul Creek

The mean difference value across all Gulungul Creek tests for 2012–13, of -47.4, continues the trend of greater egg production downstream reported in previous years. Notable for this wet season were the particularly high egg numbers observed downstream in the first three tests compared to numbers upstream, giving rise to difference values much lower than previously recorded (Figure 1B). While the wet season mean difference value is much lower than the running mean of -13.47, ANOVA testing found no significant difference between the 2012–13 difference values and those recorded in previous wet seasons ($p = 0.228$).

Apart from the primary Before/After factor and associated hypothesis, the particular two-factor ANOVA model used for toxicity monitoring also allows variation amongst years (or wet seasons) and among tests within a wet season to be estimated separately. The second 'Season' factor can be used to determine whether, within the Before and After periods, any set of difference values for a wet season are significantly different. For Gulungul Creek after both the 2011–12 and 2012–13 wet seasons, the season factor has been significant ($p = 0.005$ and 0.048 respectively), compared to Magela Creek where this factor has never been significant. A significant season factor does not in itself imply potential mine-related impact; in this (Gulungul) case, it highlights the high inter-annual variation observed in seasonal difference values, as shown in Figure 1B and as reported in Supervising Scientist (2012).

Conclusions

In Supervising Scientist (2010, 2011) (section 3.4 of both reports), the influence of water temperature and electrical conductivity on the snail egg laying response was described. In the companion paper by Humphrey & Ellis (2014) of this annual research summary, these same water quality variables, together with snail husbandry conditions, are considered in the context of explaining the lower (compared to upstream) egg production observed in Magela Creek downstream of Ranger, as well as the particularly high egg production observed in the first three tests in Gulungul Creek downstream of Ranger, in the 2012–13 wet season.

The analysis presented by Humphrey & Ellis (2014) highlights the different water temperature regimes prevailing in both creeks over the 2012–13 wet season: median water temperatures were typically between 30 and 32°C in Magela Creek but lower, between 27 and 30°C, in Gulungul Creek (Figure 1 of Humphrey & Ellis 2014). For these respective water temperature ranges, decreases and increases in snail egg production have been observed with increasing electrical conductivity (EC) over the range 10–30 µS/cm (see section 3.4 of the Supervising Scientist's annual report, 2011–2012). In both creeks, median water temperature was higher at the downstream monitoring sites across all tests. Thus, and in general, the lower Magela, but higher Gulungul, downstream egg production, compared to respective upstream sites, reflects the inhibitory and enhanced effects respectively, associated with the different water temperature regimes in each of the creeks (see Humphrey & Ellis 2014).

For only one toxicity monitoring test (the last Gulungul test, Figure 1) the median EC value was greater than 25 µS/cm, indicating for both creeks and across the 2012–13 wet season, very limited mine water inputs (Supervising Scientist 2013, section 3.2.3.1). Thus, the toxicity monitoring results for 2012–13 reflect patterns associated with natural water quality (water temperature, EC) conditions in both creeks. There was no evidence of mine-related effects upon snail egg production over the wet season.

References

Humphrey C & Ellis M 2014. Environmental factors associated with toxicity monitoring in Magela and Gulungul Creeks. In **eriss** *research summary 2012–2013* Supervising Scientist Divsion, Supervising Scientist Report 205, Darwin NT, 79–86.

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Bioaccumulation in freshwater mussels

A Bollhöfer, C Doering, P Medley, T Fox & L Da Costa

Introduction

Some metals and radionuclides bioaccumulate in aquatic biota, in particular freshwater mussels. Therefore, it is essential to check that food items collected from Magela Creek are fit for human consumption and that concentrations of metals and radionuclides in organism tissues attributable to Ranger mine remain within acceptable levels. Enhanced body burdens of mine-derived solutes could also potentially reach limits that may harm the organisms themselves, and therefore any elevation in tissue concentrations can provide useful early warning of bioavailability of these constituents. Hence, the bioaccumulation monitoring programme serves an ecosystem protection role in addition to the human health aspect.

Local Aboriginal people harvest fish and mussels from Mudginberri Billabong, 12 km downstream of the Ranger mine. Routine monitoring of the levels of radionuclides and some metals in these food items commenced in 2000. Monitoring had not shown any issues of potential concern with regards to bioaccumulation in fish. Hence, the focus of the bioaccumulation monitoring programme has been directed at mussel tissue analysis, while the two-yearly fish sampling programme was discontinued in 2007.

Up until 2008, mussels were collected annually from Mudginberri Billabong (the potentially impacted site) and Sandy Billabong (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 observed in Sandy Billabong. Two research projects concluded that this difference was due to natural catchment influences and differences in water chemistry, rather than mining-related inputs to Magela Creek (Bollhöfer et al. 2011). Thus, the scope of the monitoring programme for mussel bioaccumulation was reduced from 2009 onwards. It now involves the annual collection and analysis of a bulk mussel sample from Mudginberri Billabong, rather than analysing separate age-classed mussels from both Mudginberri and Sandy Billabongs. This is done primarily to provide reassurance that the consumption of mussels does not present a radiological risk to the public. Every three years, starting in October 2011, a detailed study (analysis of aged mussels from both Billabongs) is conducted and results compared with those from previous years.

SSD's monitoring focuses on ^{226}Ra as it has been shown that ^{226}Ra in mussels is the biggest potential contributor to mine-related ingestion dose from a hypothetical release of pond waters from the minesite (Martin 2000). Consequently, the ^{226}Ra activity concentration in Magela Creek waters is routinely monitored by both ERA and SSD and its limit is based on potential dietary uptake of ^{226}Ra by the Aboriginal people downstream of the mine (Sauerland et al. 2005). To this end no increase of ^{226}Ra activity concentrations in Magela Creek downstream of the mine has been observed (Supervising Scientist 2013).

In contrast to ^{226}Ra and uranium, ^{210}Po activity concentrations are less elevated in mine waters when compared to natural waters. Total ^{210}Po in RP1 water was 3 $mBq{\cdot}L^{-1}$ in

2013, whereas ^{226}Ra and ^{238}U amounted to 140 and 63 mBq·L^{-1}, respectively (ERA 2013). This is because ^{210}Po (and ^{210}Pb) is particle reactive and readily adsorbed onto particulate matter and quickly transported from the water column into the sediment (Cochran & Masqué, 2003). Activity concentrations of ^{210}Po in Mudginberri Billabong water are low (2-8 mBq·L^{-1}) and have not shown an increase over the past 30 years (***eriss*** data, not shown). Nonetheless, ^{210}Po has been included in this year's suite of radioanalytes, to investigate ^{210}Po uptake in more detail, and its contribution to ingestion dose from the consumption of mussels.

Methods

Mussel, sediment and water samples were collected in October 2012 from Mudginberri Billabong (Figure 1). After collection, mussels were placed into acid-washed containers holding billabong water. 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. Sampling mussels at Mudginberri Billabong, October 2012.

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).

Although the routine sampling schedule required just the analysis of a bulk mussel sample for 2012, individual age groups were analysed. ^{226}Ra and ^{210}Pb were analysed via gamma spectrometry, ^{210}Po via alpha spectrometry and uranium and other metals via inductively coupled plasma mass spectrometry (ICPMS).

Uranium in freshwater mussels

Uranium concentrations in mussels and water samples collected concurrently from Mudginberri Billabong and Sandy Billabong over the years are shown in Figure 2. The concentrations of uranium in mussels from Mudginberri Billabong are very similar from 2000 onwards, with no evidence of an increasing trend in concentration over time. Essentially constant and low levels were also observed between 1989 and 1995. Notwithstanding some bioaccumulation with age, uranium in mussels is reported to have a short biological half-life (Allison & Simpson 1989), a conclusion that is supported by the current data. The low and constant uranium concentrations up to the last sample taken in October 2012 indicate absence of any mining influence.

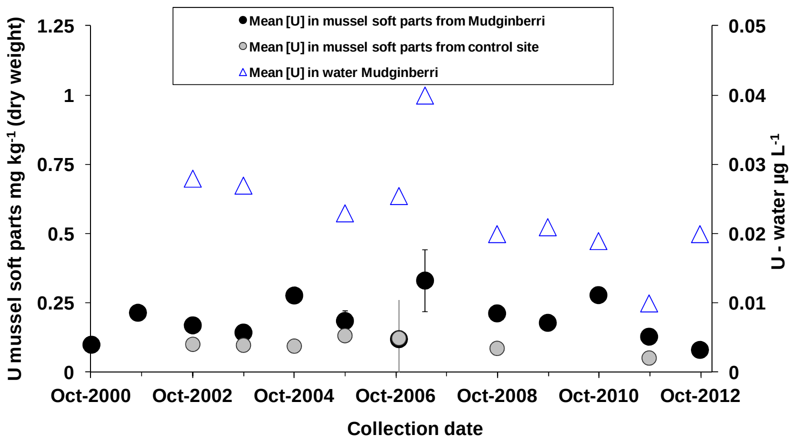

Figure 2. Mean concentrations of uranium (U) in mussel soft-parts and water from Mudginberri and Sandy Billabongs since 2000.

^{226}Ra, ^{210}Pb and ^{210}Po in freshwater mussels

^{226}Ra and ^{210}Pb activity concentrations in mussels collected from Mudginberri Billabong in 2012 are compared with the average activity concentrations measured in previous years in Figure 3. The graphs show that ^{226}Ra and ^{210}Pb activity concentrations in aged mussels are similar to the average from previous collections, although ^{210}Pb activity concentrations are somewhat lower. The lower values are most likely associated with differences in sampling location. As an insufficient number of mussels could be collected at the routine sampling site along the (eastern) bank opposite the Mudginberri boat ramp in 2012, some mussels were collected from a sandbank at the billabong inlet where ^{210}Pb activity concentrations in mussel flesh have previously been shown to be lower (Figure 4).

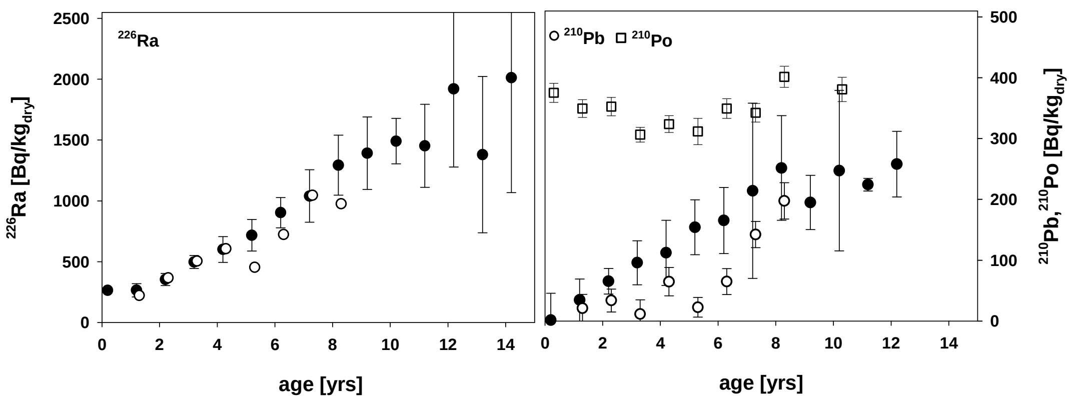

Figure 3. ^{226}Ra, ^{210}Pb and ^{210}Po activity concentrations measured in dry mussel flesh from Mudginberri Billabong plotted against mussel age. Averages of previous end of dry season collections (2000–2011) are shown as solid symbols, open symbols show the results from the 2012 collection.

^{210}Po activity concentrations in mussels from Mudginberri Billabong were between 300 and 400 Bq·kg^{-1} dry weight in 2012. ^{210}Po activity concentrations arise from both direct uptake of ^{210}Po from the water column and ingrowth from its radioactive parent ^{210}Pb. There is no apparent increase in ^{210}Po activity concentration with mussel age, consistent with its short physical half life of 138 days. The higher ^{210}Po activity concentration compared to ^{210}Pb indicates higher accumulation of ^{210}Po from the water column. This has been previously observed in mussels from billabongs in the Alligator Rivers Region (Johnston et al. 1987) and elsewhere (Carvalho & Oliveira 2008). Using the BRUCE tool (Doering 2013) a typical fresh weight activity concentration ratio (CR) for ^{210}Po for mussel flesh from Mudginberri Billabong relative to the activity concentration in total water of 13,700 has been determined.

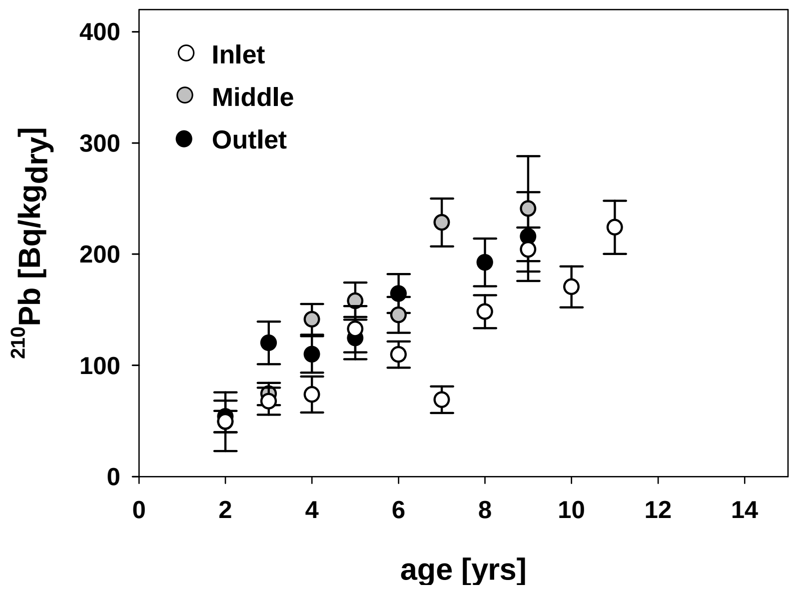

Figure 4 210Pb activity concentrations measured in dry mussel flesh from Mudginberri Billabong in 2008 at the billabong inlet, middle and outlet, plotted against mussel age.

Ingestion dose from the consumption of freshwater mussels

Based upon the measured activity concentrations of ^{226}Ra, ^{210}Pb and ^{210}Po in mussel flesh and the age distribution of mussels collected, an average annual committed effective dose from ingestion of these isotopes can be calculated for a 10 year old child who eats 2 kg (wet weight) of mussel flesh from Mudginberri Billabong. Figure 5 shows the doses from ^{226}Ra and ^{210}Pb ingestion estimated for individual years, and the median, 80 and 95 percentiles for all collections. In addition, the total dose (0.27 mSv) including ^{210}Po is shown for the 2012 collection. About 60% of the dose is from ^{210}Po, due to its comparatively high ingestion dose coefficient compared to the ingestion dose coefficients for ^{226}Ra and ^{210}Pb (ICRP 1996).

The difference between ^{226}Ra activity concentrations measured in Magela Creek upstream and downstream of the Ranger mine is only very small (see this report) and findings from previously reported research show that mussel ^{226}Ra activity loads in Mudginberri Billabong are due to natural catchment rather than mining influences (Bollhöfer et al. 2011). Due to ^{210}Po's relatively low activity concentrations in mine waters compared to ^{226}Ra and its behaviour in natural waters, it is thus unlikely that ^{210}Po activity measured in Magela Creek water is of mine-origin. Consequently, the ingestion dose reported here is almost exclusively from natural background contributions and would be received irrespective of the operation of the Ranger mine.

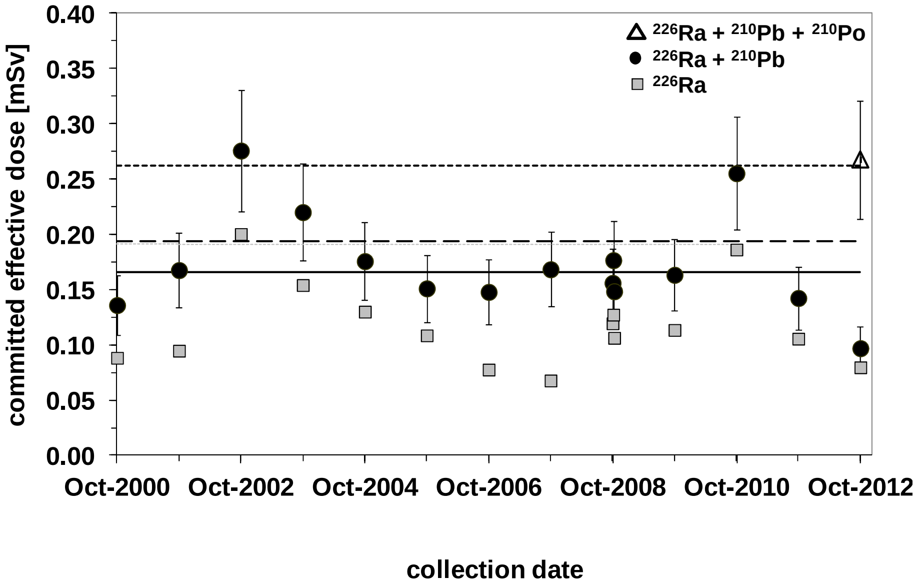

Figure 5. Annual committed effective doses (point data) from ^{226}Ra, ^{210}Pb and ^{210}Po for a 10 year old child eating 2 kg of mussels from Mudginberri Billabong. The median for ^{226}Ra + ^{210}Pb for all the data (solid line), the 80th percentile (dashed line) and 95th percentile (dotted line) are shown for reference.

Future work

Monitoring of radionuclide uptake in mussels from Mudgbinberri Billabong will continue. A review of the surface water monitoring activities of ERA and SSD by the Independent Surface Water Working Group took place in 2012. It has been suggested that SSD review existing 'metals in bush tucker' data before a more formal metal monitoring programme be re-introduced. Available ***eriss*** data on metal concentrations in mussel flesh are currently being collated and available historical samples re-analysed for metals using ICPMS, to guide decision making on biota, analytes and effort spent on a metal in bushtucker programme.

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Monitoring using macroinvertebrate community structure

CL Humphrey, J Hanley, L Chandler & C Camilleri

Macroinvertebrate communities have been sampled from a number of sites in Magela Creek at the end of significant wet season flows, each year from 1988 to the present. The design and methodology have been refined over this period (changes are described Supervising Scientist (2004), 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). A high-level protocol for the monitoring procedure is provided in Supervising Scientist Division (2013).

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 (Supervising Scientist Division 2013).

Results

Compilation of the full macroinvertebrate dataset from 1988 to 2013 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 or 'Exposure'; 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 2013) to after (2013) periods in the magnitude of upstream-downstream dissimilarity between the control and exposed streams ($p = 0.777$ and $p = 0.529$ for BA and BA*Exposure interaction respectively).

These results confirm that the dissimilarity values for 2013 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, including fluctuations in control streams. This variation over time is evident in Figure 1, particularly for recent years (2011 and 2012) in Gulungul Creek. 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. The sharp rise in dissimilarity in Gulungul Creek, particularly in 2011, was attributed to unusually high proportions of taxa with a preference for high velocity waters at the *upstream* site, associated with a wet season of high rainfall (see Humphrey et al. 2012). The magnitude of paired-site dissimilarity for Gulungul Creek in 2013 has declined from its peak in 2011, and is now back to similar values to those recorded prior to 2011 (Figure 1). This shift back to pre-2011 dissimilarity corresponds with a decline (from 2011) in the abundances of particular flow-dependant taxa at the upstream site (data not shown here), associated in turn with the low (below average) rainfall received in the 2012–13 wet season.

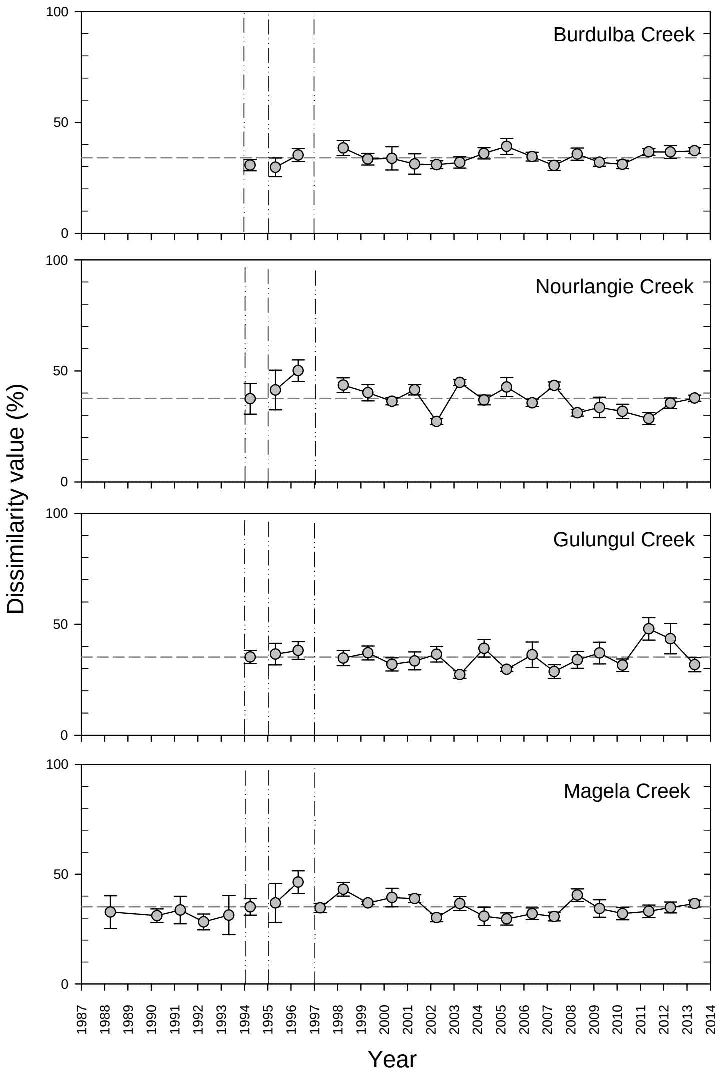

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2013. The dashed vertical lines delineate periods for which a different 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.

As demonstrated in Supervising Scientist Division (2013), the MBACIP paired-site dissimilarity approach, which reduces multi-dimensional data to just one sitepair metric and dimension (a metric scale from 0-100, as depicted in Figure 1), results in loss of information: (i) the influence of individual sites that comprise the paired-site dissimilarity cannot be determined; and (ii) *direction* of change in multivariate space can also not be determined. These constraints on the sitepair dissimilarity data limit interpretation of results; it is not possible to assign any change to either upstream or downstream site, while change in macroinvertebrate community structure at the exposed downstream site could occur in different directions in multivariate space. Thus, a non-significant before to after dissimilarity value could still mask real change that is occurring at the exposed site.

To ensure change is not passing undetected, two additional multivariate techniques are employed. Firstly, multivariate ordination is used to discern possible shifts in multivariate space in the downstream exposed sites away from controls sites. Secondly, PERMANOVA (PERmutational Multivariate ANalysis Of Variance) (Anderson 2001, McArdle & Anderson 2001, Anderson et al. 2008) is used to perform hypothesis tests on the multivariate data from *individual sites* (compared to paired site dissimilarity for the ANOVA above). Use of data from the individual sites offers increased partitioning of data variation (and hence increased factors). The ability to use the complete multivariate dissimilarity matrix enables PERMANOVA to better detect changes in *direction* in multivariate space that might otherwise be missed when using the simple sitepair dissimilarity metric data.

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 2013), relative to Magela and Gulungul Creek upstream (control) sites for 2013, and all other control sites sampled up to 2013 (Magela and Gulungul upstream sites, all sites in Burdulba and Nourlangie). Samples close to one another in the ordination indicate a similar community structure.

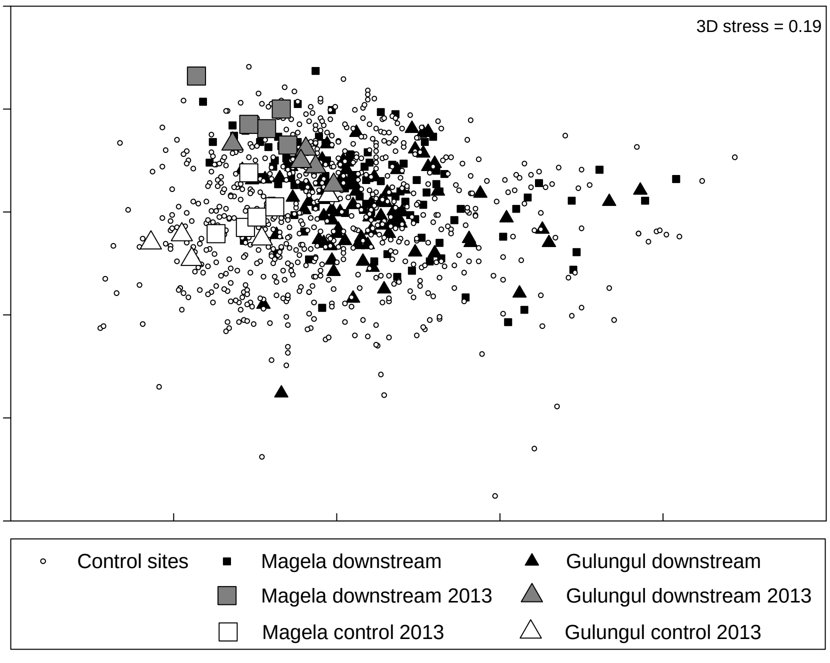

Figure 2 Ordination plot (axis 1 and 2) of macroinvertebrate community structure data from sites sampled in several streams in the vicinity of Ranger mine for the period 1988 to 2013. Data from Magela and Gulungul Creeks for 2013 are indicated by the enlarged symbols.

Data points associated with the 2013 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.

The ordination results were confirmed by PERMANOVA (PERmutational Multivariate Analysis Of Variance) testing on the individual sites. In 2011, within either the upstream or downstream sites at either 'duplicate' stream within the two exposure conditions, a significant difference occurred between macroinvertebrate community structures in the change from before (pre 2010–11) to after (2010–11) (i.e. BA*Up/Ds*Stream(Exposure) interaction significant, $p = 0.02$). Accompanying pairwise comparison tests showed that the upstream Gulungul and Burdulba sites differed from the before to after period (Supervising Scientist Division 2013). This same interaction in 2013 was not significant. Further, within either of the upstream or downstream sites across streams, and within either exposure type, there were no significant differences in macroinvertebrate community structures from the before to after period. (This latter BA*Exposure*Up/Ds interaction is the other important interaction for impact detection, Supervising Scientist Division 2013).

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 2013 have not adversely affected macroinvertebrate communities.

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Monitoring using fish community structure

M Ellis & C 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 Supervising Scientist (2004; section 2.2.3). These programmes were reviewed in October 2006 and the refinements to their design are detailed in Supervising Scientist (2007, 2008; section 2.2.3). Full protocols for the channel and lowland billabong fish monitoring techniques are provided in Supervising Scientist Division (2011a, b respectively).

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 Humphrey et al. (2013) (accompanying paper in this research summary). 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) is determined using multivariate dissimilarity indices calculated for each annual sampling occasion. A plot of the dissimilarity values from 1994 to 2013 is shown in Figure 1.

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 (2013) are consistent with the range of values reported in previous years (1994 to 2012) while the factor 'Year' tests for differences amongst years within the before or after periods. The ANOVA results for 2013 changed only slightly from those reported in Supervising Scientist annual report 2011–2012, showing no significant difference between 2013 and other years (BA factor not significant, $p = 0.771$). This indicates the relationship between Mudginberri and Sandy Billabong fish communities has remained consistent with relationships observed in previous years.

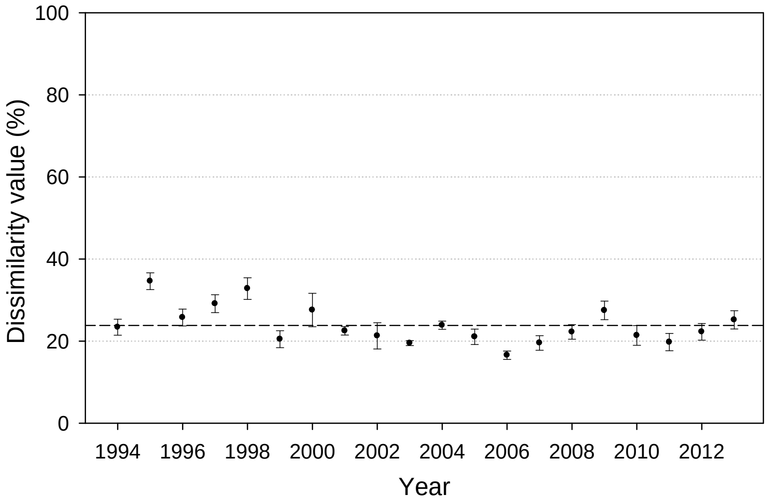

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs over time. Values are the mean dissimilarity (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of transect data between the two waterbodies. The dashed line is the mean dissimarity over all years.

However, as reported in 2012, the variation in fish assemblage dissimilarities between the two billabongs amongst years (tested by factor per year) was significantly different over the 1994 to 2013 period ($p < 0.001$). This variation over time, evident in Figure 1, has been demonstrated to be mainly associated with longer term changes in abundance of chequered rainbowfish (*Melanotaenia splendida inornata*) in Magela Creek (Supervising Scientist 2004; section 2.2.3). This species is the most common fish species in Magela Creek.

The annual changes in rainbowfish abundance in Magela Creek have been shown in previous Supervising Scientist annual reports to be negatively correlated with magnitude of wet season discharge (specifically wet season total, January total and February total) measured at G821009 in Magela Creek. More recently, rainfall at Jabiru Airport has been used in place of discharge data as it is considered more representative of regional wet season conditions (Supervising Scientist 2012). Inclusion of results from 2013 sampling support those from previous years, with negative relationships observed between rainbowfish abundance in Mudginberri Billabong and total annual rainfall ($p = 0.009$) and for the total rainfall in January ($p = 0.021$) and February ($p = 0.046$). This is particularly evident from the plotted data in Figure 2 which highlights the comparatively high rainbowfish abundances recorded in 2013, associated with a below-average wet season rainfall for 2012–13.

The 2013 results continue to support previous suggestions that reduced rainbowfish abundances occur after larger wet season rainfalls as a consequence of the more extensive upstream migration of rainbowfish past Mudginberri Billabong in response to high stream flows. This has the effect of reducing the concentration of fish in the billabong during the recessional flow period. Conversely, years of below average rainfall, such as 2012–13, have the potential to reduce upstream migration of rainbowfish, resulting in above average counts (see 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 2013 have not adversely affected fish communities in channel billabongs.

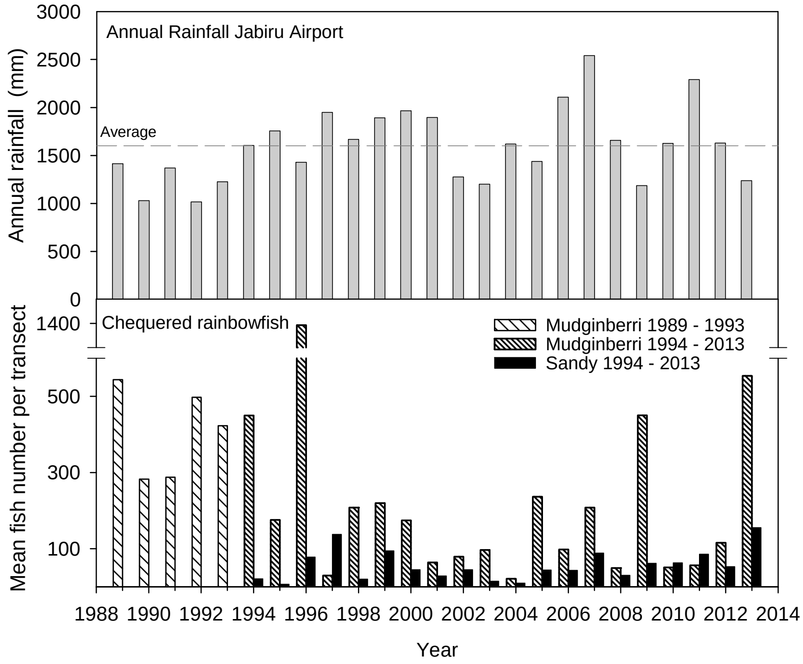

Figure 2 Relative abundance of chequered rainbowfish in Mudginberri and Sandy Billabongs from 1989 to 2013 with associated annual wet season rainfall recorded at Jabiru Airport.

Shallow lowland billabongs

Monitoring of fish communities in shallow lowland billabongs is usually conducted every other year, with the exception of 2011 where staff resources were directed to another project (see Supervising Scientist 2011). The last assessment was conducted in 2012 with results reported in Supervising Scientist (2012; section 2.2.3). The next assessment is due to be conducted between late April and June 2014.

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Environmental factors associated with toxicity monitoring in Magela and Gulungul Creeks

C Humphrey & M Ellis

Background and previous findings

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 accompanying paper by Humphrey et al. (2014) in this research summary

Following both the 2010–11 and 2011–12 wet seasons, statistical analyses and laboratory studies 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, 2013). This work helps distinguish between natural and mine-induced effects on snail egg numbers for impact assessment purposes.

The availability of continuous electrical conductivity (EC, $\mu S\ cm^{-1}$; a reliable surrogate of magnesium sulfate concentrations), water temperature and turbidity data between the 2006–07 and 2011–12 wet seasons provided opportunities to reliably assess the influence of these water quality variables on the snail egg production response. Over this period and for toxicity monitoring data from Magela and Gulungul Creeks, a number of significant correlations and interactions have been found between mean egg number and both median EC and water temperature, but not turbidity:

1. For creek water temperatures most commonly encountered (<30°C), a statistically significant positive linear relationship has been observed between EC and snail egg number.
2. A significant 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. Specifically, enhanced egg production with increasing EC at lower water temperatures (27–29°C) was noted, as well as an increasingly neutral effect at intermediate temperature (~30°C) and an increasingly reduced/negative effect at higher water temperatures (>30°C).
4. Across the tests conducted during each wet season, a mean upstream-downstream egg number difference value is derived, and this mean has an associated error or variance. This variability in the egg number difference values was observed to be significantly correlated with the same variability in corresponding upstream-downstream EC difference values.

Laboratory experiments confirmed the optimal egg production observed around 29°C (i.e. observation 2 above) but did not support an interacting effect of water temperature and EC upon snail egg production (i.e. observation 3 from above). Notwithstanding, the collective studies indicated that reproductive responses of freshwater snails exposed in Magela and Gulungul Creeks in the wet season appeared 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^{-1}). Median EC values greater than 20 µS cm^{-1} 1in the main wet season months (January–April) are typically a consequence of mine water discharges, and so inputs of water from the minesite were implicated in at least part of the stimulatory response.

An examination of environmental influences upon the snail egg production response continued during the 2012–13 wet season in Magela and Gulungul Creeks. Features of the wet season and the testing regime that had potential to improve an understanding of the snail egg production response included: (i) a focus on different husbandry conditions under which the snails were reared at the Jabiru field station facility, (ii) the below-average rainfall and stream flows during this wet season, and related to this (iii) the lack of significant mine water discharges from Ranger in the 2012–13 wet season. In relation to (ii) and (iii), the lack of significant mine-water inputs to the receiving waters would reflect responses to near-natural, but low-flow, water quality conditions, representing water quality exposure conditions very different from those experienced since 2006–07.

Influences of different snail culturing conditions on the snail egg production response

Background and methods

Until the 2012–13 wet season, no attention had been directed at the culturing conditions under which snails were reared at the Jabiru field station facility, as a potential source of variability in egg production during wet season test exposures in the creeks. Snails have typically been cultured in shallow (<0.5 m water depth) fibreglass or glass aquaria under static conditions but various systems, housed under or out of cover, have been used since 1991.

Since 2006–07, shallow containers held in a dedicated under-cover aquaculture facility have been used for culturing. Containers are 'seeded' for the ensuing wet season with stock from egg masses placed in the containers in the late dry season. Progeny, and ensuing generations, from this initial stocking would also be used for wet season testing as their shell length reached the minimum requirement (10 mm). For the 2011–12 wet season, snail stocks for sampling were sourced from both static shallow (0.5 m) and deep (~1.1 m) fibreglass containers, the latter using a Recirculating Aquaculture System (RAS). Both the shallow static and deep RAS containers and culturing regimes were also deployed for the 2012–13 wet season testing, and for this season the opportunity was taken, a posteriori, to compare the egg production under routine creek testing conditions of snails arising from the two different culturing regimes.

Husbandry-related factors that were examined for the 2012–13 wet season included (i) container type (shallow static, deep RAS), (ii) 'age' of the snail cohort, as measured by length of time from initial container seeding to use of the snails in a toxicity monitoring test, (iii) snail length and (iv) snail weight. Cohort age was a coarse measure of snail age because, as indicated above, there was continuous growth and recruitment of snails from the progeny of ensuing generations of snails held in each container. This mixed-age and size confounding was reduced by periodically eliminating snails from the containers and re-stocking. A record of the container source was made for each snail used in each of the toxicity monitoring tests conducted over the wet season.

Nevertheless, a significant deficiency in the design of this (a posteriori) study was the necessity for *pairs* of snails to be placed in egg-laying chambers - a feature of the monitoring protocol that has been in place since 1991. This gave rise to two difficulties: (i) the pairing made it impossible to assign an egg count to an individual snail; and (ii) there was potential for sourcing of any snail from the pair from different container types (see below). Because of this, some measure of independence of the egg counts was introduced for data analysis by splitting the count data into two. Thus for each test and site (i) the total egg count in each replicate egg-laying chamber (16 chambers per upstream and downstream site) made at the end of the four-day exposure period was assigned to each snail in the pair, (ii) the total count was attributed to both culturing containers from which the two snails were derived, and (iii) two datasets were derived, corresponding to each snail of the pair and the corresponding culturing containers, even though the egg production of the snail pair was a combined value and could not necessarily be apportioned to the original culture container.

In practice, it was more usual (55% of replicate snail pairs) for the snail pair to have been derived from the same container type (shallow or deep) than for the two snails to have been derived from different container types. Nevertheless, the high proportion (45%) of snail pairs for which individuals were derived from different container types represents a significant subsumption of variation in the attributed egg counts for these snail pairs.

Analysis of covariance testing (ANCOVA) was used to examine the effects of different husbandry conditions – culture container type and container cohort 'age', as well as the covariates snail shell length and shell weight, on snail egg number for the two creek systems, site in each creek (upstream/downstream) and period of the wet season (represented by test order). A problem with the analysis was the uneven distribution of container cohort age amongst the other factors. Hence, this factor could only be nested within other factors, resulting in it having little interpretative value.

Results

ANCOVA results of the analysis for one of the duplicate datasets, as defined and described above, may be summarised as follows:

1. Egg number per snail pair was significantly different between creeks (P = 0.035) and amongst tests (test order, P = 0.0002), while creek*test order (P = 0.0001), and creek*site (up/downstream, P = 0.025) interactions were also significant. From an examination of Figure 1B of the associated KKN study on "In situ toxicity monitoring", it is immediately evident, and unsurprising, that these factors and associated interactions are significant as egg production is highly responsive to water quality conditions over time and amongst different locations.

2. Egg number also correlated with snail length (P = 0.03) and snail weight (P = 0.0002) and again this is unsurprising given the positive fecundity-size relationship most freshwater organisms exhibit.

3. While container cohort age was significant in the analysis (P = 0.022), its nesting within other factors makes it impossible to ascertain interactions with other factors that are contributing to this significance.

4. Container type (shallow/deep) was highly significant in the analysis (P = 0.0001).

Results for the other duplicate dataset were very similar to those described above and hence are not considered further. To examine the significant container type effect further, for each toxicity monitoring test and each creek, the mean total egg count assigned to each snail in each snail pair (from above, combined datasets) for the two culture container types was determined, with values plotted in Figure 1.

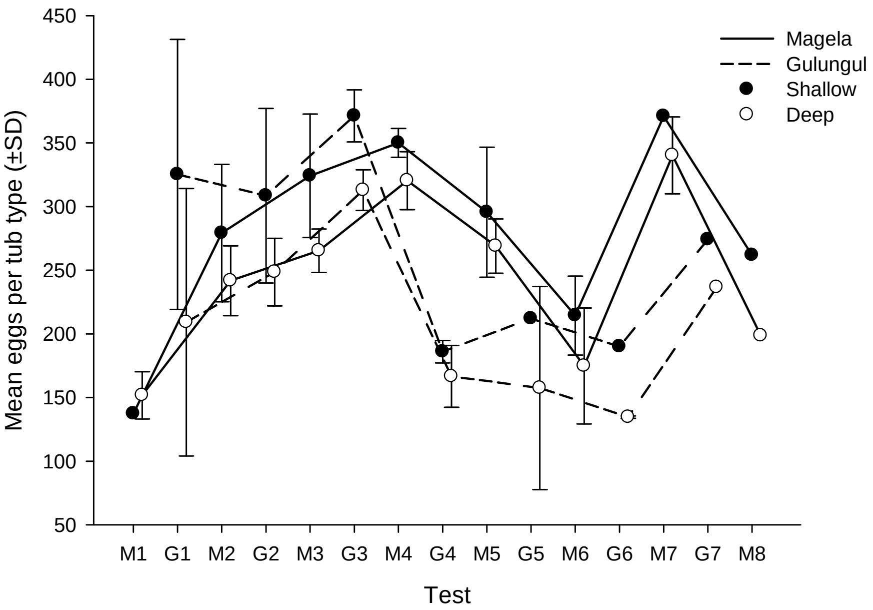

Figure 1 Mean total egg count assigned to individual snails sourced from shallow static and deep RAS culture container types for Magela and Gulungul toxicity monitoring tests conducted in the 2012–13 wet season. Horizontal axis shows the test order over the wet season, with M and G referring to Magela and Gulungul Creeks, respectively. Values with no error bars indicate single snail value.

Figure 1 shows the consistently higher egg production associated with snails from the shallow static containers compared to egg production associated with snails from the deep RAS containers. A practical concern of this result, revealed only in the latter part of the 2012–13 wet season, is the potential for snails from one of the two culture types to have been inadvertently and disproportionately allocated to one of the two sites (up/downstream) within a creek for a particular test. If a site had disproportionately more snails from one of the culture types, the upstream-downstream difference value for the test may reflect an artefact of culture type, as opposed to its proper representation of detecting changes in water quality. An examination of the datasets indicated even distribution of snails from the culture types between sites and across tests and in support of this, the Site*Tub type and Site*Tub type*Test order interactions in the statistical analysis described above were not significant. Further discussion of culture type is provided in the conclusions below.

Influences of ambient water quality on the snail egg production response

The below average rainfall for the 2012–13 wet season provided an opportunity for new insights to the snail egg reproduction response to water quality. Further, the observations of higher upstream-than-downstream egg production in Magela Creek and much higher downstream-than-upstream egg production in Gulungul Creek for the first three tests of this season (see Figure 1B from “In situ toxicity monitoring” section in this chapter) prompted further investigation of water quality influences. As discussed above, significant relationships have been observed in previous wet seasons between both water temperature and EC and the snail egg laying response. Hence, the median values of these two water quality variables over the four-day exposure periods of each test were plotted against mean egg number per snail pair for each creek and site to discern patterns (Figure 2).

The patterns depicted in Figure 2 appear to be consistent with the water temperature response reported previously (in the 'Background and previous findings' section), i.e. an optimal egg laying response was observed, between 30 and 31°C, after which egg number (in Magela Creek) declined. Lack of any apparent relationship between egg number and EC in Figure 2 is masked by an interacting effect of water temperature. To examine the possible enhanced egg production with increasing EC at lower water temperatures but increasingly reduced/negative effect at higher water temperatures, egg number versus EC plots were prepared for these temperature extremes, as shown in Figure 3. The plots confirm the earlier findings with statistically significant ($P<0.05$) increase and decrease in egg production with increasing EC observed at lower and higher water temperatures, respectively. With the median water temperatures annotated next to each test (Figure 3), it is clear that the EC effects are independent of water temperature.

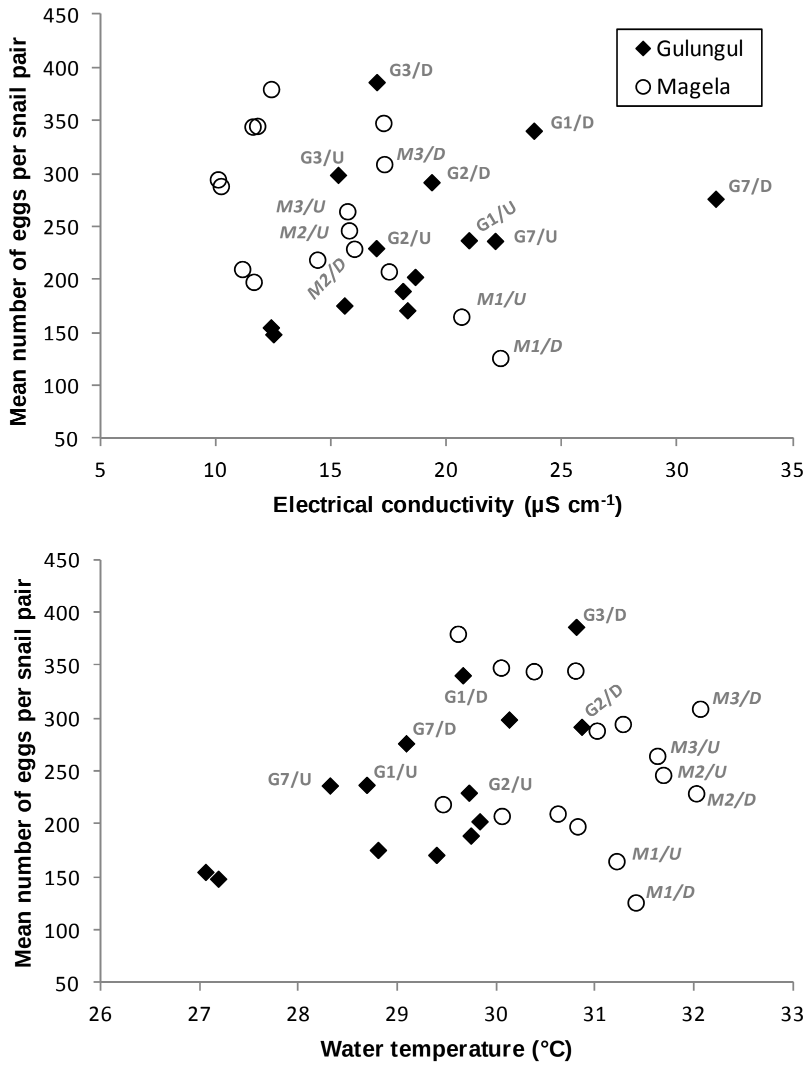

Figure 2 Relationships between mean snail egg number for each creek and site, and ambient electrical conductivity and water temperature over the four-day exposure test periods of the 2012–13 wet season. Selected tests are denoted by annotations, e.g. M1/D – Magela Creek test 1, downstream site, G7/U – Gulungul Creek test 7, upstream site etc.

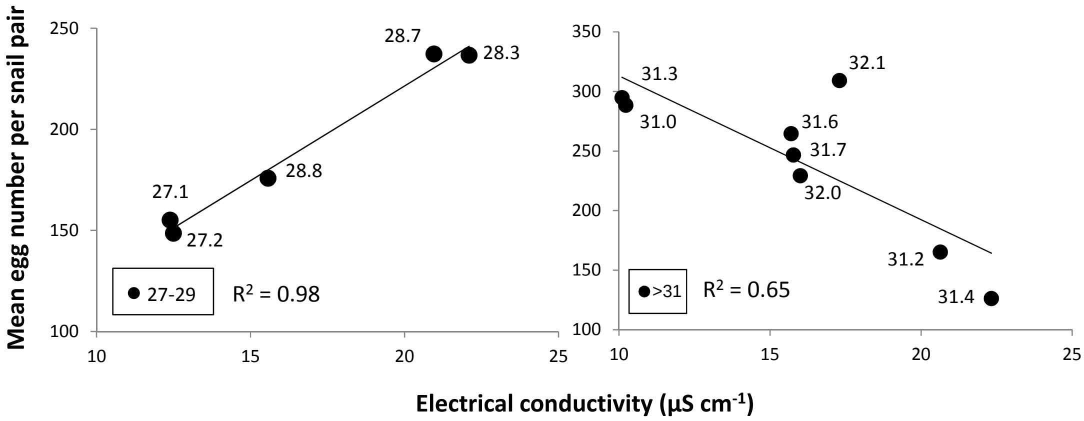

Figure 3 Significant (P<0.05) relationships between mean snail egg number for each creek and site, and ambient electrical conductivity, for specific water temperature ranges (27-29 and >31°C) of the 2012–13 wet season. Annotations are the median water temperatures measured over the four-day exposure test periods.

In light of these results, 2012–13 wet season toxicity monitoring observations may be assessed, in particular, the generally lower downstream (compared to upstream) egg production observed in Magela Creek, and the particularly high downstream (compared to upstream) egg production observed in the first three tests in Gulungul Creek. For this assessment, exemplary data points in Figure 2 have been annotated according to creek, test number and site, to aid interpretation.

From Figure 2 it is evident that different water temperature regimes prevailed in Magela and Gulungul Creeks over the 2012–13 wet season: median water temperatures were typically between 30 and 32°C in Magela Creek but lower, between 27 and 30°C, in Gulungul Creek. These respective water temperature ranges span either side of the optimal water temperature for the egg laying response observed in this and previous wet seasons, whereby the Gulungul temperature range provides enhanced egg production while the Magela range is typically associated with a reduction in snail egg production.

Further, over the lower (Gulungul) temperature range, increasing EC has historically correlated with increasing egg production while over the higher (Magela) temperature range, increasing EC has historically correlated with decreasing egg production, over the range 10–30 µS cm^{-1} (Supervising Scientist 2012; section 3.4). These patterns were affirmed for the 2012–13 wet season (see Figure 3). In both creeks and across all tests for the 2012–13 wet season, median water temperature was higher at the downstream monitoring sites.

In the light of these findings and observations, the generally lower Magela, but higher Gulungul, downstream egg production, compared to respective upstream sites, reflects the inhibitory and enhanced effects respectively, associated with the different water temperature regimes in each of the creeks. The higher downstream EC in Gulungul Creek (compared to upstream), in association with 'favourable' water temperature range, would have the effect of exacerbating a stimulatory egg laying response – evident in the first three Gulungul tests (Figure 1B of "In situ toxicity monitoring" section above). The generally lower downstream egg production in Magela Creek appears to be solely a ('high') water temperature effect, though the first test of the season (M1, Figure 2) was also associated with relatively high (natural) ECs for the season.

For only one toxicity monitoring test (the last Gulungul test, Figure 1B of 'In situ toxicity monitoring' section above) was the median EC value greater than 25 µS cm^{-1}, indicating for both creeks and across the 2012–13 wet season, very limited mine water inputs (see Supervising Scientist 2013; section 3.2.3.1). Thus, the toxicity monitoring results for 2012–13 reflect patterns associated with natural water quality (water temperature, EC) conditions in both creeks. There was no evidence of mine-related effects upon snail egg production over the wet season.

The 2012–13 wet season observations, made under near-natural, but low-flow, water quality conditions, upheld observations made in the previous two wet seasons, thus:

1. A significant ($P = 0.05$) unimodal relationship was found between water temperature and snail egg number using the combined Magela and Gulungul datasets (Figure 2), while the significant interacting effect of water temperature and EC upon snail egg counts in Magela and Gulungul Creeks continued (Figure 3).
2. Significant correlation ($P < 0.05$) between variability in the wet season upstream-downstream egg number difference values and the same variability in corresponding upstream-downstream EC difference values was previously shown in Supervising Scientist (2012; Figure 3.18). Addition of 2012–13 wet season data points to that dataset (graph not included here) supported and continued that relationship.

Conclusions

Toxicity monitoring results for the 2012–13 wet season revealed for the first time the significant influence of snail culturing conditions upon the egg laying response in ensuing toxicity monitoring tests. A measure of how important culturing history is, compared to ambient creek water quality, in determining the magnitude of the egg laying response is possible from the ANCOVA results described above. The proportion of the variation associated with treatments/factors is available from the Sums of Squares column of the ANCOVA (not provided here). Variation in snail egg counts accounted for by culture conditions and ambient creek water quality is associated with factors container type and test order respectively. For these, variation associated with ambient water quality was about 2.5 times greater than culture-type variation. Given that it was not possible in many instances to accurately assign egg count data to culture container type, this proportional estimate of variation may underestimate the true contribution of culture type.

In future wet seasons, attention will be focused on designing a rigorous experimental approach to assessing the influence of husbandry conditions on the snail egg laying response. This design would include at least the current two culturing conditions, it would consider the effects of different seasons in which containers are 'seeded' and established, and it would require that each snail pair used in a replicate egg-laying chamber was sourced from the same culture container type.

The same culturing conditions as used in the 2012–13 wet season were also employed in the 2011–12 wet season, yet the magnitude of egg production observed in the latter wet season was much lower than that observed in the 2012–13 wet season (Humphrey et al. 2014; Figure 1). Monitoring staff noted that culturing container establishment for the 2011–12 wet season may have occurred earlier in the 2011 dry season, compared to the following year. Earlier seeding would have coincided with lower water temperatures and would have allowed for establishment of older (and possibly less vigorous-for-size) snails by the time they were required for wet season testing. These hypotheses can be tested in the design of future research studies on this topic.

Acknowledgements

We thank Mr Duncan Buckle and Dr Keith McGuinness for advice on data analysis associated with the husbandry aspect of this study. (Neither researcher was consulted on the design of the study.)

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Atmospheric radioactivity monitoring in the vicinity of the Ranger mine

C Doering & A Bollhöfer

Introduction

Uranium mining has the potential to release radon (a radioactive noble gas) and dust-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 above background radiation dose received by the public in the vicinity of a uranium mine.

National radiation protection standards (ARPANSA 2005) require that the annual radiation dose received by a member of the public from practices such as uranium mining and milling must not exceed 1 millisievert (mSv) above the natural background. This dose limit applies to the sum of doses received via all exposure pathways that are traceable to the mining practice.

The Ranger mine is currently the only operating uranium mine in the Alligator Rivers Region (ARR) and the main potential source of above background radiation dose to local communities. The main areas of permanent habitation in the vicinity of the mine are Jabiru town and 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). The purpose of these measurements is twofold: (i) to provide an independent check of the values measured and reported by the Ranger mine operator (ERA); 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 radiation risk.

Method

Environmental radon daughter monitors from Radiation Detection Systems in Adelaide were used for continuous radon progeny monitoring. The monitors operated at a flow rate of 0.35 l/min drawing air through a Whatman GF/C filter positioned above an alpha counter. Hourly potential alpha energy concentration (PAEC) data were 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 and quarterly average radon progeny PAEC monitoring data from Jabiru town and Mudginberri, respectively, for the 2012 calendar year. Gaps in the data are due to instrument maintenance and data quality issues.

The spikiness in the hourly PAEC data reflects the normal diurnal pattern in radon progeny concentrations in surface air. Higher concentrations typically occur in the early morning around sunrise when atmospheric conditions tend to be most stable. 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 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.

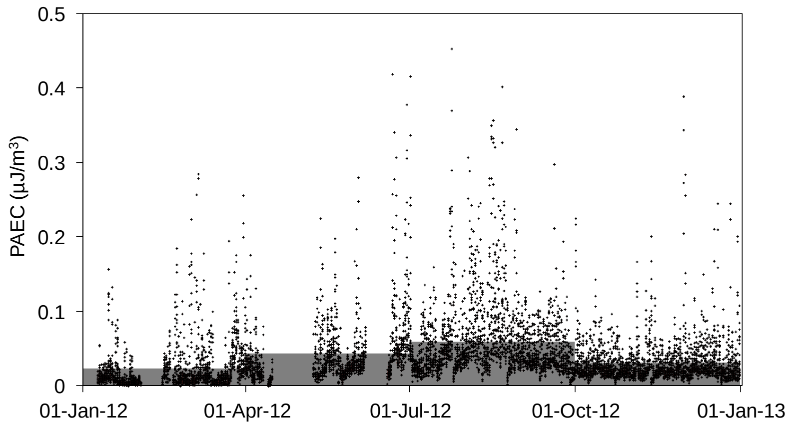

Figure 1 Hourly (black crosses) and quarterly average (grey columns) radon progeny PAEC in air at Jabiru town in 2012

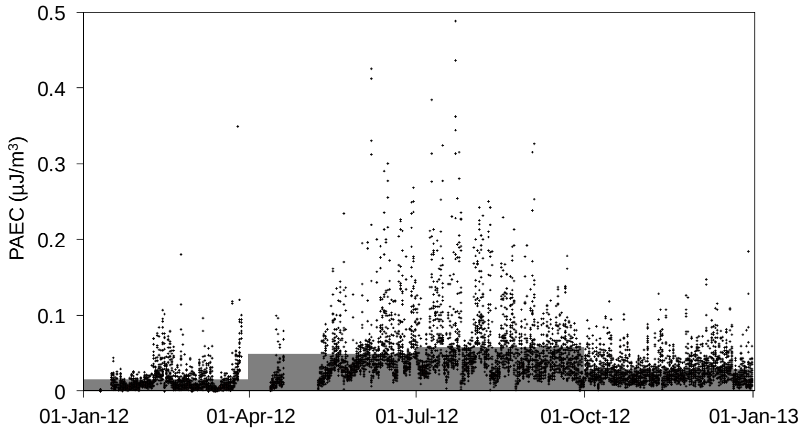

Figure 2 Hourly (black crosses) and quarterly average (grey columns) radon progeny PAEC in air at Four Gates Road near Mudginberri community in 2012

Table 1 provides a summary of annual average radon progeny PAEC in air and estimated doses to the public, as well as comparison with values reported by ERA for Jabiru town.

The total annual effective dose from radon progeny in air, which includes contribution from natural background, has been estimated to be 0.386 mSv at Jabiru town and 0.372 mSv at Mudginberri. This total annual dose has been calculated from the product of the annual average radon progeny PAEC in air, the radon progeny dose conversion factor of 0.0011 mSv per µJ·h/m³ recommended by the International Commission on Radiological Protection (ICRP) (ICRP 1993) and the assumed full year occupancy of 8784 hours.

Table 1 Radon progeny PAEC in air and estimated doses to the public at Jabiru town and Mudginberri in 2012*

	Jabiru town	Mudginberri
Annual average PAEC (µJ/m³)	0.040 (0.036)	0.039
Total annual dose (mSv)	0.386 (0.330)	0.372
Mine-related dose** (mSv)	0.030 (0.055)	0.005

* Values in brackets refer to data from the ERA Radiation Protection and Atmospheric Monitoring Programme Report for the Year Ending 31 December 2012.

** The radon progeny PAEC difference used in the SSD mine-related dose calculation was 0.016 µJ/m3.

The mine-related annual dose from radon progeny in air has been estimated to be 0.030 mSv at Jabiru town and 0.005 mSv at Mudginberri. This dose is dependent on wind direction and has been calculated from the difference in average radon progeny PAEC in air when the wind was from the direction of the mine and when the wind was from directions other than the mine, then multiplying this difference with the ICRP radon progeny dose conversion factor and the number of hours that the wind was from the direction of the mine. Hourly wind direction data for 2012 were obtained from the Bureau of Meteorology weather station at Jabiru Airport. Analysis of these data (Figure 3) suggests that the wind was from the direction of the mine for 1724 hours during the year at Jabiru town (90–110 degree sector) and 446 hours during the year at Mudginberri (140–160 degree sector).

Differences between the ***eriss*** and ERA radon progeny PAEC results and public dose estimates for Jabiru town are most likely due to differences in monitoring regime. Whereas ***eriss*** aims to monitor continuous hourly radon progeny PAEC in air over the full year, the ERA regime is based on a minimum requirement of one week per month continuous monitoring.

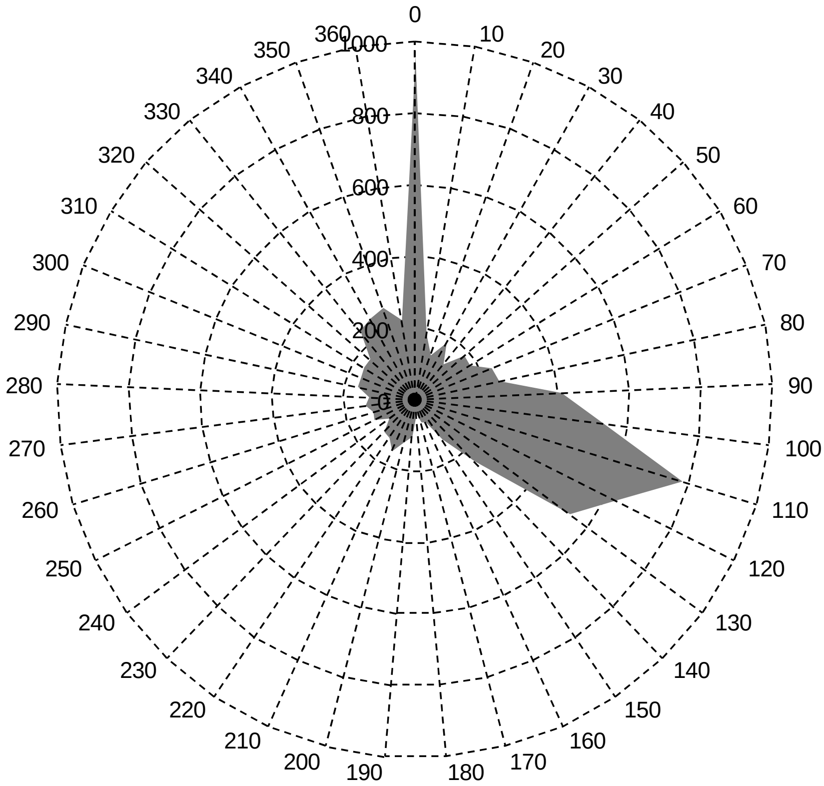

Figure 3 Histogram of hourly wind direction data at Jabiru airport for 2012. A wind direction of 0 indicates still conditions (i.e. no wind).

Dust-bound LLAA concentrations

Figures 4 and 5 show measured and quarterly average concentrations of dust-bound LLAA radionuclides in air at Jabiru town and near Mudginberri community, respectively, for 2012. Gaps in the data are due to instrument maintenance and data quality issues.

The general trend was 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.

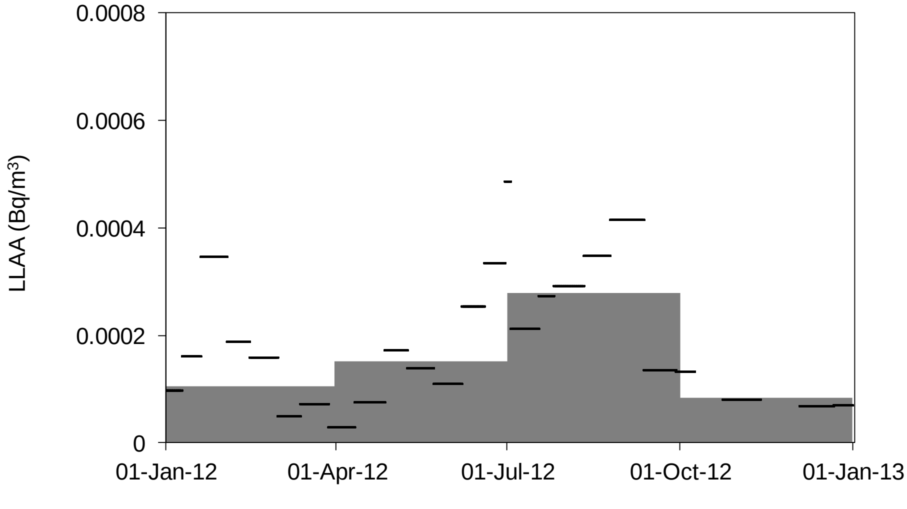

Figure 4 Measured (black lines) and quarterly average (grey columns) concentrations of dust-bound LLAA radionuclides in air at Jabiru town in 2012

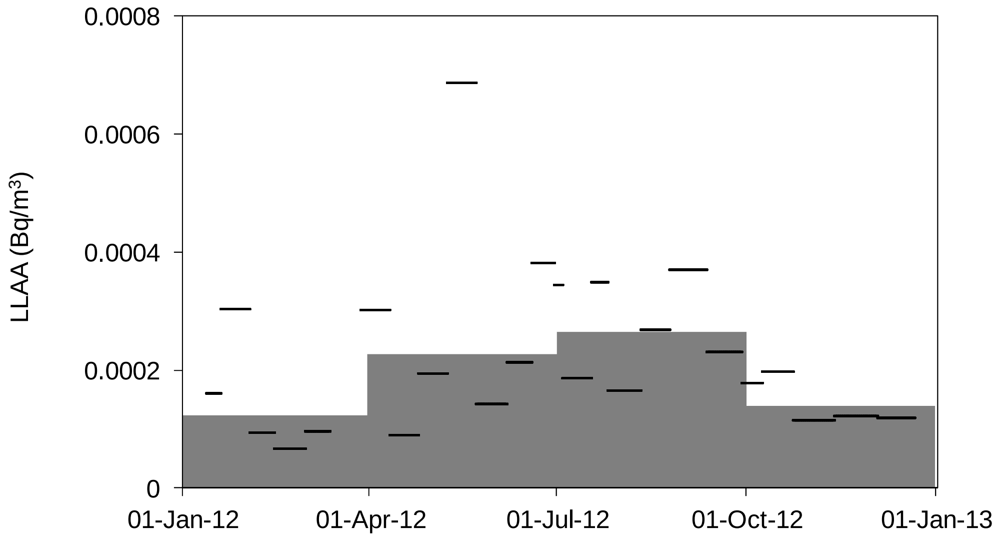

Figure 5 Measured (black lines) and quarterly average (grey columns) concentrations of dust-bound LLAA radionuclides in air at Four Gates Road radon station near the Mudginberri community in 2012

Table 2 provides a summary of the annual average LLAA radionuclide concentration and estimated total and mine-related doses to the public at Jabiru town and Mudginberri.

The total annual effective dose from dust-bound LLAA radionuclides, which includes contribution from natural background, has been estimated to be 0.007 mSv at Jabiru town and 0.008 mSv at Mudginberri. This total annual dose has been estimated by calculating the time weighted annual average LLAA concentration from the individual samples and then multiplying with a dose conversion factor of 0.0061 mSv/Bq_{α}, breathing rate of 0.75 m^3/h and assumed full year occupancy of 8784 hours.

Table 2 LLAA radionuclide concentrations in air and estimated doses to the public at Jabiru town and Mudginberri in 2012

	Jabiru town	Mudginberri
Annual average LLAA (Bqα/m3)	1.7×10^{-4}	1.9×10^{-4}
Total annual dose (mSv)	0.007	0.008
Mine-related dose* (mSv)	5×10^{-4}	1×10^{-4}

* 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 mine-related dose from dust-bound LLAA radionuclides has been estimated to be less than 0.001 mSv at both Jabiru town and Mudginberri. This dose has been calculated by assuming that the ratio of mine-related to total annual dose from dust is the same as that for radon progeny. This assumption is likely to result in an overestimate of the mine-related dose via the dust inhalation pathway. 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 for 2012 indicate that inhalation doses to the public from mine-related sources of radon progeny and dust-bound radionuclides in air are low and do not pose any unacceptable radiation risk.

The total annual effective dose from radon progeny in air, which includes contribution from natural background, was estimated to be 0.386 mSv at Jabiru town and 0.372 mSv at Mudginberri. The mine-related component of this dose was only 0.030 mSv at Jabiru town and 0.005 mSv at Mudginberri.

The total annual effective dose from dust-bound LLAA radionuclides in air, which includes contribution from natural background, was estimated to be 0.007 mSv at Jabiru town and 0.008 mSv at Mudginberri. The mine-related component of this dose was less than 0.001 mSv at both Jabiru town and Mudginberri.

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Monitoring of radionuclides in groundwater at Ranger

A Bollhöfer & P Medley

Introduction

Groundwater samples from bores around the Ranger uranium mine have been collected by ***eriss***, ERA and the Northern Territory Government Department of Mines and Energy (DME) for more than 30 years, initially to monitor seepage of contaminants from the tailings storage facility (TSF) and the land application areas (LAA) into the surrounding environment. More attention has now been given to the mined out pits due to Pit 1 being used as a tailings repository and Pit 3 being prepared to accept tailings scraped from the TSF to commence in 2015. Both routine monitoring of groundwater quality (e.g. ERA 2012) and modelling groundwater behaviour in the vicinity of the pits (e.g. Timms et al. 2010) is undertaken.

Groundwater quality parameters routinely measured include major ions (ERA/DME), heavy metals (ERA/DME/***eriss***) and radionuclides (ERA/***eriss***) (Martin & Akber 1996, 1999; Klessa 2001; ERA 2012). Between 1996 and 2003 a number of bores were routinely sampled and analysed by ***eriss*** and in 2003 the decision was made that sampling by ***eriss*** will be discontinued with radionuclides to be analysed in aliquots of bore waters sampled by DME. At their 12th meeting in April 2003, ARRTC identified the groundwater pathway as needing further research and groundwater dispersion as one of the Key Knowledge Needs that need to be addressed from a monitoring perspective, extending well into the closure phase of Ranger mine. Since 2004 ***eriss*** has received aliquots of samples collected by DME to continue its groundwater programme, in particular the measurement of ^{226}Ra and uranium isotopes.

Due to the disconnection of the various programmes involved in groundwater monitoring at Ranger, the information gathered is still not used to its full potential, with groundwater quality data currently stored in several databases and in various formats. As part of an effort to improve the Ranger groundwater knowledge base, as well as facilitate a more coordinated approach to the acquisition and storage of groundwater data (with the goal to progress the development of closure criteria for Ranger) ***eriss*** radionuclide data from bores sampled by DME up to 2009 were sent to ERA for inclusion in their groundwater GIS. The focus for ***eriss*** in 2012–13 was on the review and QA/QC of all available ***eriss*** groundwater data, including historical data, and drafting of an internal summary report.

Methodology

All groundwater data available to ***eriss*** up to May 2013, including ^{226}Ra, uranium isotopes and metals, have now been thoroughly checked and imported into a single *Excel* groundwater spreadsheet. The data has also been migrated into the *EnviroSys* database.

UTM WGS84 eastings and northings, bore screen depths, and information about the status of all investigated bores (active, buried etc.) has been included in the database. This information has mostly been taken from Ranger uranium mine's Bore Audit Project. Bores have then been grouped according to catchments and all analytes available, including metals data, plotted against sampling date. Results from this data review are currently being summarised and written up in an ***eriss*** Internal Report.

Progress to date

The focus of ***eriss***' groundwater monitoring and research projects has been on radionuclides, with time records for some bores going back to the early 1980s. More than 50 different bores have been investigated. Whereas samples from the 1980s and 90s were analysed for radioisotopes only, samples collected post 1998 were generally analysed for radioisotopes and metals. Overall more than 350 samples have been collected and analysed for radioisotopes so far, and about half of those have included the analysis of metals.

Investigation of many bores was discontinued, as they were buried as a result of various TSF wall raises, stockpile constructions or were damaged by earthworks in their vicinity. Other bores sampled by DME since 2003 have been added to the list of bores investigated by ***eriss***. A review of the available data for various bores showed that there is time series data available for some bores going back a decade or more (some to the late 1980s). Figure 1 shows a vertical aerial photograph of the Ranger mine and the location of these bores. More bore information is listed in Table 1.

Table 1 Information for bores shown in Figure 1.

BoreID	RN number	Data from-to	Easting	Northing	Screen depth	Site name
RN23551	RN023551	9/04 - 9/12	272435	8597838	4 - 4.5	RP1/Coonjimba
B11	RN007243	9/06 - 9/09	272126	8599051	0 – 13	RP1/Coonjimba
OB23	RN022937	11/89 - 9/12	271844	8597241	36 – 51	North of TSF
OB1A	RN022902	11/89 - 9/12	271561	8596865	16 – 31	North of TSF
RN9329	RN009329	9/03 - 9/12	270863	8596371	17 – 19	Gulungul catchment
OB20	RN022934	11/89 - 9/12	271822	8595161	21 – 36	South of TSF
OB21A	RN022935	11/89 - 9/12	272484	8595180	31 – 43	South of TSF
OB30	RN022941	11/89 - 9/12	273923	8595706	20 – 35	Corridor Ck catchment
OB27	RN022930	9/03 - 9/12	275523	8597063	15 – 40	Corridor Ck catchment
83_1	RN023010	5/03 - 9/12	274414	8598255	0 - 90	Magela LAA

The bores shown in Figure 1 cover strategically important locations for groundwater monitoring around the TSF (OB23, OB1A, OB20, OB21A), the Gulungul catchment (RN9329), close to Pit 1 (OB30), in the Corridor Creek catchment (OB27), close to Pit 3 in the Magela land application area (83_1) and in the RP1/Coonjimba catchment close to the Coonjimba fault line (RN23551, B11). Two of the bores, 83_1 and OB27, are statutory monitoring bores of ERA. Monitoring of these bores should continue and needs to include the measurement of ^{226}Ra, ^{238}U and ^{234}U/^{238}U activity ratios by ***eriss***. Uranium and ^{226}Ra data for the bores in the vicinity of the TSF and the Gulungul catchment have been reported and discussed in last year's ***eriss*** report to ARRTC 28.

Figure 1 Map of bores with water quality time series exceeding seven years.

Steps for completion

The routine measurement of ^{226}Ra, ^{238}U and ^{234}U activity concentrations in monitoring bores sampled by DME will continue but shall focus on bores identified in Table 1. Once analysed, the September 2013 data will be added to SSD's database. An internal report summarising all available ***eriss*** groundwater data is nearing completion and once finalised (including recommendations on the structure and extent of ***eriss***' groundwater monitoring programme in the future) the radionuclide data will be added to the spatial groundwater GIS developed by ERA for the site.

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Part 2: Ranger – Rehabilitation

Geomorphic characteristics of the Gulungul Creek catchment

WD Erskine, MJ Saynor, K Turner, S Fagan & A Chalmers

Introduction

Erskine & Saynor (2000) recognised that Gulungul Creek was likely to be impacted by the operation of Ranger Mine as well as by closure and rehabilitation works. They recommended that a greater focus needed to be given to assessing mine-related impacts on Gulungul Creek. As a result, during 2002, ***eriss*** started to implement new mine-related research on Gulungul Creek, including hydrology, geomorphology and biogeography (Crossing 2002). Crossing's (2002) initial work was commenced for a PhD thesis which was never completed. However, the work has been continued and expanded by the Hydrological, Geomorphological and Chemical Processes Group.

The aim of the project is to determine baseline morphodynamics of Gulungul Creek to provide a reference condition for assessing mine impacts and rehabilitation success. Morphodynamics refer to how a catchment, river and floodplain respond to variable inputs of rainfall, discharge, sediment and solutes in terms of morphology, and sediment and solute fluxes. The role of vegetation in mediating morphodynamic response is of particular interest.

Methods

Data have been collected since 2002 during most dry seasons on:

- cross-section dynamics at 13 locations along the channel (following the approach of Saynor et al. 2004) and
- bed-material dynamics at each cross section (following Saynor et al. 2006)

Crossing (2002) sited the cross sections and completed the first survey. Some of the subsequent data are reported in Saynor et al. (2005), Saynor & Smith (2006) and Saynor et al. (2010).

In addition, detailed work has been completed on:

- scour and fill using scour chains (following Saynor et al. 2004) at selected cross sections for selected wet seasons
- morphology and sediment stratigraphy of the floodplain (Erskine & Saynor 2014)
- riparian vegetation effects on channel and floodplain stability at selected cross sections (similar to Erskine et al. 2009)
- the role that riparian vegetation plays in controlling channel and floodplain stability (similar to Erskine et al. 2012a)

There were three gauging stations initially installed on Gulungul Creek for this project and subsequently used for continuous monitoring, one upstream of Ranger Mine (GCUS), one downstream of Ranger mine (GCDS) and one in between at the former gauge operated by the former Department of Natural Resources, Environment, The Arts and Sport (NRETAS) of the Northern Territory Government, G8210012. Moliere (2005) has previously analysed the recorded data collected by NRETAS at G8210012. During each wet season the following data are collected at either 5- or 6-minute intervals at each gauging station:

- Stage height
- Rainfall
- Discharge measurenments.

Manual velocity-area gaugings initially using a current meter (Figure 1) and then an Acoustic Doppler were used to construct rating curves for the conversion of the stage data to discharge at each gauging station. Rating curves have been constructed for the complete period of record at each station. There is a natural channel control at each station so frequent gaugings are essential to maintain accurate ratings.

In addition, at the upstream and downstream gauging stations, continuous measurements (5 or 6 minute intervals) are taken of:

- Turbidity and
- Electrical Conductivity (EC).

Bulk water samples are collected by pumping with Gamet autosamplers at the up- and downstream stations and various analytes have been determined. Some bedload gaugings (following Erskine et al. 2011) have also been completed at the up- and downstream stations.

Figure 1 Velocity-area gauging using a current meter and wading rod out of a boat at the gauging wire at G8210012 on 8 February 2005. A number of gaugings are needed to construct a rating curve (water surface elevation or stage on the gauge board versus discharge) to convert the stage record to discharge.

Preliminary results:

a) Hydrology

The catchment areas at the three gauging stations are 40 km² at GCUS, 46 km² at G8210012 and 66 km² at GCDS. They are located at cross sections UG01 (GCUS), MG06 (G8210012) and DG10 (GCDS) in Figure 2. Table 1 shows the annual rainfall, runoff and peak discharge for each year with complete records at each station. Data are recorded for the water year from September to August to avoid combining data from different wet seasons which usually persist from November to April. The data were taken from Moliere et al. (2005; 2007a; 2007b; in preparation) for the first five years of record (2003–04, 2004–05, 2005–06, 2006–07 and 2007–08) because they had infilled any gaps in the record that could be reliably estimated. Additional research is required to infill the remaining gaps in the station record.

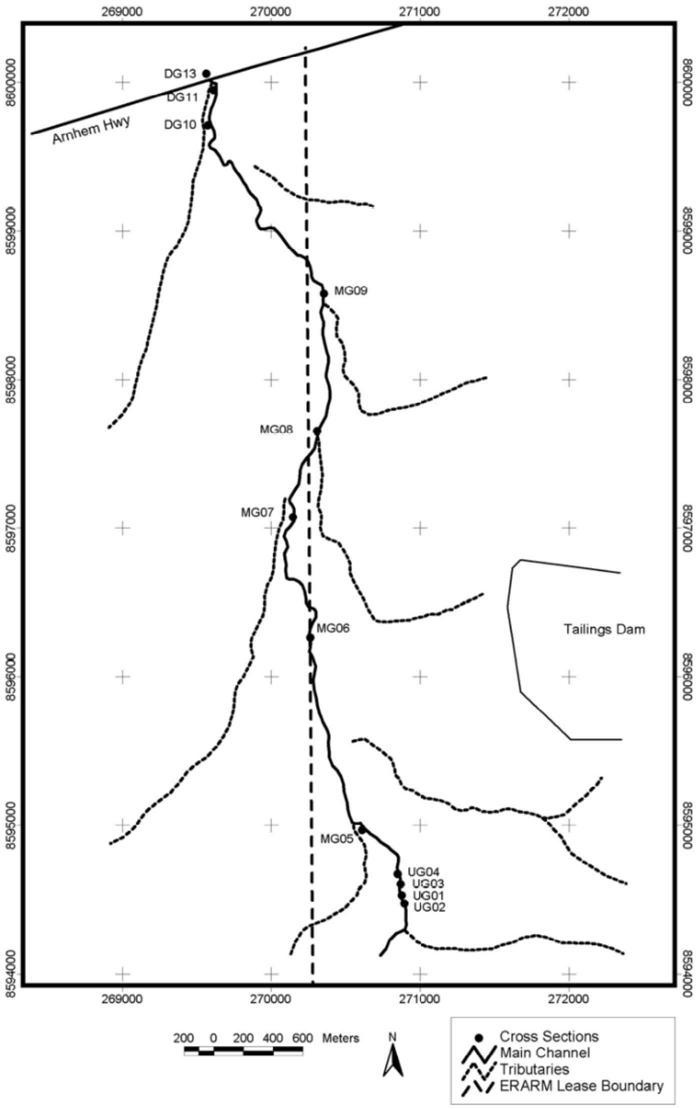

Figure 2 Gulungul Creek next to Ranger Mine showing locations of permanently marked cross sections. Cross section DG12 is not shown but is located on the upstream side of the culvert on the Arnhem Highway (from Crossing 2002).

Table 1 Rainfall, runoff and maximum discharge for each year of complete record at the three ***eriss*** gauging stations on Gulungul Creek

Gauging Station	Year	Rainfall (mm)	Runoff (ML)	Runoff (mm)	Runoff Coefficient (%)	Maximum Discharge (m^3s^{-1})
GCUS	2003-04	1540	27041	676	43.9	23.2
	2004-05	1591	23846	596	37.5	31.4
	2005-06	2137	49105	1228	57.4	59.1
	2006-07	2218	51822	1296	58.4	284
	2007-08	1518	27703	693	45.6	62.6
	2008-09	1199	11051	276	23.0	39
	2009-10	1369	20892	522	38.2	44
	2010-11	2285	51703	1293	56.6	96
	2011-12	1501	40495	1012	67.4	28
	2012-13	1286	19708	493	38.3	72
G8210012	2003-04	1611	31412	683	42.4	67.6
	2004-05	1492	25113	546	36.6	78.8
	2005-06	2172	52010	1131	52.1	92.4
	2006-07	2204	77303	1681	76.2	319
	2007-08	1447	N/A	N/A	N/A	N/A
	2008-09	1086	25119	546	50.3	63
	2009-10	1191	60892	1324	111	89
	2010-11	2160	119040	2588	120	123
	2011-12	1510	45656	993	65.7	33
	2012-13	1086	24249	527	48.5	132
GCDS	2005-06	Incomplete	71561	1084	N/A	101
	2006-07	2362	N/A	N/A	N/A	460
	2007-08	1219	52684	798	57.2	140
	2008-09	1158	16535	250	21.6	35
	2009-10	1525	38142	578	37.9	56
	2010-11	2238	124624	1888	84.4	269
	2011-12	1308	49509	750	57.3	30
	2012-13	1300	26527	402	30.9	256

N/A – Not Available because of missing record

The gauging data in Table 1 contain occasional significant errors which need to be corrected before this project can be completed. For example, G8210012 exhibits two years (2009–10 and 2010–11) with more than 100% runoff which is impossible in a seasonal creek because Gulungul Creek always stops flowing every dry season and hence flow does not persist from one water year to the next. Figure 3 shows a plot of runoff coefficient versus annual rainfall for each water year at GCUS and GCDS. As expected, runoff coefficient increases with annual rainfall. GCUS frequently produces more runoff than GCDS but this is not confined to just low rainfall years when transmission losses between the two stations could reduce runoff at GCDS to less than that at GCUS. Consistency checks of the annual runoff data for each station must be conducted to identify any potential problems with the rating curves which would then need to be corrected before further analyses.

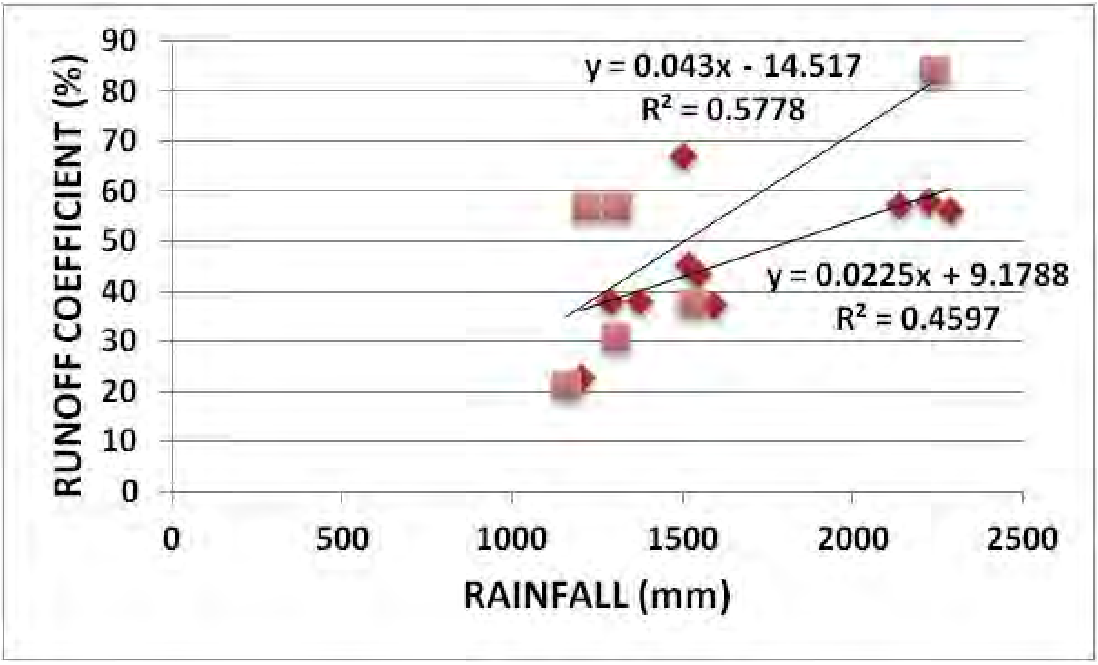

Figure 3 Annual runoff coefficient versus annual rainfall relationships for GCUS (diamonds) and GCDS (squares)

Annual rainfall ranged from a low of 1086 mm at G8210012 in 2008–09 to a maximum of 2362 mm at GCDS in 2007–08. Runoff commences in either November, December or January each year and terminates between April and August, depending on wet season rainfall. At least 21.6% of annual rainfall is converted to runoff each year. The above problem with the runoff data needs to be corrected before further analyses are undertaken..

Flood variability is low with maximum discharge between years at all stations only varying by two orders of magnitude (Table 1). Erskine (1996) reported peak flows that varied by six orders of magnitude on highly flood variable rivers in south-eastern Australia. The largest recorded flood on Gulungul Creek occurred in February–March 2007 (Table 1) and this event was much larger than the second highest event which occurred in February 1980 (Moliere et al. 2007b; Moliere 2005). As a result, the measured morphodynamic responses to specific events are important for assessing the landforming significance of extreme events, such as February 1980 and February–March 2007, on Gulungul Creek.

b) Suspended Sediment Transport

Suspended sediment is operationally defined as the material transported by a river in continuous suspension and retained on a 0.45 µm cellulose nitrate filter (Evans et al. 2004). The pumped water samples were first sieved through a 63 µm sieve before being filtered through the 0.45 µm filter. The relationship between suspended sediment concentration for the silt and clay fraction (>0.45 µm and < 63 µm) and turbidity was then determined at GCUS and GCDS based on the results for the pumped water samples collected by the autosampler for the period 2004–2013 at GCUS and for the period 2004–2010 for GCDS. The results are shown in Figure 4. Both regressions are highly significant as shown by the following equations:

$$Cs = 1.687\ Turb + 1.2374 \qquad (1)$$

where Cs is suspended sediment concentration for the silt and clay fraction (mg/L) and Turb is Turbidity (NTU).

F ratio = 1.473; ϱ <0.00001; Adjusted R^2 = 0.808774; n = 349

$$Cs = 0.4059\ Turb + 1.5913 \qquad (2)$$

F ratio = 715; ϱ < 0.00001; Adjusted R^2 = 0.79855; n = 181

Equation 1 refers to GCUS and equation 2 to GCDS. Clearly there is a strong relationship between suspended sediment silt and clay concentration and turbidity which can then be used to calculate suspended sediment (silt and clay) loads.

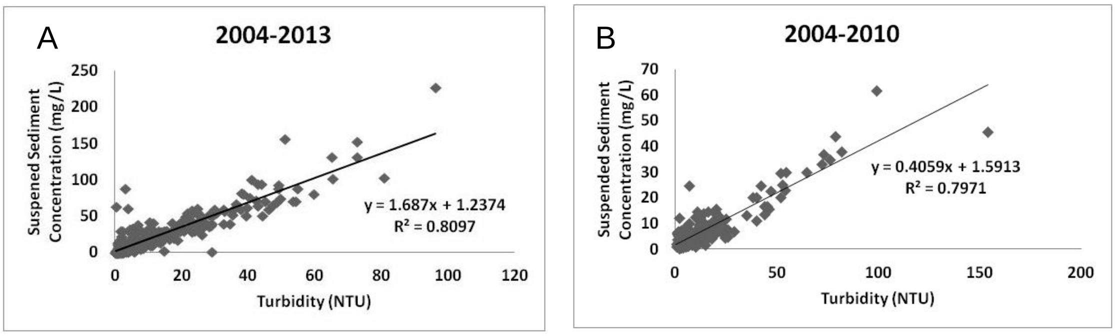

Figure 4 Suspended sediment concentration for the silt and clay fraction versus turbidity relationships for (A) GCUS and (B) GCDS

Each 5- or 6-minute turbidity reading was converted to suspended sediment silt and clay concentration by equation (1) at GCUS and by equation (2) at GCDS. Where there were short gaps in the turbidity record, missing data were estimated by relationships based on discharge. The concentrations were then converted to loads by multiplying by the mean daily discharge for the 5- or 6-minute period and then dividing the result by the number of 5- or 6-minute mean daily discharge intervals per day and summing.

The preliminary results based on current rating curves are included in Table 2. There were significant gaps in the record at GCUS for the 2003–04 and 2006–07 wet seasons and so total loads could not be determined. The large flood of February–March 2007 damaged both gauging stations which were totally reconstructed after the event. Similarly, there were significant gaps in the record for GCDS for the 2003–04 and 2004–05 wet seasons and so total loads could not be determined. Further work will be conducted at both stations for the years with missing data in an attempt to infill all gaps.

The mean annual suspended silt and clay load for GCUS and GCDS for the years with complete data, including estimates, were 469 ± 68 (SE) and 265 ± 67 t/yr, respectively (Table 2). On a unit area basis, these loads convert to 11.7 and 4.0 t/km^2.yr, respectively. Clearly there are significant losses of silt and clay between the two gauging stations for every year for which there is complete data (Table 2). This is expected because there is a floodplain between the two stations and the silt and clay fraction would be deposited as vertical accretion sediments (Wolman & Leopold 1957; Nanson & Croke 1992) on the floodplain surface (see below). This mode of floodplain formation is discussed further below under floodplain morphology and stratigraphy.

The suspended sand fraction (>63 μm and <2 mm) was also measured for many suspended sediment samples. This is essential so that total suspended sediment loads, not just the silt and clay fraction, can be calculated. For GCUS, suspended sand constituted 89.88 ± 0.81 % of the total suspended sediment concentration for 214 samples. For GCDS, suspended sand constituted 89.89 ± 1.29 % of the total suspended sediment concentration for 83 samples. As the suspended sand fraction is 8.88 times greater than the suspended silt and clay fraction at GCUS, the suspended silt and clay load was multiplied by this conversion factor to obtain the suspended sand yield of 4165 t/yr. The total suspended sediment yield at GCUS is the sum of the two or 4634 t/yr. Using the same method at GCDS produces a suspended sand load of 2356 t/yr and a total suspended sediment load of 2621 t/yr. Duggan (1994) measured total suspended sediment load at GCDS (about 100 m downstream of SSD's GCDS) for 1984–85, 1985–86 and 1986–87 and recorded a mean annual load of 2156 t/yr. Her measurement period was much drier than that measured by SSD (mean annual rainfall of 1289 versus 1587 mm/yr) and this alone would account for the small difference between the two values. Our work conclusively proves that suspended silt and clay is a small proportion of the total suspended sediment load and that much greater effort needs to be directed at sampling suspended sand.

When suspended sand is added to the suspended sediment load, a much greater mass is stored on the floodplain between GCUS and GCDS than for the suspended silt and clay fraction (1809 versus 204 t/yr). Clearly this indicates that the floodplain should be dominated by sand.

Depth integrated suspended sediment sampling was conducted at both stations using the USDH48 sampler and the Equal Transit Rate or Equal Width Increment method (Erskine 2005). This is the recommended method of collecting suspended sediment by US Federal Agencies (Erskine 2005) because it provides a mean time- and velocity-weighted concentration for the whole cross section, something that a pump sampler with a single fixed level inlet can never do (Erskine 2005). At GCUS, the mean suspended sand concentration obtained with the USDH48 suspended sediment sampler and the Equal Transit Rate method was 71.76 ± 31.36 % and at GCDS, was 99.75 ± 0.04 %. Further sediment sampling is clearly required.

"No matter how sophisticated, analysis and interpretation can never substitute for well collected data" (Thomas 1985). These words are particularly appropriate for stream sediment loads (Thomas 1985) and further work is clearly required on Gulungul Creek.

c) Bedload

Bedload gaugings using the Helley-Smith pressure difference sampler and the same field procedure as at Jabiluka (Erskine et al. 2011; Erskine & Saynor 2013) were conducted at GCUS and GCDS. The bedload rating relationships for the currently available data are shown in Figure 5. At GCUS, the following regression equation was derived by the method of least squares:

$$Qs = 55.064\ Q - 19.906 \quad (3)$$

where Qs is bedload flux (g/s) and Q is discharge (m3s-1).

F ratio = 22.25; $\varrho = 0.042$; Adjusted $R^2 = 0.876$; n = 4

At GCDS, the relevant regression equation is:

$$Qs = 39.325\ Q - 5.503 \quad (4)$$

F ratio = 2.542; $\varrho = 0.186$; Adjusted $R^2 = 0.236$; n = 6

Table 2 Suspended sediment silt and clay yields (tonnes) for each year of record for the upstream and downstream ***eriss*** gauging stations on Gulungul Creek

Gauging Station	Water Year	Suspended Sediment Silt and Clay Yield (tonnes)	Gauging Station	Water Year	Suspended Sediment Silt and Clay Yield (tonnes)
GCUS	2003-04	>476 N/A	GCDS	2003-04	N/A
	2004-05	528		2004-05	>53 N/A
	2005-06	715		2005-06	420
	2006-07	>522 N/A		2006-07	600*
	2007-08	558		2007-08	302
	2008-09	157		2008-09	78
	2009-10	296		2009-10	185
	2010-11	679		2010-11	372
	2011-12	347		2011-12	176
	2012-13	472		2012-13	145
	Mean	469 ± 68 (SE)		Mean	265 ± 67 (SE)

N/A – Final figure Not Available because of missing record

* - Estimated data for part of the largest flood

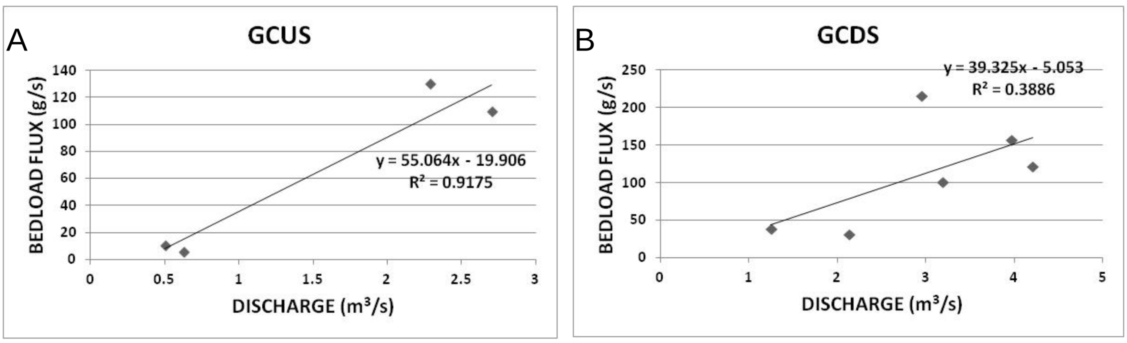

Figure 5 Bedload rating relationships for (A) GCUS and (B) GCDS

Clearly additional data are required, especially for higher discharges. This work is planned for the 2014–15 wet season. Bedload yields have been calculated for GCUS because the regression was the only one significant at $\varrho < 0.05$. For the six water years between 2007 and 2013, the mean annual bedload yield at GCUS was 1318 ± 318 (SE) t/y. Grain size analyses of the bed load show that the material is dominantly sand.

The mean annual bedload yield at GCUS is much greater than the suspended silt and clay yield but is less than the suspended sand yield. Therefore, sand is the dominant sized sediment transported by Gulungul Creek and its measurement needs far greater attention in the future so that the accuracy of sand loads is improved.

d) Cross-sectional and bed-material changes

Crossing (2002) installed 12 permanently marked cross sections (01 to 11, and 13) on Gulungul Creek next to Ranger Mine and, in addition, surveyed the main culvert (cross section 12) on the upstream side of the Arnhem Highway (Figure 2). All of the surveys undertaken at three cross sections, MG08, MG09 and DG10, are shown in Figure 6. Cross section MG08 is located immediately downstream of where the tributary which drains the tailings dam wall enters Gulungul Creek, cross section MG09 is located about 25 m upstream of ERA's downstream gauging station and cross section DG10 is located at the GCDS gauging station (Figure 2).

Cross section MG08 has varied in bed level by 1.1 m between August 2002 and October 2013 (Figure 6A). The deepest scour occurred during the 2012–13 wet season when a late storm caused major flooding during Easter 2013. Up to 1.0 m of bed level variations have occurred at cross section MG09 over the same time period (Figure 6B) with the deepest scour effected by the largest flood on record in February–March 2007 (Moliere et al. 2007b). At GCDS the maximum variation in bed level was only 0.4 m over the same time period (Figure 6C) but was caused by the downstream migration of large-scale transverse sand bars. As the bars reached the section, bed levels rose and as the bars migrated downstream bed levels fell. Levee deposition was also active on at least one bank at cross sections MG08 and MG09 (Figure 6A & B). Although not shown here, about 1 m of bed scour was also recorded at cross sections UG01, UG03, UG04, MG06 and MG07. These sections are usually located at pools. Sections located at runs and riffles usually exhibited much smaller bed level changes due to the movement of large-scale sandy bedforms.

Four of the bed-material samples collected in 2002 at cross sections UG03, MG06, MG09 and DG13 were subjected to full grain size analysis by the hydrometer and sieve method to determine the percentage of each sample < 63 μm. All four samples had less than 5 % <63 μm so all subsequent samples were only subjected to sieving at Φ /2 intervals which is appropriate for sands. The phi (Φ) notation system is used by sedimentologists to describe the grain size of clastic sediment. It is a logarithmic scale in which each grade limit is twice as large as the next smaller grade limit (Folk 1974) and is denoted by:

$$\Phi = -\log_2 d \qquad (5)$$

where d is grain size in mm.

Bulk bed-material samples were not collected every year and all samples have not been analysed for grain size as yet. Preliminary analysis indicates that sample masses were large enough to obtain reproducible measures of the grain size distribution. The available data indicate that bed material of Gulungul Creek at the permanently marked cross sections is characterised in terms of the graphic grain statistics of Folk (1974) as mesokurtic, near symmetrical, moderately well sorted to well sorted, medium to coarse sand. When all the data have been processed, changes in grain size statistics over time will be determined.

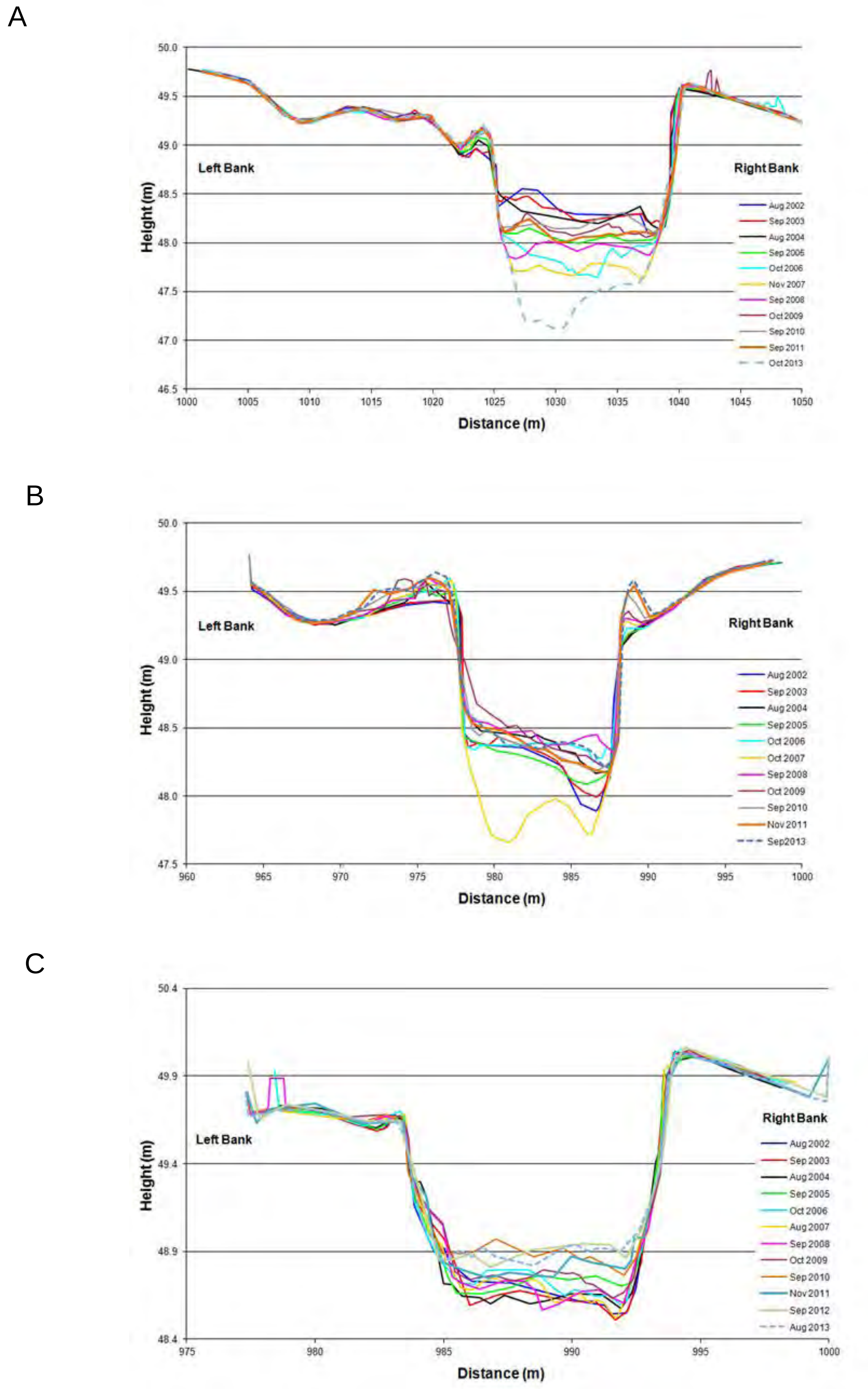

Figure 6 Cross-sectional channel changes at (A) MG08, (B) MG09 and (C) DG 10. See Fig 5 for locations.

e) Scour chain results

The purpose of scour chains is to measure the total depth of scour and fill during each wet season. The depth from the bed surface to the top of the chain is the depth of scour and the depth from the top of the chain to the bed surface is the subsequent fill. An excavated scour chain is show in figure 7. A total of 15 scour chains were installed in the late dry season of 2002 at six cross sections (two chains on cross sections UG02, DG10 and DG11, and three chains on cross sections UG03, MG06, MG08). Attempts to recover scour chains were made in the late dry season of 2003, 2004, 2005, 2006, 2007 and 2008. The results are presented in Saynor et al. (2005), Saynor & Smith (2006) and Saynor et al. (2010). In summary, minimum scour was 50 mm at cross section UG02 chain 2 during the 2003–04 wet season (Saynor et al. 2005) and the minimum fill was 0 mm at cross section UG03 chain 1 during the 2005–06 wet season (Saynor et al. 2010). The maximum scour was 870 mm at cross section MG06 chain 1 during the 2004–05 wet season (Saynor & Smith 2006) and the maximum fill was 930 mm at cross section MG06 chain 1 during the 2003–04 wet season (Saynor et al. 2005). Further analysis of the scour chain results is required.

Figure 7 Excavated scour chain 1 at cross section DG11 on 12 December 2007. Flow direction is from left to right.

f) Riparian vegetation

The repeated surveys of the channel cross sections discussed above show that the banks of Gulungul Creek are stable over the survey period (12 years) while the same surveys and scour chains demonstrate that the river bed can vary in elevation during a wet season by up to 1.1 m. Clearly wet season scour has not been deep enough to increase the critical bank height above that required to maintain stability. This is surprising given that scour effectively doubles bank height although the weight of water in the channel at the time of scour would help support the bank. *Melaleuca* spp. are common trees on the banks, floodplain and in the bed of Gulungul Creek (Figure 8). Their trunks reduce flow velocity and protect the banks; their roots bind bank sediment and form resistant root mats which also protect the banks; large roots also form steps across the river bed which prevent bed scour; trees senesce and supply large wood to the channel which perform many stabilising functions (Erskine et al. 2012b).

Figure 8 (A) *Melaleuca argentea* growing in the channel and on the banks of Gulungul Creek at cross section UG04 on 30 August 2002. (B) *M. argentea* growing in the bed of Gulungul Creek at the control for G8210012 on 19 December 2012. Roots form log steps in the bed. (C) *M. viridiflora* growing in a floodplain depression on lower Gulungul Creek on 4 June 2002. (D) Regenerating *M. argentea* growing on the banks of Gulungul Creek downstream of GCDS on 25 September 2002.

We now intend to investigate the intrusion of various *Melaleuca* spp. into the channel of Gulungul Creek to determine if there is a threshold encroachment amount before bank erosion occurs. Erskine et al. (2009) found that oak benches on the sand-bedded Widden Brook in the Hunter Valley, NSW encroached 0.2 of a channel width and did not produce bank erosion. Bennet et al. (2008) concluded that 0.5 of a channel width encroachment initiated bank erosion of the opposite bank. Therefore, we assume that a value between 0.2 and 0.5 of a channel width induces bank erosion.

We also plan to determine if the apparently even-aged stands of *Melaleuca* spp. on the banks of Gulungul Creek represent episodic recruitment. Tree age will be approximated by diameter at breast height over bark (dbhob) (at 1.3 m above ground) so that the frequency distribution of dbhob will be used as a surrogate of recruitment age (Erskine et al. 2009). Processes of recruitment will also be investigated, where possible.

The role of native grasses in stabilising river banks needs evaluation following the finding of Erskine et al. (2012a) that grasses can be effective sand traps on sand-bed streams because only relatively small changes in flow velocity are needed to cause deposition. However, many of the native grasses on Gulungul Creek have longer and wider stems than those investigated by Erskine et al. (2012a) on Widden Brook. Furthermore, plant functional traits such as rhizomes need to be assessed on Gulungul Creek because they can be responsible for developing dense grass swards that resist erosion (Erskine et al. 2012a).

g) Floodplain Morphology and Stratigraphy

Professor Erskine completed a special studies programme from The University of Newcastle at ***eriss*** during the first half of 2010. As part of that programme, Professor Erskine and Dr Saynor completed some detailed work on floodplain morphology and stratigraphy of Gulungul Creek between the upstream gauging station, GCUS, and the Arnhem Highway (Erskine & Saynor 2014). Crossing (2002) has also described the channel and floodplain system for the same reach of Gulungul Creek based on her understanding of the geomorphology at that time.

Gulungul Creek has incised a trench into river terraces and the deeply weathered Koolpinyah surface (Hays 1967). The channel network comprises one main, slightly sinuous channel with anabranches at various stages of development. Some anabranches have a sand-bed, some are well vegetated, some are combinations of both and some are discontinuous. The individual channels are usually separated by floodplain. Figure 2 does not show these anabranches because there is no image that we have found to date that clearly depicts every channel. The sand-bed channel dissipates downstream of the Baralil Creek junction but has formed a backflow billabong (Hart & McGregor 1980) on the lower part of Baralil Creek (Figure 9). Sand aggradation along Gulungul Creek has extended from the Arnhem escarpment to downstream of Baralil Creek and has been significant enough to dam the tributary, Baralil Creek (Figure 9). However, this sand finishes in a terminal floodout before reaching Gulungul Billabong (Figure 9). Clearly sand generation rates from the Arnhem Plateau and the Koolpinyah surface throughout the Holocene in the Gulungul catchment have been low because a continuous sand-bed channel from the Arnhem escarpment to Magela Creek has still not formed. Further work is continuing on the floodplain morphology of Gulungul Creek.

The floodplain sediments on cross sections UG01 and DG10 (Figure 5) were manually cored and described by Erskine & Saynor (2014) in 2010. Consistent with the results of Nanson et al. (1993) for a single cross section of Gulungul floodplain near cross section DG09 (Figure 2), the contemporary floodplain (labelled 'floodplain' in Figure 10) consists of a thin veneer of sediment over an eroded stump of a Pleistocene floodplain, called a buried terrace in Figure 10. The detailed sediment descriptions are contained in Erskine and Saynor (2014).

At cross section UG01 (Figure 10), there is an inset bench fill next to the current channel which comprises a fining upwards sequence of coarse sand to medium sand to fine-medium sand to loamy fine-medium sand to humic fine sandy loam. This represents a phase of channel contraction following initial expansion. The floodplain on the left bank consists of a texturally variable unit of humic loam fine sandy, loamy fine sand, loamy medium sand, fine sandy loam, light sandy clay loam and sandy clay loam. The floodplain overlies an eroded stump of a Pleistocene terrace which is characterised by mottled slightly pebbly clayey medium-coarse sand and sandy loam. The pebble fraction consists of rounded quartz, rounded ironstone and iron concretions. The river terrace on the left bank overlies weathered bedrock and comprises a fining upwards sequence of clayey medium sand to loamy medium sand to loamy fine-medium sand. The basal clayey medium sand is mottled and contains hard red iron segregations. A possible higher river terrace is also present above the described terrace.

At cross section DG10 (Figure 10), there are a series of abandoned channels on the right bank floodplain with channel fills which fine upwards from medium-coarse sand to loamy medium sand to humic loamy sand. A high water table prevented augering any deeper in the medium-coarse sand. The remainder of the floodplain was texturally variable ranging from humic and fibric loam fine sandy to fine sandy loam to loamy fine sand to fine-medium sand to clayey fine-medium sand. The surface of the floodplain was veneered by a variable layer 5 to 10 cm thick of recent overbank deposits of fine-medium sand, loamy fine-medium sand and humic loamy fine sand. This sediment was deposited by overbank flows which occur every wet season and consists dominantly of sandy material, as expected from the results of the sediment budgetting carried out above. Their sandy nature reflects the predominance of sand in the sediment load. Again the floodplain overlied an eroded stump of a Pleistocence river terrace which consisted of mottled clayey medium sand and fine-medium sand. An extensive high river terrace is present on the right bank and comprises a deep (at least 2.7 m deep) fill of mottled fine-medium sand and clayey fine-medium sand. There are many soft iron segregations of pebble size at depth.

Figure 9 An aerial photograph of Gulungul Creek around the Arnhem Highway in October 1982. Flow direction is from the top of the photo to the bottom.

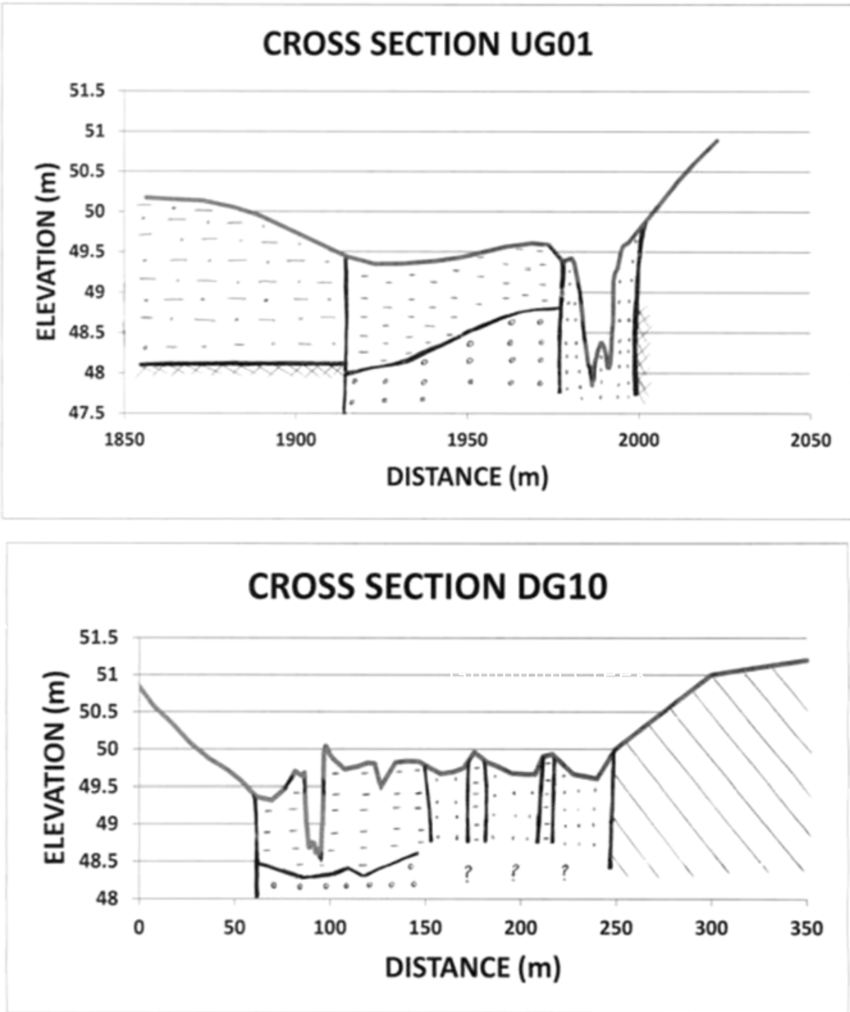

Figure 10 Generalised floodplain stratigraphy at cross sections (A) UG01 and (B) DG10. See Figure 2 for location of cross sections.

Nanson et al. (1993) used thermoluminescence to date the valley fill at one cross section on Gulungul Creek. They showed that thin Holocene sediments veneered the floodplain which was composed of an upper sandy unit which was late Pleistocene in age and which in turn capped a deeper muddy sand and gravel unit up to 79.9 ± 8.5 ky old. A gravel lag deposit blanketed the bedrock basement.

Floodplains form by up to six geomorphic processes (Nanson & Croke 1992) with lateral accretion (coalescing point bars) often being dominant. Our preliminary data for the Gulungul floodplain indicates that abandoned channel accretion combined with overbank deposition and bench accretion are impotant. Bench accretion was a significant omission from the floodplain formation processes listed by Nanson & Croke (1992). Nevertheless, the Gulungul floodplain consists of a stripped or eroded late Pleistocene terrace that has been capped by Holocene overbank deposits, abandoned channel deposits and and bench deposits.

Steps for completion

Apart from the annual data collection and laboratory analysis, further data analyses are required but have been limited by lack of staff resources and the higher priority allocated to the continuous monitoring and the trial landform programmes. Gulungul Creek gauging stations will continue to collect data during the 2013–2014 wet season for regulation purposes and the dry season cross section surveys and bulk bed material samples will be collected in 2014. Missing data in the gauge records need to be estimated and the consistency of the hydrological data must be checked to identify any problems with the rating curves. Furthermore, additional suspended sediment sampling and bedload gaugings are required.

Work will proceed on this project as staff resources allow. It is planned to complete work on riparian vegetation during the dry season of 2014 with Dr Anita Chalmers (School of Environmental and Life Sciences, The University of Newcastle) who will visit in June 2014 to provide additional expertise on plant functional traits of riparian vegetation and their role in stabilising rivers and floodplains. This project will receive higher priority when the data for the first five years of the trial landform project have been analysed and published.

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Analysis of landscape change on the Ranger site pre-mine using historical aerial photography

RE Bartolo, T Whiteside & A Esparon

Introduction

It is important that closure criteria and our understanding of analogue sites for the Ranger mine site are developed in the context of temporal change in the landscape and disturbance of the site. Analysis of historical aerial photography for the site pre-mine operations (1981) will show if and how the landscape has changed over multiple temporal scales (e.g. annual [1978–1980], decadal [1950, 1964, 1976]). Information will be extracted from digitised, georeferenced and mosaiced aerial photography using spatial analysis. Examples of features to be extracted include: vegetation communities, canopy cover (from larger scale aerial photos), and drainage. Once these features have been extracted from the multi-temporal data, any changes measured will be reviewed in relation to disturbance (e.g. fire and feral animals).

The current analogue plots (see KKN 2.5.2- *Use of vegetation analogues to guide planning for rehabilitation of the Ranger minesite*) provide some of the information required for informing terrestrial vegetation criteria, but it is not the whole picture. A major limitation of all ground-based surveys is that only small samples in the landscape are taken. A landscape approach, such as deriving measures from remote sensing, provides a total dataset to compare rehabilitated mine sites with the surrounding environment.

The objective of this project is to measure and assess landscape change on the Ranger site pre-mine using historical aerial photography 1950–1981.The specific aims to meet this objective are:

- To digitize and georeference key epochs of aerial photography (selected based on scale, film type, availability and temporal resolution) of Ranger pre-operations (1981).
- Map biophysical parameters and landscape disturbance that can be identified in the aerial photography and analyse change in the landscape over multiple temporal scales.
- Inform closure criteria and characterisation of analogue sites.

The outputs from this project will provide an envelope for natural variability in setting closure criteria (rather than a single measure) and provide a continuous representation of the landscape rather than discrete sampling points both spatially and temporally.

Progress to date

Orthomosaics of aerial photography over the site have been obtained for the following years: 1950; 1964; 1976; 1978; 1979; and 1980-81. Figure 1 shows a sub sample of some of the mosaics. Photos were scanned to a spatial resolution of 1m so that the images can be compared despite differing scales. Table 1 provides a summary of the photo details.

Photos were all scanned at 1600dpi in TIFF (Tagged Image File Format) format. This has resulted in varying grain size. The mosaic for 1980–81 is captured over two dates.

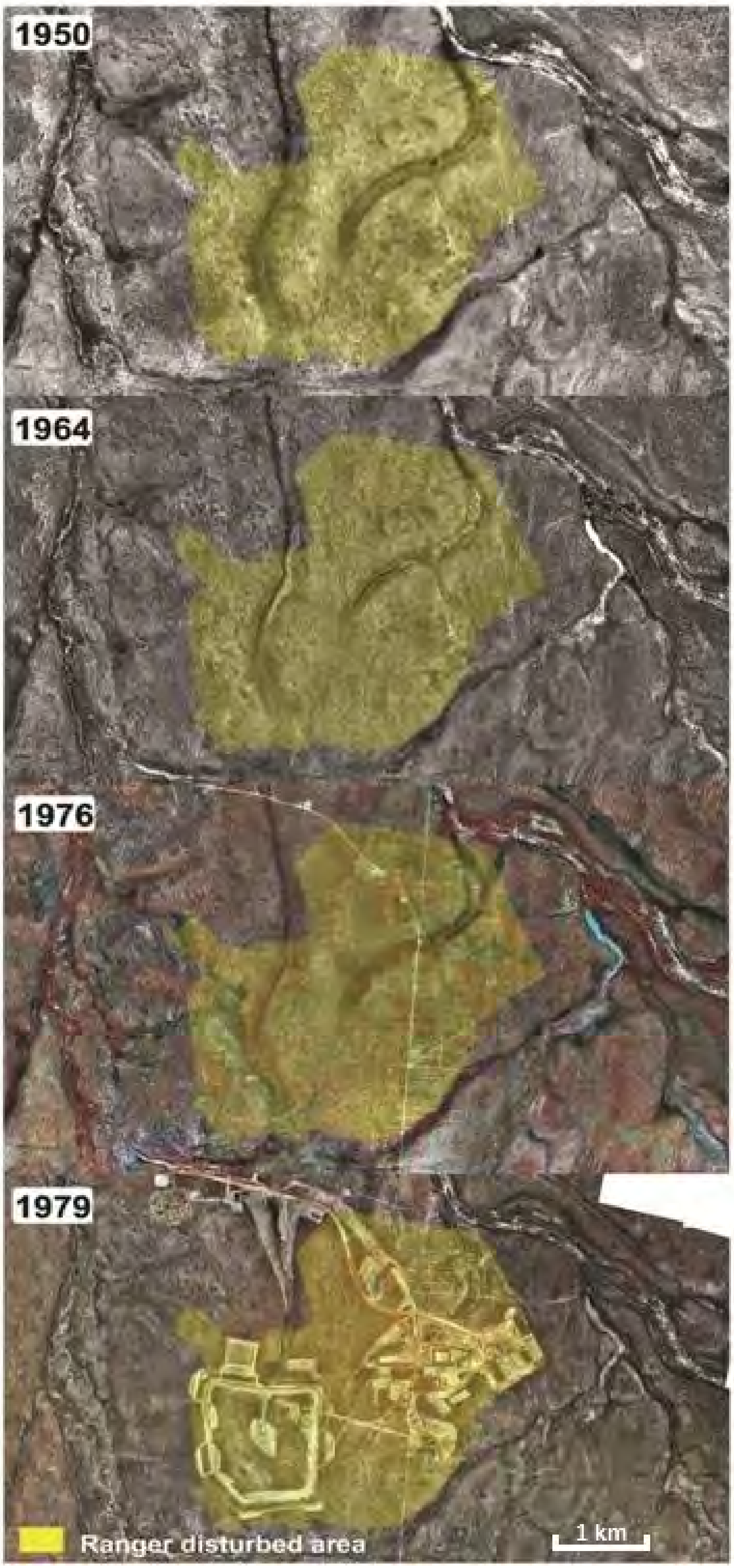

Figure 1 Aerial photography mosaics for the Ranger site.

Table 1 Details of aerial photography

Epoch	Scale	Film Type	Grain size (cm)
1950	1:50000	Black and White	80
1964	1:16000	Black and White	25
1976	1:25000	Colour Infrared	40
1978	1:25000	Colour	40
1979	1:10000	Colour	16
1980	1:25000	Colour Infrared	40
1981	1:25000	Colour	40

Steps for completion

The difference in grain size between photos of different years needs to be investigated to determine whether this will impact analysis. As the differences are in the order of decimetres the difference in grain size may not be an issue for analysis and comparison across years. However if it is, the photo mosaics will be resampled to a common grain size. Methods for analysis are currently being investigated including the use of wavelet transformations which should be able to isolate vegetation in the 1950 data which is of a poorer quality and smaller scale (1:50000). The photos will be processed and change in the landscape will be analysed over multiple temporal scales. The results will be compared to the landscape change studies for Kakadu National Park (Banfai & Bowman 2006, Banfai et al. 2007, Lehmann et al. 2008, Lehmann et al. 2009, Bowman et al. 2010), which have shown that change is locally variable in terms of scale. The results will be reported to the Closure Criteria Working Group.

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Solute and sediment losses from the trial landform

M Saynor, K Turner & W Erskine

Introduction

With the current mine lease at Energy Resources Australia Pty Ltd's (ERA) Ranger uranium mine due to expire in 2021, there is an increasing focus on key aspects of progressive rehabilitation. ERA is establishing a mine closure plan with a focus on reshaping and revegetating the final mine landform so that it may eventually be handed back to the traditional owners for possible incorporation into Kakadu National Park. It is important that the final landform resembles the surrounding landscape, is radiologically stable, exhibits erosion characteristics similar to the surrounding environment, and that it acts as a functional containment structure for the mine tailings which must be physically isolated from the environment for 10,000 years post closure.

A further challenge for rehabilitation will be to ensure that the Ranger Project area does not become a significant future source of elevated sediments and solutes to surrounding areas. Pits 1 and 3 are designated containment areas for mine tailings as well as the brine residue from the brine concentrator. While the final landform will include various engineered solutions to prevent expression of contaminated waters from Pit 3, potential for mobilisation of sediments and solutes via surface water runoff and seepage through the landform remains. Further work is required to understand the processes associated with sediment and solute generation and transport.

In collaboration with the ERA, SSD is undertaking research to determine the optimal design and composition of the final, rehabilitated landform for the Ranger Project Area. This work includes measurement of sediment and solute loads generated and transported from the landform during rainfall events (Saynor et al. 2011). These data are also used for validation of predictive computer modelling of the long-term geomorphic behaviour of the proposed landform designs for the Ranger minesite, also carried out by the SSD (Lowry 2013; Saynor et al. 2012a).

A trial landform of approximately 8 ha was constructed in late 2008 and early 2009 adjacent to the north-western wall of the tailings storage facility (TSF) at Ranger mine (Figure 1). The trial landform was designed to assess the effect on sediment and solute generation of:

1. two types of potential capping material: (i) waste rock, and (ii) waste rock blended with approximately 30% fine-grained lateritic material to a depth of 2 m.
2. two types of potential planting methods: i) direct seeding and ii) tube-stock.

The trial landform was segmented into six main treatment areas and the surface was ripped on the contour before planting was carried out (Saynor et al. 2009). SSD's erosion plots did not investigate sediment and solute generation on waste rock blended with approximately 30% fine-grained weathered horizon material (lateritic material) to a depth of 5 m.

Since construction, measurements have been carried out to assess the generation and transport of sediments and solutes, and ability to sustain growth of plant species native to the region. This report provides the first preliminary assessment of the generation and transport of solutes on plot 1 since monitoring commenced, as well as an update on the hydrology and bedload yields, which have been reported in previous years.

Methods

Erosion plots were installed during the 2009 dry season (Saynor et al. 2009) on each of four treatment areas by physically isolating an approximately 30 x 30 m area (indicated by the cross hatched squares in Figure 1) from the surrounding landform surface by raised damp course and concrete borders on three sides and an open PVC drain on the down-slope side. Plots 1 and 4 were planted with tube-stock in March 2009 with infill planting to replace dead specimens in January 2010 (Daws & Gellert 2011). 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 (Gellert 2012).

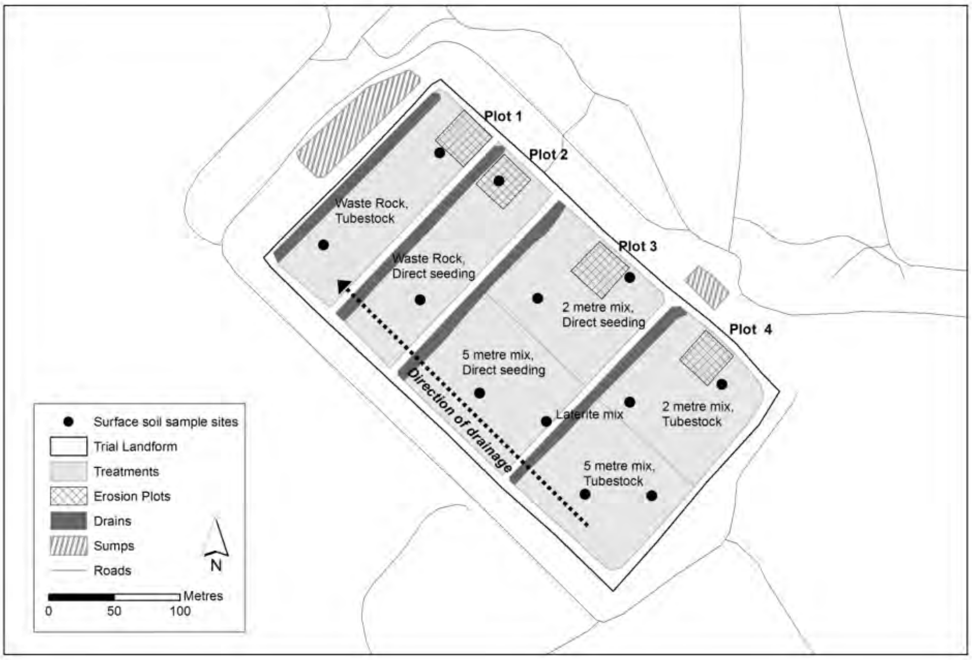

Figure 1 Layout of the erosion plots on the trial landform

Each erosion plot was instrumented with a range of sensors that were described in detail in Saynor et al. (2012b). In summary, these included: rectangular broad-crested (RBC) flume to accurately determine discharge; a tipping bucket rain gauge, a primary shaft encoder with a secondary pressure transducer to measure stage height; a turbidity probe from which suspended sediment concentration could be determined; electrical conductivity (EC) 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 solutes in the runoff; an automatic pump sampler to collect event based water samples triggered by predetermined changes in EC and turbidity readings; and a data logger with mobile phone telemetry connection (Figure 2).

Figure 2 Runoff through the flume on the trial landform erosion Plot 3 during a storm event

The water level above the crest of the flume was used to calculate discharge which was then used with time-integrated concentration data to calculate solute and sediment loads.

The samples triggered by EC changes were analysed in the laboratory for general water quality (EC and pH) as well as concentrations of dissolved (<0.45 μm) trace metals (uranium, manganese, aluminium, iron, zinc, copper, barium, nickel, silica and lead using ICP-MS) and major ions (magnesium, sodium, potassium, calcium, chloride and sulphate using ICP-OES).

The samples triggered by turbidity changes were analysed in the laboratory for turbidity and suspended sediment concentration. The suspended sediment concentration was measured by filtering a standard volume of sample through a 0.45 μm filter of known weight and then determining the oven dry weight of the sediment.

Bedload samples were collected at weekly to monthly intervals during each wet season, depending on the magnitude of runoff events and staff availability. The samples were processed in the laboratory by weighing (after oven drying) as well as measuring the particle size using the Wentworth size fractions for gravel (> 2 mm), sand (63 μm to 2 mm) and silt and clay (< 63 μm). Each sample was sieved to determine the sediment fractions.

Rainfall and runoff

Overview

Data are presented for a 'water year', from September to August. Preliminary sediment and solute losses from the four erosion plots were presented for the first wet season of monitoring in Saynor et al. (2011). Rainfall for all four plots and runoff from erosion Plot 1 are reported here. Plot 1 is the only plot for which runoff has been calculated to date for all four water years.

Rainfall and runoff results and discussion

The rainfall data for each plot and each water year are contained in Table 1 while 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 four 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
2012–13	1283	1274	1260	1264	1274 ± 5

Table 2 Rainfall and runoff data for erosion plot 1 on the trial landform for the four 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
2012–13	72.8	92	112858	122	8.1

Mean annual rainfall at Jabiru Airport (Station No. 014198, located 2.3 km from the trial landform) is 1578 mm (Bureau of Meteorology 2013). The annual rainfall for the 2012–13 water year on the trial landform was the lowest for the four years of study (Table 1) and was much lower than the mean annual rainfall at Jabiru airport. The water year, 2010–11 was the only year of above average rainfall. A storm occurred on 22 February 2011 which had a return period well in excess of 1:100 years for durations of 1- to 3 hours. This storm damaged plot borders.

The number of discrete runoff events for 2012–13 is lower than for the three previous years because of the lower rainfall. Similarly the number of runoff events was greatest in the wettest year (2010–11). Unusually, annual runoff was lowest in the 2009–10 water year than in both the drier 2011–12 year and the much drier 2012–13 year (Table 2). This was the result of the infilling with water of the initially empty pore space in the waste rock (Plots 1 and 2) and in the waste rock and laterite (Plots 3 and 4) 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, the runoff coefficient for plots is much less than for larger catchments in the Alligator Rivers Region (ARR). Areas up to about 1 km^2 behave as if they are drylands in the seasonally wet tropics.

There is a curvilinear relationship between event rainfall and event runoff over the full range of rainfall for all four years for Plot 1. Figure 3 shows this curvilinear relationship between total event rainfall and runoff for all 92 events on Plot 1 for 2012–13. When event rainfall exceeds 30 mm there is proportionately greater runoff than for smaller events (Figure 3). 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 the surface storage, hence generating runoff from the whole plot surface.

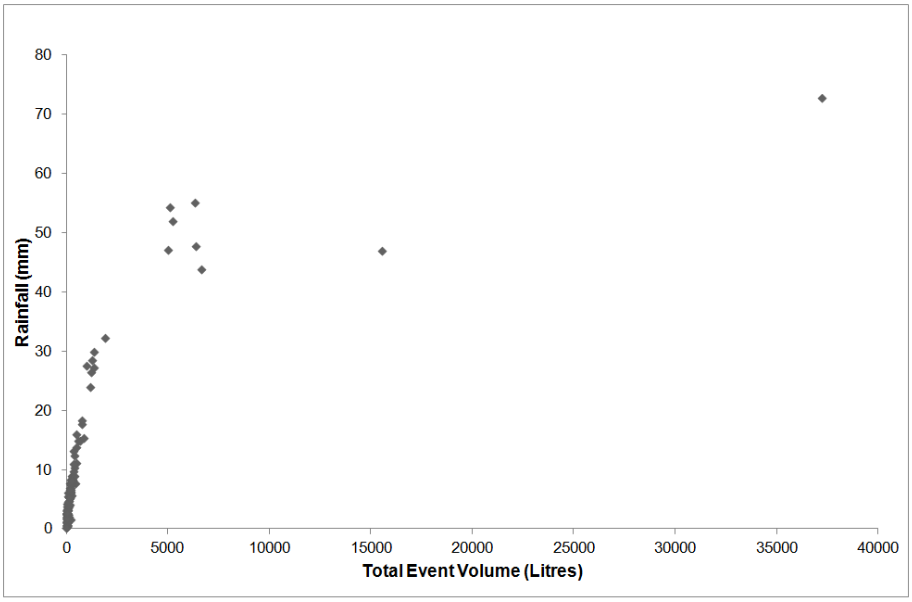

Figure 3 Relationship between total event rainfall and runoff for Plot 1 for every runoff event in the 2012–13 water year

Results and discussion of Suspended Sediment and Bedload

Suspended sediment

Since monitoring of the trial landform commenced, a large number of water samples have been collected based on turbidity increases measured in the flume of each plot. These samples have been analysed in the laboratory for turbidity and suspended sediment concentration. The associated data have not yet been analysed because of the difficulty of establishing a relationship between turbidity and suspended sediment concentration where there are large amounts of platey mica particles in the runoff. When this relationship is determined, the results will be reported.

Bedload

The bedload yields recorded for each plot for each water year are shown in Table 3. The data clearly show that the annual bedload yield for each plot has declined progressively since construction. Time since construction has had a much greater effect on annual bedload yields than cover material type and development of vegetation because yields have declined exponentially over time on all plots, except for plot 2 for 2012–13. Sediment yields for major land disturbances, such as construction or landslides, are usually characterised by an initial pulse followed by a rapid decline (Duggan 1994; Erskine & Saynor 2000). This is true for the trial landform annual bedload yield, which is characterised by an exponential decline in annual bedload yield over the four years since construction (Figure 4).

Table 3 Annual bedload yields ((t/km^2.yr) for each plot for each year of measurement

Water year	Plot 1	Plot 2	Plot 3	Plot 4	Mean Annual Bedload Yield ± Standard Error
2009–10	106	147	111	143	127 ± 11
2010–11	59	113	54	56	71 ± 14
2011–12	34	48	38	15	34 ± 7
2012–13	28	50	14	14	26 ± 9

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 as the greatest rainfall occurred in the second year (Table 3). Using the average rainfall per rain day as an index of rainfall intensity, the values for the four years were 14, 15, 12 and 10 mm/d for the 2009–10, 2010–11, 2011–12 and 2012–13 water years, respectively. The 2010–11 wet season was not only the wettest season but also had the most intense rainfall, further supporting the fact that rainfall is not a key driver for annual bedload yield on the trial landform.

The highest annual bedload yields were always generated from Plot 2 (Table 3). While it is still not clear why this happens, shallow rip lines dominate the lower part of Plot 2, resulting in direct connection of diffuse overland flow with the down slope plot drain.

In the fourth year after construction, a clear signature of the effect of vegetation establishment is evident in the annual bedload yields. The two plots (1 and 4), originally planted with tube stock, now have the greatest average tube stock heights (2.77 m for Plot 1 and 2.64 m for Plot 4) and both show lower bedload yields than plots 2 and 3 initially planted by direct seeding (0.3 m for Plot 2 and 1.04 m for Plot 3) followed a year later by infill planting with tube stock. Plot 4 has the lowest yield because it has also been invaded by weeds which densely cover about half of the plot and mitigate against erosion.

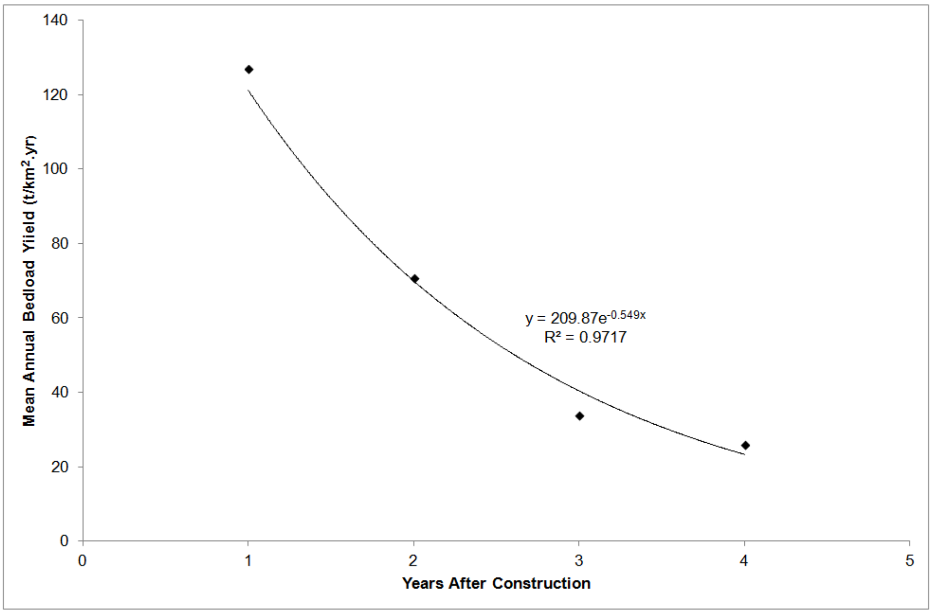

Figure 4 Exponential decrease in mean annual bedload yield with time since construction for the four erosion plots on the trial landform

Particle size analysis

Table 4 shows the particle size of the bedload for each year of measurement. For Plots 1, 2 and 3 the sand fraction has the highest percentage (all over 50%). On Plot 4 the sand fraction was higher for years 2009–2010 and 2012–2013 and was essentially the same as for gravel for the other two years. This indicates the importance of the sand fraction and shows that it is the main erosion product in the early years after construction of a rehabilitated landform. The gravel fraction makes up the next highest percentage with little silt and clay contained in the bedload. Surface armouring of coarse gravel has occurred by the washing out of silt and clay from the ground surface.

Table 4 Annual yield of bedload fractions as a percentage of total bedload yield for each plot for each year of measurement

Plot	EP1			EP2			EP3			EP4		
Water year	***% gravel***	***% sand***	***% silt and clay***	***% gravel***	***% sand***	***% silt and clay***	***% gravel***	***% sand***	***% silt and clay***	***% gravel***	***% sand***	***% silt and clay***
2009/10	34	60	6	34	55	11	37	59	4	35	61	4
2010/11	33	64	3	40	55	5	46	53	1	50	49	1
2011/12	44	53	3	42	55	3	47	52	1	50	49	1
2012/13	40	57	3	31	65	4	45	54	1	45	54	1

Solute generation and transport

The waste rock used to construct the trial landform is largely mica schist and originates from Ranger Pit 3 (Map 2). Waste rock is the most abundant substrate for capping the final rehabilitated landform. Previous work on the chemical characterisation of Ranger mine waste rock (largely chlorite schist) has shown its potential to generate soluble manganese, magnesium, sulphate, uranium, calcium, aluminium, iron and potassium

(Hollingsworth et al. 2003; East et al. 1994; Taylor et al. 1996). These solutes are mobilised during rainfall events and are transported via both surface runoff and seepage pathways.

Typically, the rate of solute generation from waste rock declines over a period of years as the source becomes exhausted, and eventually stable, with inert end-products remaining (Hollinsworth et al. 2003). An intra-seasonal decline is also observed, with high solute concentrations causing a 'first flush' effect at the commencement of the wet season followed by a subsequent decline in concentration towards the end of the wet season. The first flush effect is attributed partly to the fact that rainwater in the region has a lower pH during the early wet season (compared to the late wet season) and is more effective at remobilising solutes and also to the fact that epsomite ($MgSO_4.7H_2O$), which is the dominant soluble salt formed as a result of weathering of Ranger waste rock, accumulates at the surface during dry periods and is flushed away during the first rains (Noller et al. 1990, East et al. 1994). This behaviour would be true for other major salts present.

It is important to monitor the quality of the surface runoff from the Ranger trial landform for a number of years to gain an understanding of the types of solutes present, their concentrations and exported loads (yields) and their behaviour over time. This will lead to more accurate assessments of the environmental significance of the surface runoff from the trial landform and ultimately, from the final Ranger rehabilitated landform. This paper presents findings for the first three years of monitoring, focussing on chemical characterisation of the surface runoff and the inter- and intra-annual variation in runoff solute concentrations and loads. Data for Plot 1 only are presented, as the quality of the EC and solute data cannot be verified until the discharge data have been determined. Furthermore, at the time that this work was completed, the discharge data for Plot 1 for the 2012–13 water year had not been processed.

Solute results and discussion

Total dissolved solids load

The relationship between EC and TDS is typically linear with a slope between 0.5 and 0.9, depending on the solution matrix . The slope can be used as a conversion factor and a value of 0.67 is typically used for naturally occurring freshwater with an EC < 100 µS/cm if the actual relationship is unknown. The EC-TDS relationship for the trial landform is currently being investigated, however until a reliable relationship is determined the TDS has been calculated by applying the assumed standard conversion coefficient of 0.67 to the time-series EC data. Table 5 shows the total rainfall and runoff and the loads of total dissolved solids (TDS) for Plot 1 for the first three water years. The total dissolved solid loads were calculated using the method described by Walling (1984), and shown by Equation 4.1, where t is time, i is a defined period of time, $[TDS]$ is instantaneous TDS concentration (mg/L) estimated using the EC data and Q is instantaneous discharge (L/s).

$$\text{total load} = \int_{t=0}^{t=i} [TDS]\, Q\, dt \qquad (4.1)$$

By multiplying the TDS concentration by the corresponding discharge for each time increment and then summing over time, the total mass of TDS over a water year can be calculated. The TDS load/unit runoff (kg/m^3) was calculated by dividing the total annual TDS load by the total annual discharge. This normalises the TDS load data for the seasonal variation in runoff between wet seasons.

Table 5 Plot 1 annual rainfall, runoff volume and TDS load data from 2009 to 2012

Wet Season	Rainfall (mm)	Runoff (m^3)	TDS Load (kg)	TDS load/ unit runoff (kg/m^3)
2009–10	1528	72.17	0.74	0.01
2010–11	2224	273.4	1.19	0.004
2011–12	1509	94.52	0.49	0.005

While the second wet season (2010–11) had the highest rainfall, runoff and TDS load, the TDS Load/Unit Runoff was highest for the first wet season (2009–10) at 0.01 kg/m^3, after which the value halved and stabilised at around 0.005 kg/m^3. This initial flush of solutes was expected to occur during the first wet season as the freshly-exposed rocks came into contact with the slightly acidic rainwater and the various salts present dissolved and (re)mobilized the trace elements. These initial results are somewhat similar to the sediment transport observations described earlier.

Inter-seasonal variation

The mean annual concentration (and standard error) of key chemical analytes are presented in Table 6. Mann-Whitney tests (a non-parametric test used to test the equality of two annual medians) showed that since monitoring commenced, Cu and Ni concentrations have increased significantly each season and Ca, K, Na, SO_4, Mn, Si and Al concentrations have decreased significantly ($P<0.05$). There were no significant annual changes in the concentrations of Pb, Zn, U, Mg and Ba. These data show that the post-construction surface of the landform contained soluble salts and particularly fast weathering sources of Si and Al.

Table 6 also shows the mean concentrations of analytes measured at the upstream (control) sites on Magela and Gulungul Creeks between the 2009–10 and the 2011–12 wet seasons. The elevated concentration of SO_4, Ca, Na, K, Ba and U in the landform runoff, compared to the background levels in the creeks, show that the final landform may contribute significant amounts of these analytes to the surrounding waters during the early phase of rehabilitation. The surface water quality monitoring programme currently carried out by SSD measures total concentrations of these analytes plus discharge in Magela and Gulungul Creeks on a fortnightly basis, except for Ba, which may need to be incorporated into the routine monitoring programme. Previous Supervising Scientist reports describe observation of a significant Mg signal in Magela Creek in relation to input of mine-derived waters from RP1 and the Corridor Creek catchment (Turner et al. 2012). However, the mean concentrations of Mg measured in the trial landform runoff are similar to background levels in the creeks. While the surface runoff is low in Mg it is likely that the higher Mg yields would be expressed in the seepage from the TLF, where the rainwater has a longer residence period in contact with the waste rock. The occurrence of efflorescence (precipitation of epsomite) frequently occurs at the toe of the tailings storage containment walls, which are also constructed from waste rock. ERA has collected data from two seepage collection sumps at the base of the trial landform but this data has not been assessed to date.

Intra-annual variation

The variation in major ion and U concentrations for Plot 1 within each water year is displayed in Figure 5 against cumulative runoff (which eliminates the effects of seasonal variation in runoff). As expected, a number of analytes (U, K, Ca, Mg, Na and SO_4) were inversely related to cumulative runoff volume, with higher concentrations measured at the beginning of the season (in the first 20 m^3 of runoff) compared to the end of the season (Figure 5). For K, Na and SO_4, this first flush effect was most obvious during the 2009–10 and 2010–11 wet seasons and by the 2011–12 wet season the relationship between concentrations and discharge was not as pronounced, confirming exhaustion of the key sources of these ions. The exhaustion is not as obvious for Ca, Mg and U, with the concentrations measured in the 2011–12 wet season spanning a similar range to previous wet seasons. The remaining analytes (Al, Ba, Ni, Si, Pb, Cu and Zn) did not exhibit any relationship to discharge over the duration of a wet season.

These results offer some insight into the intra-annual variability of each analyte. However, for the majority of runoff events, less than three good quality water samples were collected, which limits the extent to which the chemical variation during individual events can be characterised. There are no significant relationships between analyte concentration and event discharge (m^3/s). Concerns associated with collection of good quality water samples for chemical analysis will be rectified in future wet seasons by altering the control elements of the datalogger programme as well as by the physical sensor mounts on the trial landform to ensure that more samples are collected during runoff events.

Table 6 Annual mean concentrations in trial landform surface runoff and Magela and Gulungul creeks (standard error).

	Ca	K	Mg	Na	SO_4	Al	Ba	Cu	Mn	Ni	Pb	Si	U	Zn
	mg/L							µg/L						
2009–10	1.6	1.3	0.61	2.9	5.8	32	18	0.18	1.6	0.06	0.15	2.3	10	0.66
SE (n = 6)	(0.4)	(0.6)	(0.1)	(1.3)	(1.3)	(8.7)	(7.0)	(0.1)	(0.8)	(0.0)	(0.1)	(1.3)	(4.4)	(0.3)
2010–11	1.3	0.47	0.59	0.61	2.9	7.9	20	0.22	0.66	0.06	0.03	0.45	4.7	0.69
SE (n = 15)	(0.5)	(0.1)	(0.2)	(0.2)	(1.8)	(0.6)	(5.0)	(0.0)	(0.2)	(0.0)	(0.0)	(0.0)	(0.7)	(0.2)
2011–12	1.2	0.56	0.56	0.57	2.0	16	43	0.56	1.6	0.11	0.09	0.63	6.4	0.78
SE (n = 15)	(0.2)	(0.1)	(0.1)	(0.1)	(0.6)	(2.2)	(9.5)	(0.1)	(1.0)	(0.0)	(0.0)	(0.1)	(1.0)	(0.2)
All Water Years	1.3	0.69	0.58	1.1	3.2	15	30	0.35	1.2	0.08	0.08	0.70	6.4	0.72
SE (n = 39)	(0.2)	(0.1)	(0.1)	(0.3)	(0.8)	(2.0)	(5.0)	(0.1)	(0.4)	(0.0)	(0.0)	(0.1)	(0.9)	(0.1)
Magela Creek	**0.4**	**0.2**	**0.5**	**0.6**	**0.3**	**39**	**1.42**	**0.3**	**5.3**	**0.1**	**0.02**	**n/a**	**0.02**	**0.6**
Gulungul Creek	**0.4**	**n/a**	**0.9**	**n/a**	**0.2**	**52**	**n/a**	**0.2**	**3.1**	**n/a**	**0.03**	**n/a**	**0.08**	**1.1**

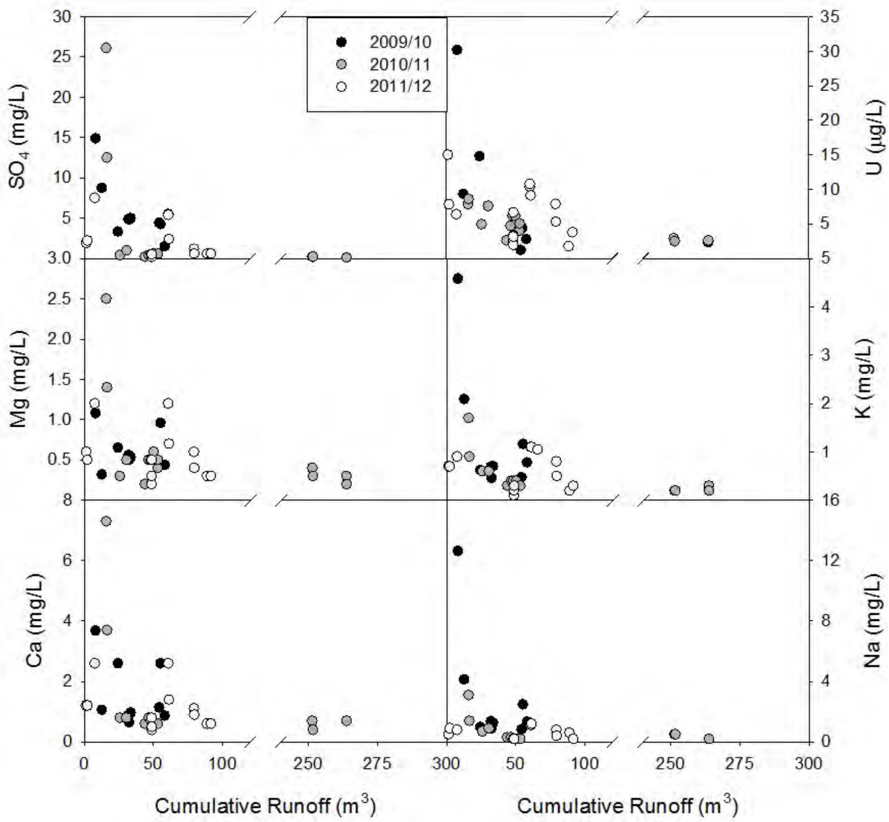

Figure 5 Major ion and uranium response to cumulative runoff on Plot 1. Each wet season is represented by a different shaded point.

Major ions

The ternary diagram (Figure 6) shows the relative proportions of Ca, Mg and 'K plus Na' in each of the samples collected from Plot 1. The centre of the triangular crosshairs indicates equal contribution (33.33%) of each cation to the total amount of cations in each sample. Figure 6 shows that during the 2009–10 wet season, the majority of the samples were K/Na dominant, with these cations contributing 40–50% of the total cations present. The contribution of K/Na decreased over the following wet seasons, with the majority of samples collected in the 2010–11 and 2011–12 wet season being Ca dominant with significant amounts of Mg. This supports the results of the Mann-Whitney tests that show a decline in K and Na concentrations over time. Unfortunately, the corresponding anion ternary diagram cannot be constructed as alkalinity (HCO_3^- and CO_3^{2-}) was not measured.

Figure 7 shows that Mg and Ca both have significant linear relationships with EC. These relationships were used, with the continuous EC record, to predict the continuous Mg and Ca concentrations. The discharge measured for each event was then used to calculate the Mg and Ca loads using equation 3.1. Table 7 summarises the Mg and Ca load data, including total annual loads, the load/unit runoff (as described earlier for the TDS loads) and the maximum event loads, along with the date and the rainfall associated with the event.

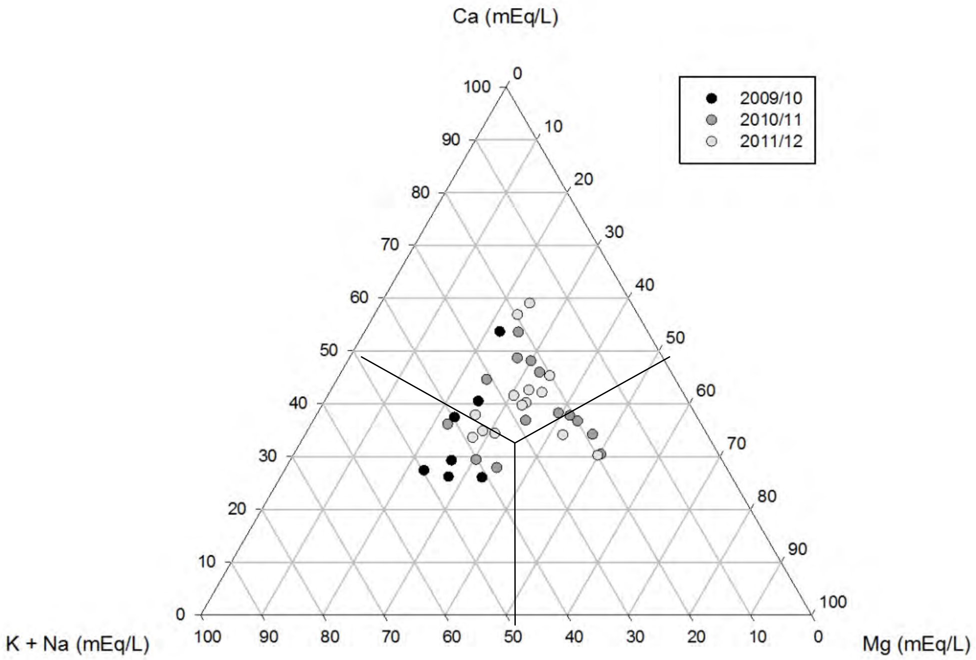

Figure 6 Ternary plot showing cation (mEq/L) composition of surface water runoff over the 2009–10 (black), 2010–11 (dark grey) and 2011–12 (light grey) wet seasons

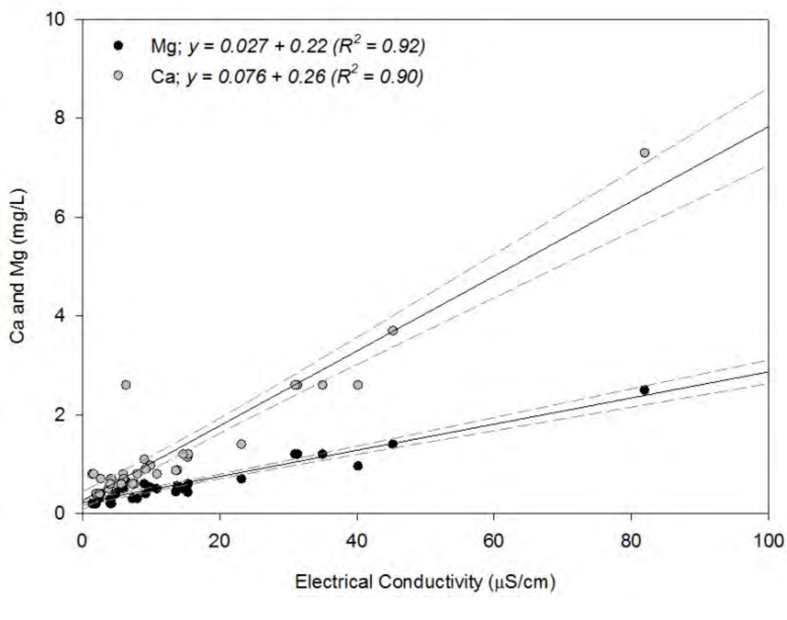

Figure 7 Linear relationships between electrical conductivity and magnesium (black points) and electrical conductivity and calcium (grey points).

Table 7 Total seasonal load (g), load/unit runoff (g/m^3) and maximum even loads (g) for Ca and Mg

	Total seasonal load		**Load/unit runoff**		**Maximum event load (date, mm rainfall)**	
	Mg	**Ca**	**Mg**	**Ca**	**Mg**	**Ca**
2009–10	37	92	0.52	1.28	6 (23/12/09, 45 mm) 6 (1/2/10, 77 mm)	16 (23/12/09, 45 mm)
2010–11	93	178	0.34	0.65	59 (21/2/11, 189 mm)	101 (21/2/11, 189 mm)
2011–12	31	66	0.33	0.70	17 (3/12/11, 85 mm)	36 (3/12/11, 85 mm)

The data in Table 7 show that the greatest loads of Mg and Ca were exported from the trial landform in the 2010–11 water year. This is not surprising given that the runoff during this year was more than double that of the other water years. However, using the load/unit runoff the rate of export was greatest during the first water year and then decreased and stabilised over the following years.

Conclusion and future work

The priority for further work 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 but is progressing well.

Work will also be carried out during 2013 to assess the movement of mine-related contaminants in suspended sediment. As stated earlier, a number of suspended sediment samples have been collected and require chemical analysis, which is scheduled for 2013. This work will be aligned with a number of other SSD projects, including the assessment of transport of mine-related contaminants in sediments in Magela and Gulungul creeks, spectral imaging of sediments and radiological components in sediments on the trial landform.

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. Future monitoring 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) that are being used to predict long-term erosion performance and to 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 plan to manage the export of erosion products.

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Definition of sediment sources from landslides and their effect on contemporary catchment erosion rates in the ARR

MJ Saynor, W Erskine, G Staben & J Lowry

Introduction

The aim of this project is to characterise landslides in the Magela catchment that occurred during the extreme event of February and March 2007, to determine their contribution to stream sediment loads and to investigate the historic frequency and distribution of landslides in the Magela Catchment.

Progress to date

This project is nearing completion and had much work undertaken on it during 2012–2013 which culminated in two papers being published (Erskine & Saynor 2012, Saynor et al. 2012). A collaborative project with Andy Fourie and a PhD student, (University of Western Australia) is continuing following a field visit to the site in August 2012. This latter project addresses the geotechnical properties of the regolith in which the landslides occurred. A publication from this PhD work is being prepared.

Results

An extreme storm occurred over the Magela Creek catchment for the 72 h period between 17:00 h on 27 February and 17:00 h on 2 March 2007 when 740, 784 and 692 mm were recorded at the mid-Gulungul gauging station, Jabiru Airport and Ranger Tailings Dam respectively. Probable Maximum Precipitation here for 72 h duration is 2200 mm. Bureau of Meteorology rainfall intensity-frequency-duration analyses for the mid-Gulungul station showed that for durations greater than 6 h, rainfall intensities exceeded 1:100 year. However, return periods for durations between 48 and 72 h exceeded 1:1000 years. This intense rain generated the the highest flood since gauging commenced in 1971 on both Magela and Gulungul creeks. Further details are contained in Erskine & Saynor (2012).

The February–March 2007 storm triggered a series of landslides where there were surface outcrops of Oenpelli Dolerite (Figure 1). From a combination of field and aerial inspections and interpretation of Landsat and ALOS AVNIR-2 imagery, seventeen landslides were identified in the Magela Creek basin and a further 32 were identified in the East Alligator River basin. There may be more landslides in the East Alligator River basin because of the limited satellite coverage. The landslides were classified as mudslides, multiple mudslides, debris slides and multiple debris slides using standard engineering classification schemes. Field observations indicated that the failure plane was usually associated with the contact of the soil with the weathered dolerite and was always highly irregular. Landslides only occurred on slopes greater than 17^{o} and where soil depth was greater than 1 m.

From field measurements of the dimensions of selected representative landslides and laboratory determinations of landslide sediment bulk density, the estimated eroded sediment mass by landsliding was 134,950 t. This is much larger than the 78,100 t estimated by Saynor et al. (2009) because of a larger and more reliable data set used for the present calculation. The updated eroded sediment mass is only equivalent to 11 years of total sediment yield for Magela Creek at the G8210009 gauging station downstream of Ranger mine.

"Red" flood pulses were recorded on Magela Creek on 24, 25 and 30 January and 5 Fenruary 2008 by Supervising Scientist staff (Figure 2). When first observed on 24 January 2008, the "red" water was followed upstream by helicopter directly to the landslides in the Magela catchment. Tributaries that drained catchments without landslides produced "clear" water. These "red" flood pulses produced a different turbidity-suspended sediment concentration relationship to that before and after the "red" pulses (Moliere & Evans 2010). Erskine & Saynor (2012) used the Munsell Soil Colour® (Anon 2010) of the silt and clay suspended sediment during the "red" flood pulses to determine the source of the "red" suspended sediment.

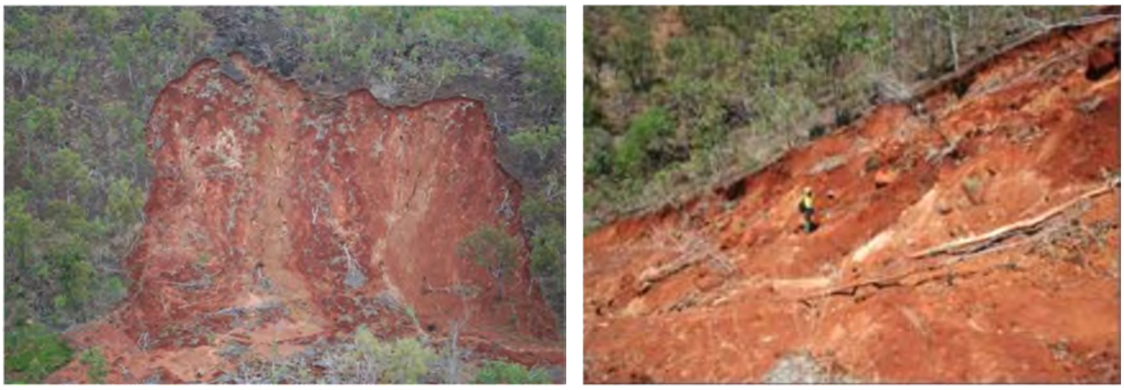

Figure 1 Oblique aerial photographs of two landslides triggered by the February-March 2007 storm in the Magela catchment

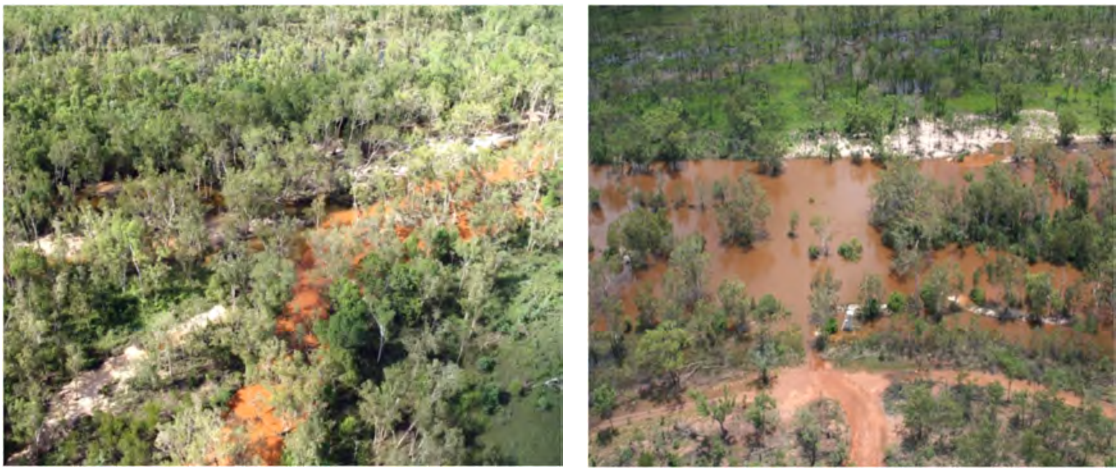

Figure 2 "Red" flood pulse on Magela Creek upstream of Ranger mine on 24 January 2008 (Left photo) and "red" flood pulse on Magela Creek at 8210009 gauging station downstream of Ranger mine on 5 February 2008 (Right photo)

Erskine & Saynor (2012) found that the silt and clay fraction of the suspended sediment during the "red" flood pulses had hues produced by the mixing of landslide sediments with that from the rest of the catchment. The Munsell colour names of the measured soil colours of the silt and clay fraction of the suspended sediment of the "red" flood pulses were 'yellowish red', 'reddish yellow', 'very pale brown', 'yellow', 'brownish yellow' and 'strong brown'. Clearly none were "red" although 51% of landslide sediment samples were 'dark red'. In 2008 when "red" flood pulses did not occur the colour of the silt and clay suspended sediment was essentially the same as previous and in subsequent wet seasons. A series of local storms in 2008 generated "red" flood pulses from just 0.3 km^2 of landslides in the 600 km^2 catchment at the G8210009 gauge. The landslide-derived sediment was progressively mixed and diluted with sediment-derived from the remainder of the catchment because the upstream station (MCUGT) recorded more and redder silt and clay suspended sediment samples than the downstream station (G8210009). Traditional owners were concerned about the mine's contribution to the coloured water and a combination of sediment fingerprinting using soil colour and up- and downstream of mine measurement of suspended sediment was capable of demonstrating that a landslide source was responsible for the short term "red" flood pulses on Magela Creek in 2008. Erskine (2013) has also found that soil colour is a reliable sediment fingerprint for another area where landslides were an important but short term sediment source.

Field and aerial inspections plus stereoscopic interpretation of vertical air photographs and analysis of ALOS images have so far failed to reveal any evidence of older landslides, except for occasional rockfalls along sandstone excarpments. None of the landslides identified here existed in February 2007 before the extreme event. The diameter at breast height over bark (DBH) was determined for 23 trees which were located on landslides and which were moved by the 2007 landslides. The maximum diameter was 0.55 m and the mean, 0.21 $\pm$ 0.02 m (SE). While we cannot convert these diameters to tree ages, none are particularly large which would indicate a long period of landform stability.

Given that the February–March 2007 storm was only about one-third of the Probable Maximum Precipitation for a range of durations, larger events are likely to cause more extensive landsliding, especially on Oenpelli Dolerite. However, we do not know the true frequency of these extreme events and plan to continue research on the subject.

Steps for completion

A telemetered raingauge has been installed on top of one of the dolerite hills near where the landslides occurred. An event relationship between SSD's rainfall data and Bureau of Meteorology radar data now needs to be established. Such a relationship would enable archived radar data to be used to quantify historical storms, including the large rainfall event of February–March 2007. This rain gauge is also used for early warning of floods that might impact on Magela Creek downstream.

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Initial assessment of the conceptual rehabilitated Ranger landform

J Lowry, G Hancock & T Coulthard

Introduction

The Supervising Scientist Division (SSD), in collaboration with research partners at the University of Hull (Professor T. Coulthard) and the University of Newcastle (Associate Professor G. Hancock), have carried out an initial assessment of the geomorphic stability of a conceptual rehabilitated landform of the Ranger mine using the CAESAR-Lisflood landscape evolution model (LEM). CAESAR-Lisflood is an enhanced version of the CAESAR LEM (Coulthard 2000, 2005) that had previously been used to assess the geomorphic stability of the Ranger trial landform (Lowry et al. 2011, Saynor et al. 2012) . Crucially, this is the first time that CAESAR-Lisflood has been applied to an entire conceptual rehabilitated mine landform. Energy Resources of Australia Ltd (ERA) provided two conceptual landform scenarios to SSD for assessment: the first represented a landform with an additional quantity, or surcharge, of material on the surface of the landform to allow for consolidation of the capped tailings in the pit; and the second a non-surcharged landform representing the final consolidated land surface (Figure 1). For the purposes of this study, the digital elevation models (DEMs) of both conceptual landforms (surcharged and non-surcharged) were divided into a series of sub-catchments which were individually modelled (Figure 2).

The assessment was divided into two distinct parts. Part 1 focussed on the potential impacts of the consolidation of the Pit 1 landform, and its impact on the catchment of Corridor Creek. Part 2 focussed on the impact of the conceptual landform on the catchments of Djalkmara, Coonjimba and Gulungul creeks. It is important to note that the modelling results reported here assumed that the whole model domain is covered by waste rock material. This condition will be refined in subsequent modelling evaluations following the provision of a more precise distribution of final surface types.

It should be noted that this project is a component of a wider suite of projects being run by SSD that will ultimately assess the geomorphic stability of the conceptual Ranger landform over a simulated period of 10,000 years under a range of climate scenarios, including extreme events. The results of the 10,000 year modelling will be reported separately.

Methodology

The application of the CAESAR-Lisflood LEM to the trial landform required the collation and integration of data from a range of different sources. The key data inputs used by the model were a digital elevation model (DEM) for each scenario; rainfall data and surface particle size data.

The DEMs were generated through the integration of two-metre interval contour data (produced from a LiDAR survey of the mine in 2011) with two-metre contours representing the various conceptual landform scenarios to produce a grid surface with a horizontal resolution of two metres. The final DEMs used for modelling purposes were compiled to a horizontal spatial resolution of 10 metres. Ten metres was determined to be the optimal resolution at which the CAESAR-LisFlood LEM could function within the spatial extent of the study catchments, and over the temporal periods modelled.

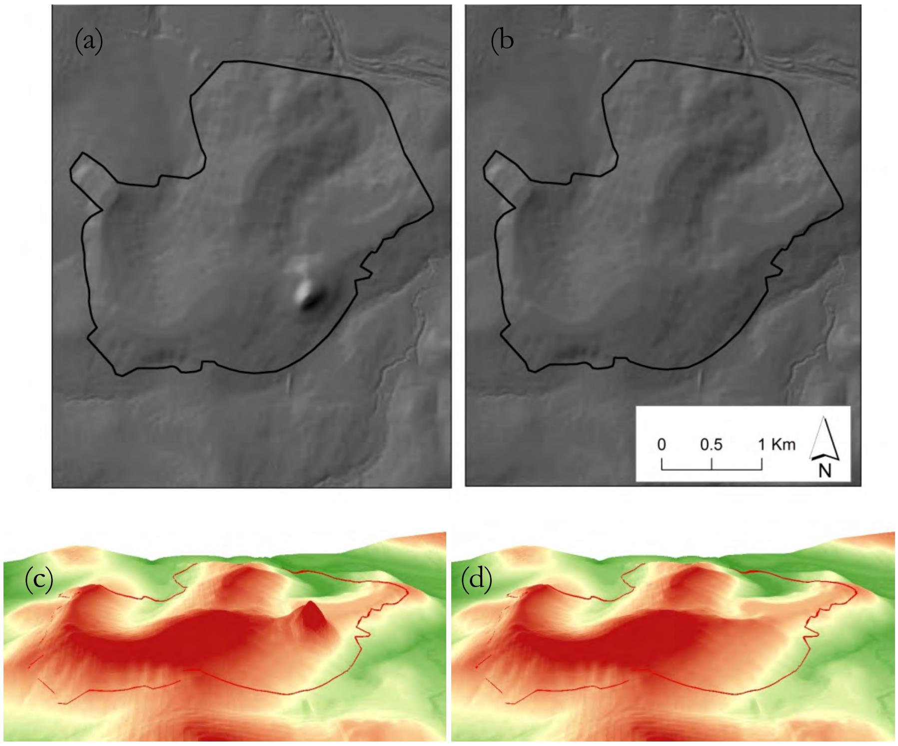

Figure 1 Surcharged (a) and non-surcharged DEMs (b) of the landform. Oblique perspectives of the surcharged and non-surcharged landforms are shown in (c) and (d). The black line represents the boundary of the rehabilitated landform.

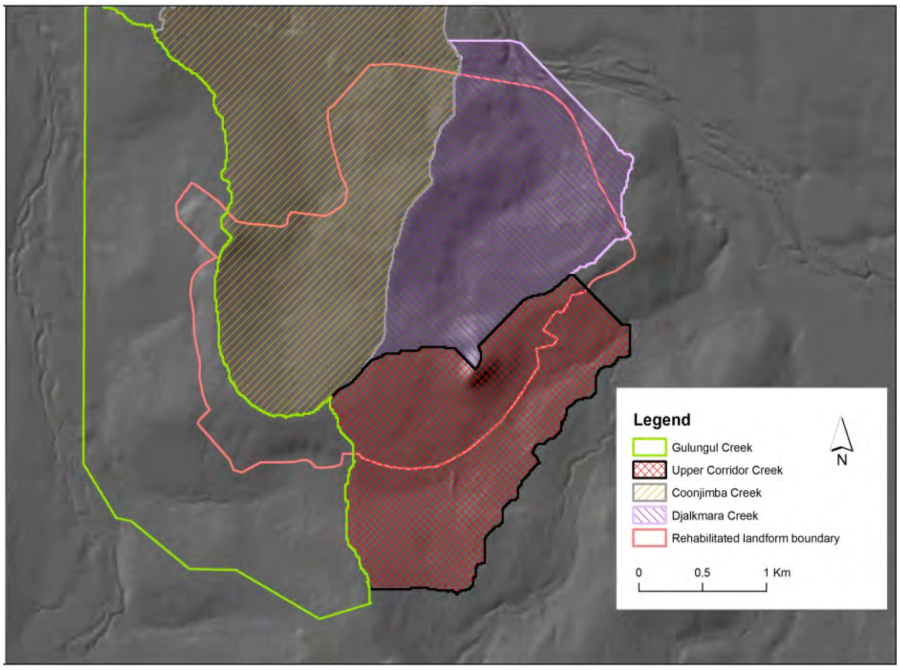

Figure 2 Catchment areas used for assessing the Ranger conceptual landform.

Rainfall data collected at Jabiru airport over the period 1971–2006 were processed and used to produce a dataset containing 22 years of continuous rainfall data. This dataset was used to produce 2 rainfall scenarios:

1. A 45-year simulation, incorporating two iterations of the 22- year data with the addition of data from an extreme rainfall event from March 2007. In the course of the latter event, 785 mm of rainfall was recorded in the three day period between 27 February and 2 March; rainfall intensity for most durations in this period exceeded a 1-in-100 year storm event.
2. A 1,000 year simulation was run in which the 22-year Jabiru rainfall was looped out to a period of 1,000 years. The 2007 extreme rainfall event was not used in this simulation due to computational limitations.

Grain size data for CAESAR-Lisflood were obtained from size fractionated bulk samples of surface material collected at eight points on the waste rock surface of the Ranger trial landform. Grain size analysis was completed on these samples and the results averaged into nine grain size classes (Saynor & Houghton 2011).

The effect of differential consolidation on the surface of the surcharged landform was done by calculating the depth of tailings fill placed in Pit 1. The depth of tailings was calculated by subtracting the present day LiDAR DEM of Pit 1 (filled with tailings) from a DEM of the unfilled Pit 1 and creating a file of tailings depths. ERA engaged consultants ATC-Williams in 2009 who simulated consolidation over five core depths (30, 60, 90, 120 and 150 m) and provided a point data file for each of these depths (ATC-Williams 2009). Exponential decay functions were fitted to the consolidation time series data. These functions were then built into the model so that consolidation occurred at the correct rates and extents across the backfilled pit area of the model domain. CAESAR-Lisflood determined the consolidation rates for fill depths between these values by interpolating between the rates for different depths.

Simulations were done on a catchment-by-catchment basis, for vegetated ('best case') and unvegetated ('worst case') scenarios of simulated periods of 45 years and 1000 years. Vegetated scenarios simulated the development of a mature grass community on the landform. In the case of the Corridor and Djalkmara Creek catchments, additional simulations were done to investigate the impact of the consolidation of the landform. The 45 year time frame was used to model the evolution of the landform for the duration that consolidation was likely to occur on the landform, whilst the 1000 year time frame was used to provide a longer-term analysis of the landform evolution.

Results

The catchments studied in Part 2 produced very different results from the catchment studied in Part 1. Specifically, large scale erosion producing a channel/incised floodplain hundreds of metres wide by tens of metres deep is predicted in the catchments of Djalkmara and Coonjimba catchments over a period up to 1000 years, under both 'best case' (where vegetation is present) and 'worst case' (vegetation absent) scenarios. This is a particular concern in Djalkmara catchment (Figure 3), where the predicted path of the channel would pass through the current area of Pit 3, which may contain contaminated material as part of a future rehabilitated landform design.

An example of the sediment load predicted by the model from the conceptual landform, using Djalkmara catchment as an example is shown in Table 1. For all catchments, the predictions of sediment load and denudation rate at both the 45 year and 1000 year periods are much higher than published rates for the region, regardless of the presence of vegetation. These results may be partially attributed to the nature of the material being modelled (that is, the assumption that the entire modelled landform was composed of waste rock) and the increased vulnerability of the freshly constructed surface at the beginning of the simulation period. During this initial phase, finer particles within the freshly made landform surface will be preferentially eroded and irregularities in the topography are smoothed, resulting in initially higher sediment yields and denudation rates.

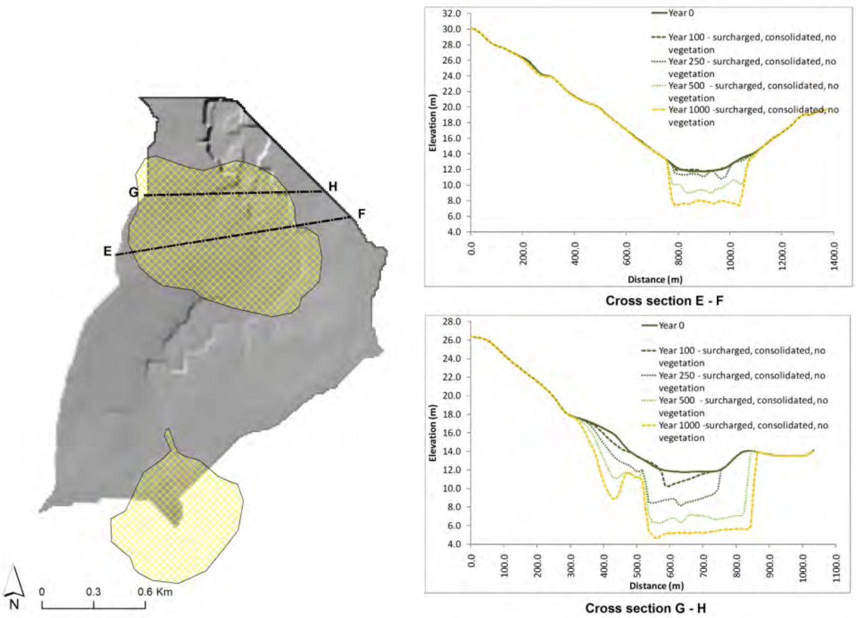

Figure 3 Unvegetated ('worst case') 1000 year surface of Djalkmara Creek catchment. Hatched areas represent current extent of Pit 1 (lower) and Pit 3 (upper).

Table 1 Sediment yield and denudation rates after 1000 years from CAESAR-Lisflood – Djalkmara catchment

	Surcharge				No Surcharge	
	Vegetated		No Vegetation			
	Consolidation	No Consolidation (static)	Consolidation	No Consolidation (static)	Vegetated	No Vegetation
Total Load (m^3)	582365	572237	3381335	3338057	546430	3522639
Denudation rate (mm y^{-1})	0.20	0.20	1.15	1.14	0.19	1.20

The relative proportions of the total sediment yield in Djalkmara catchment were representative of the sediment load in all the catchments modelled, with the majority composed of bedload, with a small proportion (10–20%) of suspended material (Figure 4). Importantly, the impact of an extreme weather event can also be seen in Figure 4, where the rainfall data for the extreme rain event recorded in March 2007 is introduced at the end of the 45 year cycle and can be seen to produce increased sediment loads off the (unvegetated) landform. Clearly, it is essential to have as much data as possible on the magnitude and frequency of such extreme events for future model simulations.

The results of this project were recently presented at the 8th International Mine Closure conference in the United Kingdom (Lowry et al. 2013). Significantly, this presentation was recognised in the conference summary minutes as one of the 'technological highlights' of the conference, indicating an acceptance amongst the mine closure community of the ability of CAESAR-Lisflood to successfully model the evolution of a landform.

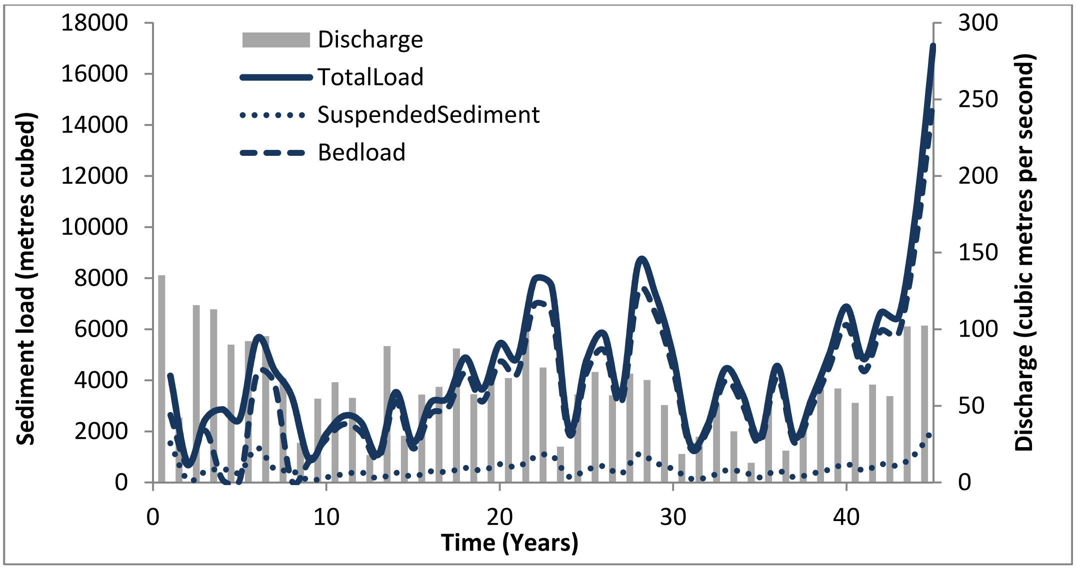

Figure 4 Variation in bedload and suspended load in the unvegetated Djalkmara catchment over the 45 year period. The largest historical storm occurred in year 45.

Steps for completion

The predictions of the geomorphic stability of the Ranger conceptual landform are prefaced with two important caveats. First, all simulations assumed that the entire surface of the area modelled was composed of the same type of material (waste rock). This is an unrealistic scenario for a final landform, where (1) a proportion of the footprint will comprise natural land surface; (2) erosion control structures including armouring of steeper drainage lines and sediment traps will be in place; (3) lined and engineered catchment drainage lines will be present; and (4) there will be an initial post rehabilitation programme of works to correct or remediate any excessive local areas of erosion that occur. Second, it is also recognised that the 'worst case' scenario of an unvegetated landform for a simulated period of 1000 years is unrealistic for the Ranger site. However, it has been retained to provide an example of an extreme worst case.

The study identified several engineering issues which should be considered when designing the final landform, and which may serve to reduce or minimise the erosion predicted. These include reducing the slope within the catchments (thereby reducing the flow velocity and erosive capacity of surface runoff); increasing the elevation and/or armouring areas containing buried materials which need to be protected (ensuring water flows around, rather than through them); and reducing the size of the catchments (reducing the ultimate energy available for erosion and transport).

Future designs submitted for modelling assessment should therefore incorporate erosion control devices, different surface treatment options to reduce sediment yield, and downstream sediment traps to contain eroded material.

The introduction of vegetation into model scenarios was found to reduce total load and denudation rate. Further work needs to be undertaken to quantify the effect of the development of different vegetation communities on erosion rates and to assess approaches to including the potential effects of fire and climate change (e.g. rainfall event frequency, duration and intensity) in the model runs.

Further simulations need to extend the period of model simulations out to 10,000 years, to align with the statutory period required for monitoring and assessing long-term landform stability of the final landform. The results of the simulations to date provide a guide for future enhancements both to the landform design and to the landform software model and provide increased confidence that the CAESAR-Lisflood model will be able to correctly predict the evolution of a rehabilitated landform once it has been constructed.

Finally, Terms of Reference for a review of the landform evolution modelling activities at SSD (as requested by ARRTC at its 30th meeting in May 2013) have been drafted. The review will focus on the landform modelling approach and technologies utilised by SSD for modelling and assessing the long term geomorphic stability of a final rehabilitated landform for Ranger mine. The review is planned to be completed by June 2014.

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Model the geomorphic stability of proposed rehabilitated Ranger landform for up to 10,000 years

J Lowry, T Coulthard & G Hancock

Introduction

The project aims to assess the geomorphic and erosional stability of the rehabilitated Ranger landform for a period of 10,000 years. It is an extension of a research project undertaken in 2012–2013 which provided an inital assessment of a proposed rehabilitated Ranger landform (Refer to report KKN2.2.4 – Geomorphic stability of the Ranger landform). As part of the current project, a variety of agreed, extreme-rainfall, climate scenarios will be identified and modelled, using the CAESAR-Lisflood and SIBERIA landscape evolution models. The CAESAR-Lisflood and SIBERIA models will be run both individually and in combination, to give the greatest possible confidence to simulation results. This research will be undertaken in collaboration with A/Professor Greg Hancock (University of Newcastle), who has extensive experience in the use of the SIBERIA model, and Professor Tom Coulthard (University of Hull), who is the developer of the CAESAR-Lisflood model. Outputs from this project are required by Q3 2015, according to ERA's latest closure timeline.

Background

In December 2011 at the 27th meeting of the Alligator Rivers Region Technical Committee (ARRTC), ERA announced a concerted focus on rehabilitation and closure research needs for the Ranger uranium mine. This work was to be carried out as part of a closure pre-feasibility project. At this time it was apparent that an increased focus on rehabilitation and closure research needs should be factored into the 2012–13 research planning cycles for both ERA and SSD. To expedite this process, ERA identified their key closure-related tasks, together with associated knowledge requirements, and prioritised these against the current Key Knowledge Needs (KKNs). Consultations between ERA and SSD then took place to map these needs against current KKNs, and to assign organisational responsibilities for completing the required work.

Among the identified priority needs was an assessment of the geomorphic stability of the proposed landform (KKN 2.2.1 *Landform Design* and KKN 2.2.4 *Geomorphic Behaviour and Evolution of the Final Landform*), with results to be available by the end of the third quarter of 2012. This information is required for use by ERA in the finalisation of the landform design, and to assist with the development of closure criteria.

Historically, the landform evolution modelling assessments have been performed by SSD, with SSD having invested substantially in the development and application of suitable models. In particular, SSD has invested significant resources in assessing, developing and adapting landform evolution modelling software, such as CAESAR-Lisflood, to assess the geomorphic stability of rehabilitated mine landforms.

Methodology

The project will utilise both the CAESAR-Lisflood and SIBERIA landform evolution models. In this way, the key strengths of both models will ensure the most reliable and accurate assessment of the evolution of the landform over a simulated period of 10,000 years. A key strength of CAESAR is its ability to simulate individual storm events (driven by the hourly rainfall record) whereas SIBERA determines erosion based on average annual rates. On the other hand, while CAESAR has a greater focus on shorter time-scale fluvial processes (using multiple grain sizes and calculating flow patterns using multiple flow directions), SIBERIA focuses on slope and catchment-wide processes over longer time scales. It is intended that the models will be run both independently and in an integrated fashion. For the latter, simulation outputs for an initial simulated period are produced by CAESAR-Lisflood and these are then used as input variables for the SIBERIA model to run the balance of the simulated period out to10,000 years.

Progress to date

Noting that model outputs are currently not required by ERA until Q3 2015, this project is currently progressing at a low intensity. Activity to date has focussed on:

- Identifying and sourcing particle size distribution data for the likely different surface cover types on the final landform
- Identifying sources of rainfall data which may be used to generate the range of climate scenarios that may occur over a period of 10,000 years, and
- Determining the optimal model (software) and infrastructure (hardware) settings for long-term simulations.

A paper, *"Assessing the long-term geomorphic stability of a rehabilitated landform using the CAESAR-Lisflood landscape evolution model"* was presented at the 8th International Mine Closure Conference at the Eden Project, United Kingdom in September 2013 (Lowry et al. 2013). Significantly, this presentation was recognised in the conference summary minutes as one of the 'technological highlights' of the conference, indicating an acceptance amongst the mine closure community of the ability of CAESAR-Lisflood to successfully model the evolution of a landform.

Steps for completion

In order to meet the deadline of Q3 2015 identified by ERA for an assessment of the geomorphic stability of the landform over 10,000 years, the following datasets will need to be provided by ERA within the time frames noted below:

- Information on the likely type / composition of the different surface cover types that will occur on the rehabilitated landform, e.g. area of waste rock; armoured surfaces; laterite; and undisturbed / natural surfaces outside of the landform; etc. by Q2 2014.
- Information on the latest predicted rates of consolidation around Pit 1 on the landform by Q2 2014.
- Information on proposed rainfall and climate scenarios that could be applied within the time frame of the 10,000 year scenario, by Q2 2014.

- The digital elevation model of the final rehabilitated landform (incorporating design and control structures) and the distribution / extent of the different surface cover types, by Q3 2014.

Professor Tom Coulthard (University of Hull) and Associate Professor Greg Hancock (University of Newcastle) will be engaged during 2013–14 to provide ongoing support to landform modelling activities at SSD. Professor Coulthard will support the continued development of the CAESAR-Lisflood model – in particular the development of the vegetation component, and application of multiple surface cover types to the landform model. Associate Professor Hancock will support the integration of the SIBERIA and CAESAR-Lisflood models, and the generation of climate scenarios for modelling activities extending out to 10,000 years.

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Radon exhalation from a rehabilitated landform

A Bollhöfer, C Doering & R Akber

Introduction

The trial landform provides a unique setting to investigate radon (^{222}Rn) exhalation for various rehabilitation options at Ranger uranium mine, its seasonal and long-term changes and dependency on soil ^{226}Ra activity concentration, cover type, soil moisture, weathering and compaction effects. The objective of this project is to determine ^{222}Rn activity flux densities from four erosion plots (EP) on the trial landform with various combinations of cover types (waste rock and waste rock-laterite mix) and re-vegetation strategies (direct seeding and tubestock) and to investigate seasonal and long-term changes in ^{222}Rn exhalation from the substrate.

Almost 1000 measurements of ^{222}Rn activity flux density have been made across four erosion plots since early 2009, covering both wet and dry seasons. Seasonal variability of ^{222}Rn activity flux densities have previously been reported for soils in the Alligator Rivers Region (Todd et al. 1998; Lawrence et al. 2009) with wet season radon exhalation generally suppressed due to the higher soil moisture. The hypothesis is that radon exhalation rate and its seasonality may change as the final landform evolves after rehabilitation of the site, in particular due to the highly weatherable nature of Ranger waste rock (Riley & Waggitt 1992).

In 2012 gamma dose rates were measured across the whole of the trial landform (Bollhöfer & Doering 2012). The variability across the erosion plots is low, with averages of the terrestrial gamma dose rate between 0.20 $\mu Gy \cdot hr^{-1}$ (EP4) and 0.24 $\mu Gy \cdot hr^{-1}$ (EP2). The measured gamma dose rates were used to estimate average ^{226}Ra activity concentrations across the entire trial landform. Actual ^{226}Ra activity concentration measurements were performed in 2013 to confirm the estimated ^{226}Ra activity concentrations for the erosion plots only.

An important parameter required to predict radon exhalation from a substrate is the radon diffusion length (see for example Porstendörfer 1994). This parameter gives a measure of the depth from which radon that emanated from a soil grain can diffuse through the soil, reach the soil surface and exhale into the air. Typically in natural soils, the radon diffusion length is about 1.5 m. To determine this parameter experimentally for waste rock, two sets of six 240 mm diameter PVC tube columns filled with waste rock to a depth of 0.5 m to 3.0 m were set up at the Jabiru Field Station in 2013 (Figure 1). This new collaborative project aims at determining the radon diffusion length in waste rock and any changes in this parameter with ongoing weathering of the material. More information about the project can be found in the ERA research summary to ARRTC.

Methods

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 and deployment are provided in Bollhöfer et al. (2006). 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 the count time on the detector measurement periods, enable ^{222}Rn flux densities from the surface of the soils to be determined (Spehr & Johnston 1983).

Surface soil samples were collected from directly underneath the radon cups deployed in 2012 and used to determine the ^{222}Rn activity flux density normalised to the ^{226}Ra activity concentration at the sites ($R_{E\text{-}R}$ in mBq·m^{-2}·s^{-1} per Bq·g^{-1}). Approximately 15 g or 150 g are pressed into a standard geometry and then measured on the ***eriss*** high purity germanium detectors to give ^{238}U, ^{226}Ra, ^{228}Ra, ^{228}Th, ^{210}Pb and ^{40}K activities in the samples. Procedures for sample collection, preparation and measurement via gamma spectrometry are described in Marten (1994) and Pfitzner (2010). An in-house programme is used for analysis of sample radionuclide activity concentrations (Esparon & Pfitzner 2001.

Volumetric water content was measured using Frequency Domain Reflectometry (FDR) soil moisture sensors (*Campbell Scientific*) that were installed during trial landform construction at various depths. FDR sensors measure the propagation velocity of an electromagnetic pulse through the soil, which is converted to the volumetric water content using a calibration equation. It is important to note that this calibration equation is unique to the soil type and accurate sensor calibration has not been possible at the time of moisture sensor deployment. A standard calibration equation has been used for all moisture sensors.

Radon columns at the Jabiru Field Station (Figure 1) were filled by ERA and ***eriss*** staff in April 2013 with a 4:1 mix of of different size waste rock crushed to 70 mm (rocks), and gravel and finer material less than 40 mm, respectively. After filling the columns, a 0.2 m deep head space remained at the top of the columns. To measure the ^{222}Rn accumulated in this head space over time, a plastic lid is held tightly over an O-ring at the top of the column and ^{222}Rn concentration build up is measured using two radon monitors (*Durridge Rad 7*). The build up of ^{222}Rn with time is used to determine ^{222}Rn activity flux densities from the various height columns, which is then used to determine the radon diffusion length. For more detail see SafeRadiation (2013).

Figure 1 Set up of radon column experiment at Jabiru Field Station, April 2013 (photo from SafeRadiation (2013)).

Results

Table 1 shows the results of the average ^{226}Ra activity concentration measured across the four plots in 2009 (Bollhöfer & Pfitzner 2010) and 2012, and a comparison with ^{226}Ra activity concentrations determined from terrestrial gamma dose rates measured in 2012 at ~5m resolution, using equations provided in Saito & Jacob (1995). Direct ^{226}Ra activity concentration measurements for 2012 agree well with the average determined from the gamma dose rates for Plot 1, Plot 3 and Plot 4, but measured ^{226}Ra activity concentrations in Plot 2 are slightly higher. The measured ^{226}Ra activity concentrations for 2009 are all lower than 2012, with Plot2 still the highest.

Table 1 Average estimated (from terrestrial gamma dose rate measurements) and measured soil ^{226}Ra activity concentrations in the four erosion plots.

Plot	^{226}Ra [$Bq \cdot g^{-1}$] average ± error (95% conf)		
	calculated	measured 2012	measured 2009
EP 1	0.32 ± 0.05	0.35 ± 0.07	0.19 ± 0.02
EP 2	0.38 ± 0.05	0.56 ± 0.12	0.42 ± 0.03
EP 3	0.36 ± 0.04	0.43 ± 0.08	0.33 ± 0.03
EP 4	0.32 ± 0.04	0.35 ± 0.04	0.26 ± 0.02

To determine the ratio of the ^{222}Rn activity flux density to the ^{226}Ra activity concentration (R_{E-R}) in the substrate of the plots, the ^{226}Ra activity concentration inferred from gamma dose rate measurements has been used. The geometric mean R_{E-R} ($mBq \cdot m^{-2} \cdot s^{-1}$ per $Bq \cdot g^{-1}$), or typical ^{222}Rn activity flux density normalised to the ^{226}Ra activity concentration, has been calculated for erosion plots 1 and 2 (waste rock), and erosion plots 3 and 4 (waste rock - laterite mix) combined, and is shown in Figure 2. Saynor et al. (2011) report that the direct seeding method did not reach the required number of plants per ha, and the two erosion plots (EP2 and 3) were in-fill planted with tubestock one year after initial seeding. Hence the data have not been differentiated between the two revegetation strategies, although EP2 in particular lacks behind the other plots in terms of establishment of vegetation.

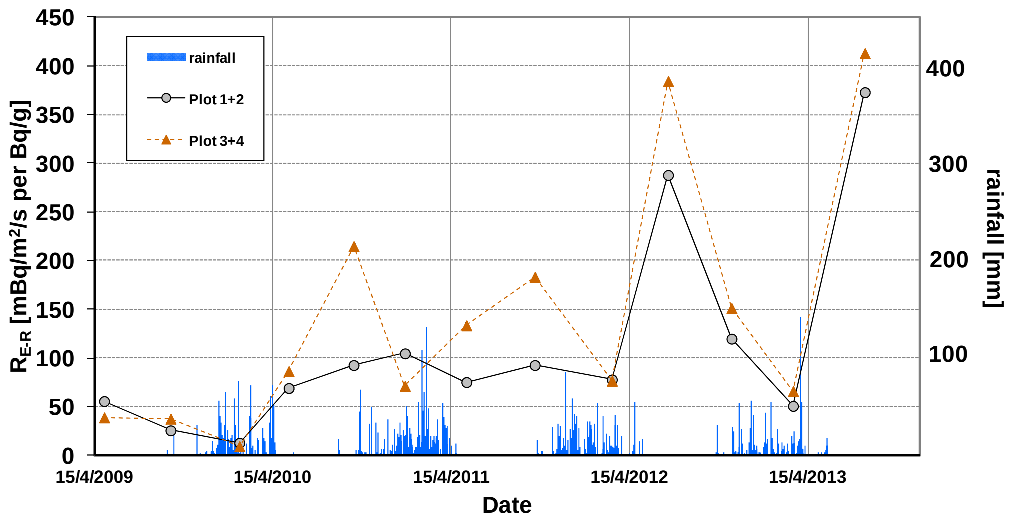

Figure 2 Geometric mean R_{E-R} (ratio of the ^{222}Rn activity flux density to the ^{226}Ra activity concentration) measured across the waste rock and waste rock - laterite treatments at the trial landform.

The waste rock - laterite mix shows the variability of radon exhalation from the beginning of trial plot construction, with higher R_{E-R} during the dry season and lower R_{E-R} during the wet. The waste rock only treatment shows little seasonal variability in the first couple of years, but similar variability and magnitude in years 4 and 5 after trial landform construction. Most likely, during the early stages of the trial landform water was infiltrating more easily into the coarse waste rock and reduction of the ^{222}Rn activity flux density due to a persisting higher soil moisture was small. In years 4 and 5, the waste rock has weathered, with smaller particles filling voids and pore spaces leading to higher ^{222}Rn exhalation during the dry, but typically low ^{222}Rn exhalation during the wet.

ERA has made available data from TDR soil moisture probes buried at the trial landform at various depths. Half-hourly volumetric water content (VWC) data have been logged over the past 4 years, and data are available from the surface 5 cm to a depth of 6 m. The half hourly data have been cleaned and are shown in Figure 3 for 0-0.3 m and 0.3-0.6 m depth. It is important to note that the soil moisture probes are not located directly in the erosion plots.

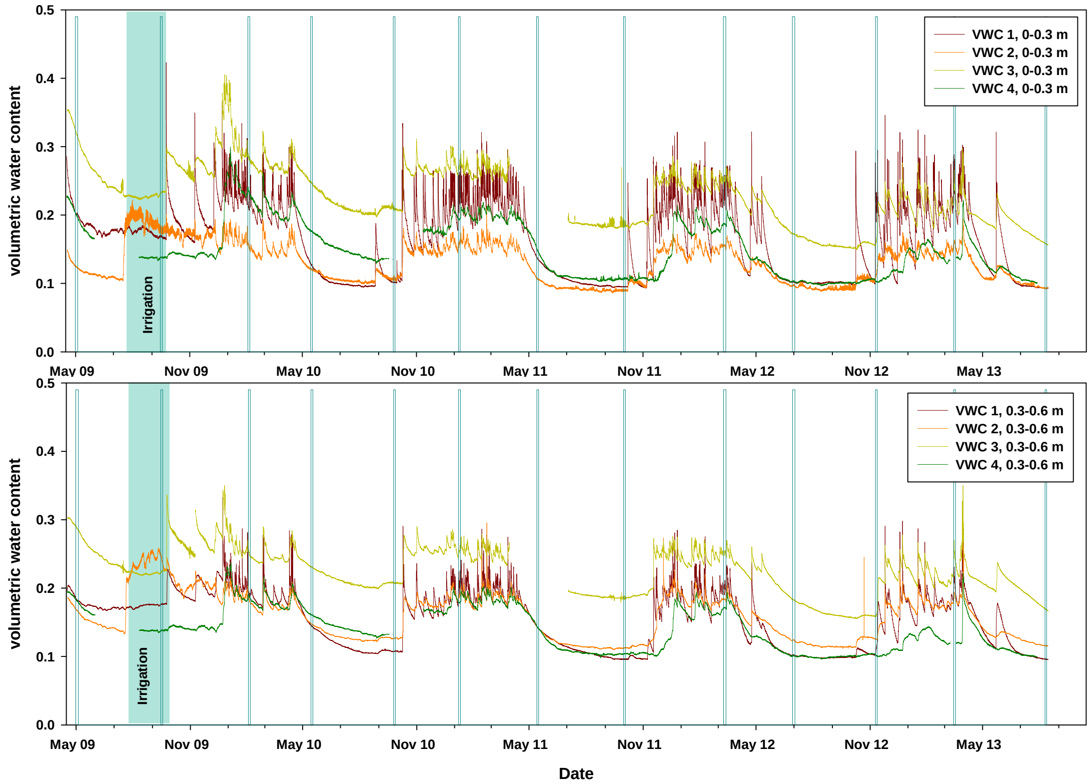

Figure 3 Volumetric water content measured at the trial landform for the various treatments (1-4) between May 2009 and September 2013. The bars indicate the periods of radon cup deployment.

Data for the top 0.6 m indicate that VWC is generally higher in treatment 3 (waste rock – laterite) than the other treatments. In contrast, VWC in treatment 4 (waste rock – laterite) is comparatively lower but similar to treatment 2 (waste rock). VWC in treatment 1 (waste rock) is highly variable during the wet season. The reason for the variability between and within different treatments is most likely due to the different soil characteristics, soil heterogeneity and associated uncertainties due to the calibration of the probes.

For further data analysis, the average VWC for the top 0.6 m during periods when radon cups were deployed (indicated in Figure 3 by the blue bars) was calculated for treatments

2 and 4, omitting the first two sets of measurements (during which some irrigation has occurred). The $R_{E\text{-}R}$ calculated for the waste rock and waste rock - laterite treatments shown in Figure 2 have then been plotted against the inverse of the average VWC in Figure 4.

Figure 4 shows that for a VWC around 0.25 ($VWC^{-1} \approx 4$) the ^{222}Rn flux density equals zero.These results are similar to results presented by Hosoda et al. (2007) who studied the effect of soil moisture on radon exhalation. Soils are typically saturated with water when the volumetric water content is between 0.2 to 0.5 (O'Geen 2012) and Hosada et al. (2007) report a drop in the ^{222}Rn diffusion coefficient by two orders of magnitude for soil moisture contents between 0.15-0.25. These are soil moistures that can be encountered during peak wet season conditions at the trial landform (Figure 3).

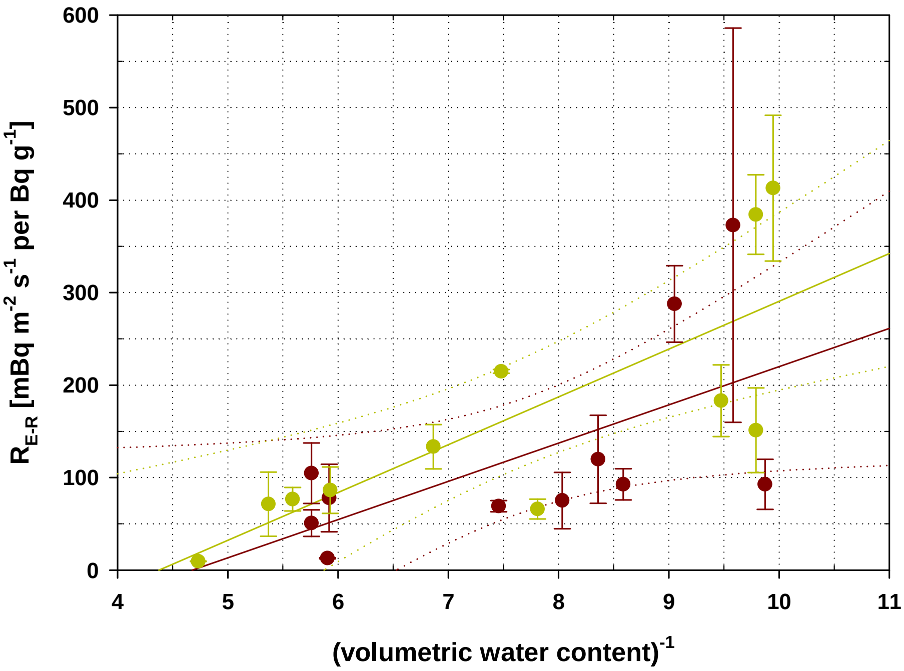

Figure 4 $R_{E\text{-}R}$ plotted against the inverse VWC for waste rock (brown) and waste rock – laterite (green) treatments on the trial landform. Dotted lines show the 95% confidence interval of the linear models fitted to the data. Waste rock: $R^2 = 0.37$; $p < 0.05$; Waste rock – laterite: $R^2=0.61$; $p < 0.005$.

The increase in ^{222}Rn activity flux density with decreasing VWC appears more pronounced for the waste rock - laterite treatment than for waste rock only, however within uncertainties there is no difference between treatments. For peak dry season conditions with a VWC of 0.09, $R_{E\text{-}R}$ is approximately 250 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ per $Bq{\cdot}g^{-1}$ for waste rock and 350 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ per $Bq{\cdot}g^{-1}$ for the waste rock - laterite treatment. These values are similar to the geometric means determined from direct ^{222}Rn flux density and soil ^{226}Ra measurements in 2012 of 210 and 370 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ per $Bq{\cdot}g^{-1}$, respectively. The highest $R_{E\text{-}R}$ of about 400 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ per $Bq{\cdot}g^{-1}$ was measured in the dry season of 2013. This value is similar to the geometric mean $R_{E\text{-}R}$ reported previously for waste rock areas at Ranger that had weathered for 7-10 years, between 410 and 660 $mBq{\cdot}m^{-2}{\cdot}s^{-1}$ per $Bq{\cdot}g^{-1}$ (Lawrence et al. 2009).

The dry season $R_{E\text{-}R}$ will allow to predict maximum ^{222}Rn activity fluxes from the entire rehabilitated landform for a given waste rock grade and footprint. This is provided that the waste rock cover is similar in depth to the waste rock on plots 1 and 2 (approximately 4 m). However, cover thickness may vary and knowledge of the ^{222}Rn diffusion length is required to determine ^{222}Rn activity fluxes for variable waste rock cover thicknesses. A first set of measurements performed in May 2013 using the experimental setup at Jabiru Field Station (Figure 1) gave a diffusion length for ^{222}Rn through Ranger waste rock of 2.32 ± 0.22 m (SafeRadiation 2013).

Conclusions and further work

^{222}Rn exhalation measurements will continue throughout 2013-14 to determine whether the high dry season ^{222}Rn activity flux densities measured in August 2013 prevail. The R_{E-R} will then be determined for dry and wet season conditions, to determine typical ^{222}Rn activity flux densities from the trial landform and predict ^{222}Rn activity fluxes from Ranger mine post rehabilitation. These fluxes will be compared to pre-mining fluxes in Bollhöfer et al. (2014).

Knowledge of the R_{E-R} and ^{222}Rn diffusion length will allow to predict dry and wet season ^{222}Rn activity fluxes from the rehabilitated landform for given grades and thicknesses of waste rock material used. The diffusion length will also allow to determine the effectiveness of the waste rock cover to reduce ^{222}Rn exhalation from buried tailings. A project currently underway is measuring the ^{222}Rn diffusion length (see ERA research summary to ARRTC) and measurements will be continued by ***eriss*** to investigate the effects of weathering on the radon diffusion length in Ranger waste rock.

Once information is available on type and thickness of waste rock material used for rehabilitation at Ranger, an above baseline GIS of soil ^{226}Ra activity concentration and ^{222}Rn activity flux densities will be produced in ArcGIS. The above baseline ^{222}Rn activity flux can then be used as an input into atmospheric transport models, to determine above background doses from the inhalation of radon decay products in the far field. An estimate of radon equilibrium factors on top of the rehabilitated landform is still required to assess doses from the inhalation of radon decay products when accessing the landform for hunting and gathering activities, in particular during the early morning hours.

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2.5 Ecosystem establishment

Under KKN 2.5.1, status reports are provided on two studies designed to progress the development of surface water quality criteria relevant to both operations and closure: Firstly, development of turbidity closure criteria for Ranger billabongs and stream channels; and secondly, development of solute closure criteria for Ranger billabongs.

Proposed closure values and advice arising from these studies, together with development of assessment criteria for monitoring and sign-off, will be considered by an Aquatic Ecosystems Technical Working Group, that will report to the Ranger Closure Criteria Working Group.

Development of turbidity closure criteria for receiving surface waters following Ranger minesite rehabilitation

A George & C Humphrey

Background

Turbidity is a water quality measure commonly used in aquatic ecosystems to represent the total suspended particulate and colloidal matter in the water column. Excessive and sustained turbidity in aquatic ecosystems can disrupt ecological function in aquatic ecosystems, including changes to other water quality variables (temperature, dissolved oxygen), changes to light conditions and hence to primary (plant) production, and disruption of feeding and respiration activities of aquatic organisms. Historically and following minesite construction, enhanced turbidity in aquatic ecosystems around Ranger mine associated with mine operations has been minimal. However, as Ranger moves into the decommissioning and rehabilitation phases, the risk to aquatic ecosystems from suspended sediment in runoff increases significantly, through the erodibility of newly-formed and initially unvegetated landforms.

The Environmental Requirements (ERs) for closure at Ranger stipulate that the minesite and associated waterbodies must be rehabilitated to a state which allows them to be incorporated into the surrounding Kakadu National Park. Water quality closure criteria are being developed in response to these requirements. The closure criteria aim to provide a management approach that allows water quality to remain within a range that will not compromise the long-term environmental objectives of the area. Turbidity is one of a number of water quality measures being developed to represent surface water quality closure criteria for receiving waters, including billabongs and stream channels, adjacent to the Ranger mine.

Biological-effects information for turbidity for the Alligator Rivers Region is available for two different ecosystem types, billabongs and sandy stream channels. The information for each ecosystem type is reported separately below, with an assessment then made as to whether or not different criteria would apply to each ecosystem type. For example and a priori, the ecology and biota of billabongs and flowing stream channels of the region differ, and so it is conceivable that each may differ in its general integrated response and sensitivity to enhanced and sustained turbidity.

Closure criteria derived from billabong data

Increased turbidity primarily acts to limit light penetration which reduces primary production, including that associated with phytoplankton. Turbidity can thus cumulatively affect various trophic levels, incurring direct biological impacts both to single biotic groups as well as to ecological processes (George & Humphrey 2013). Chlorophyll-a is an indirect measure of the amount of phytoplankton in the water column and is thus often used as a surrogate for phytoplankton biomass and a component of primary production.

A study being undertaken for Georgetown Billabong (GTB), adjacent to the minesite (see Map 2), aims to combine existing historical turbidity and biological-effects data with similar recent field data to determine the appropriate boundaries for turbidity closure criteria. The closure criteria for turbidity would be relevant to the area around Ranger mine and would aim to meet the ERs for Ranger rehabilitation such that the integrated aquatic ecosystems, natural and (possibly) re-constructed, represent fully functional natural systems. This study evaluates the indirect relationship between turbidity, as suspended particulate matter, and primary production, as a representation of ecosystem function. While at the ecosystem level, cause-effect relationships are complex and difficult to discern, simplistic effects can be inferred when multiple lines of evidence are evaluated and where good conceptual understanding is available.

All data for the study have been collected from GTB. It is one of the first waterbodies to receive runoff from Ranger and thus has been used extensively to monitor water quality and biological changes over time. Most recently, chlorophyll a and turbidity have been continuously monitored over a 10 month period (i.e. since the 2012 dry season) to determine the relationship between changes in turbidity and primary biological response (chlorophyll-a). The dry season increases in turbidity in GTB are natural and are a consequence of (mainly) wind-induced re-suspension of the silt-clay sediments, exacerbated over the dry season as the billabong becomes increasingly shallow through evaporation.

Surface chlorophyll-a and turbidity data from the dry season of three sample years, 1981, 2009 and 2012, are presented in Figure 1. Data from 2009 suggested that phytoplankton are inhibited at values between 25 and 70 NTU (Buckle et al. 2010). Such a threshold is masked in the data of Figure 1 because individual fortnightly data have been averaged on a monthly basis. This was supported by data collected in 1981 which suggested inhibited phytoplankton production at values around 50 NTU (Figure 1). The 2012 dry season data reflect a similar threshold effect, with phytoplankton biomass peaking at an average turbidity value of 68 NTU. The rate of change in turbidity and chlorophyll-a was gradual over the 2012 dry season with no 'extreme' values occurring (i.e. turbidity values greater than 150 NTU). However, other monitoring programmes in GTB have recorded turbidity values spiking from relatively modest to highly extreme values within a one month time period. The most dramatic example was found in 2003 when ERA recorded a turbidity increase from 16 to 782 NTU within one month and then a further increase to 1517 NTU within the next month. The present study did not experience such exceedingly high turbidity values or such rapid changes, warranting an assessment of ERA's long-term turbidity data, particularly for values that are not aligned either with SSD's continuous data or field checked monthly data.

The dry season studies summarised above reported a peak in chlorophyll-a in the mid dry season. In the current study, apart from the mid dry season chlorophyll-a peak, even higher values occurred, albeit briefly, within one week of initial wet season flows. In early January 2012, chlorophyll-a values reached 165 mg/m^3, a value four times that measured during the dry season. This value occurred when turbidity values had dropped from 75 to 10 NTU due to flushing flows. The low turbidity measured during this event indicates that phytoplankton biomass may have a secondary, though short, response to rapid changes in turbidity, in this case, elicited by a return to higher water clarity with wet season re-wetting and inundation. Short-lived peaks in nutrients with first-flush waters in tropical systems have also been implicated in corresponding spikes in primary production

(Walker & Tyler 1982). The full nutrient dataset associated with the present study has not been analysed yet to corroborate this.

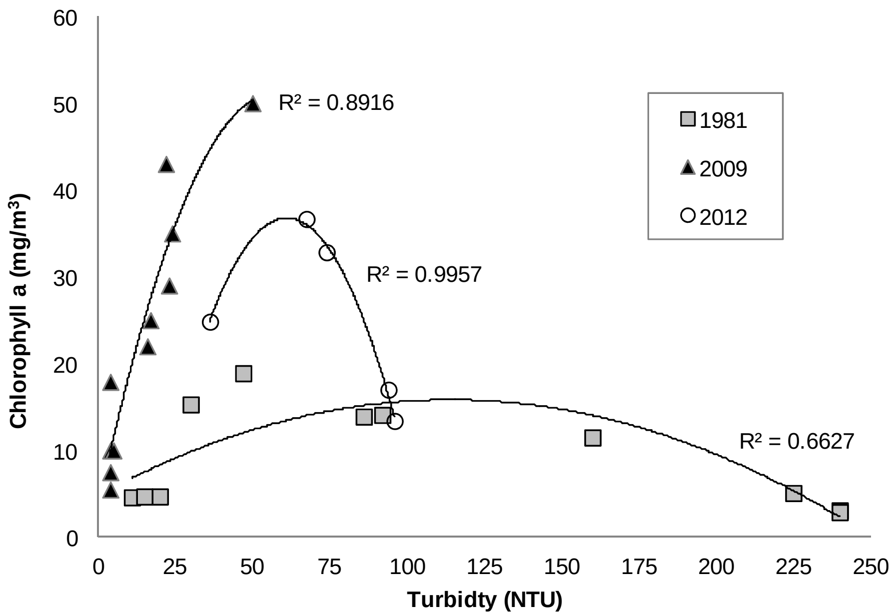

Figure 1 Chlorophyll and turbidity relationship in Georgetown Billabong during 1981, 2009 and 2012. Each year shows the R2 value for the second-order polynomial regression equation, indicating the strength of the relationship between chlorophyll a and turbidity. Data shown for 2012 are monthly averages (August to December) derived from weekly sampling, while data for 1981 are monthly measures (January to November). Data for 2009 regression are based upon fortnightly measures taken between May and October.

In the current GTB study, phytoplankton in three preserved water samples were identified and enumerated (Table 1). These samples represented periods of turbidity between 3 and 99 NTU and chlorophyll-a values between 7 and 26 mg/m^3. Cursory examination of the data suggest that the greatest diversity in phytoplankton may correspond with higher turbidity, while also suggesting the lowest diversity was associated with flushing flows and reductions in turbidity. Further data are required to adequately assess if a relationship exists between turbidity and taxa richness for this dry-wet season transition period. Half of the species that were found in the January sample were not present during the late dry period, thus illustrating a shift in species composition that may be a result of flow-related water quality changes. When the late dry season-early wet season chlorophyll data (from Table 1 and the value at initial billabong filling in January 2012, cited above) are compared with data from earlier in the (2012) dry season (Figure 1), it is evident that, in general, phytoplanktonic productivity is greatest during the mid dry season, although short-lived, but higher peaks are evident in response to flushing flows and billabong inundation at the very beginning of the wet season.

Table 1 Phytoplankton diversity and abundance in Georgetown Billabong, 2012–13

Sample date	1/11/2012	6/12/2012	17/01/2013
Chlorophyll a (mg/m3)	26	12	7
Turbidity (NTU)	99	91	3
Number of phytoplankton taxa	37	27	24

Closure criteria derived from sandy stream channel data

In the 1990s, an unformed stream-bed crossing on the upper reaches of Jim Jim Creek in Kakadu National Park (KNP) caused elevated turbidity in the flowing waters downstream of the crossing over of a period of several consecutive dry seasons. In response to KNP management concerns, research studies were conducted by SSD in 1996 to assess possible adverse effects of these increased suspended solids on macroinvertebrate and fish communities in the receiving water (Stowar 1997, Stowar et al. 1997). While impacts upon the biota were demonstrated, data analysis techniques at the time were limited in their ability to better define possible thresholds of effect and to identify and distinguish key taxa affected by the disturbance. Statistical techniques available today offer improved analytical performance in understanding the nature and extent of impact, including assessment of possible biological thresholds. The following study re-analyses the datasets acquired in 1996 to assess their potential for setting general water quality, including closure, criteria for turbidity in flowing-stream environments of the Alligator Rivers Region.

Jim Jim Creek in its upper reaches below Jim Jim Falls is a clear-water, sandy stream channel that flows for most of the year. A tourist road leading to the adjacent Twin Falls crosses Jim Jim Creek and is open to traffic for much of the northern dry season. Prior to 1997, early four-wheel drive access and traffic over the crossing washed away overlying sand deposited during the preceding wet season, to expose a clay creek-bed that gave rise, with ensuing regular traffic, to high suspended solids downstream for the remainder of the dry season.

Determining a threshold ecological response to elevated turbidity

Water physico-chemistry, including turbidity, and macroinvertebrate and fish communities were monitored in 1996 for two months prior to the opening of the creek crossing to tourist traffic in the early dry season (24 June) and for four months thereafter. A modified Before-After-Control-Impact Paired differences (BACIP) design was employed, which included at least paired sites in both Jim Jim Creek (upstream and downstream of the road crossing) and Twin Falls Creek (a control stream, with analogous but undisturbed upstream and downstream sites) – see Figure 2.

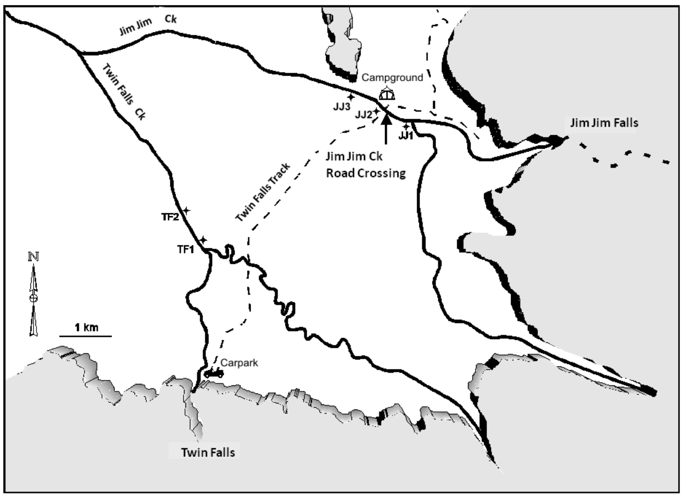

Figure 2 Map showing location of upper Jim Jim and Twin Falls Creeks and road crossing, with associated sampling sites studied in the 1996 dry season.

Replicate macroinvertebrate samples were collected from each of five sites: one upstream (JJ1) and two downstream (JJ2 and JJ3, 200 and 1000 m downstream respectively) of the road crossing on Jim Jim Creek, and two in a similar upstream-downstream configuration in Twin Falls Creek (TF1 and TF2 respectively) (Figure 2). Data reported here refer to samples collected from root mat habitat (ie dense mats of fine fibrous roots of riparian shrubs and trees, mainly *Pandanus aquaticus* and *Melaleuca* spp). Sampling was conducted at each site on three occasions prior to opening of the road crossing and on four occasions after. Fish sampling was undertaken in large pools at sites JJ1, JJ2, TF1 and TF2 sites on one occasion prior to, and one occasion four months after, opening of the road crossing.

The modified BACIP design used for macroinvertebrate community study applied the same principles as used in the equivalent biological monitoring study in Magela Creek catchment (see KKN 1.3.1 "Monitoring using macroinvertebrate community structure"in this report). Thus, for each sampling occasion and for each pair of (upstream-downstream) sites for Jim Jim and Twin Falls Creeks, dissimilarity indices were calculated. These indices are a measure of the extent to which macroinvertebrate communities of the two sites differ from one another. A value of 0 indicates macroinvertebrate communities identical in structure while a value of 1 indicates totally dissimilar communities, sharing no common taxa. Disturbed sites may be associated with significantly higher dissimilarity values compared with undisturbed sites. The site pairs examined in this way included JJ1-JJ2, JJ1-JJ3 and TF1-TF2.

In the earlier studies, a plot of the paired-site dissimilarity data from the two streams revealed a temporal trend in the values for any paired sites that were not under the influence of elevated downstream turbidity (i.e. pairs TF1-TF2, and pairs JJ1-JJ2 and JJ1-JJ3 prior to the road opening). The seasonal dissimilarity values are reproduced here in Figure 3C. (Stream discharge is used to depict and account for the trend, so that when interpreting the plots shown in Figure 3, time of year is effectively reversed on the horizontal (X) axis.) The same trends are also evident in other unimpacted ARR streams for which seasonal data have been gathered, including those of permanent flow – see Figure 3A & B. The trends indicate that as flow recedes after wet season flooding, the macroinvertebrate communities between any two adjacent sites of a stream become more similar to one another, presumably in response to lack of natural disturbances and the onset of predictable (but declining) flow.

Conventional ANalysis Of VAriance (ANOVA) testing for differences before and after impact at 'exposed' compared to control sites cannot be applied for data that display trend, and instead, incorporation of (natural) covariates of the trend can be used in ANalysis Of COVAriance (ANCOVA) testing. (The simplified null hypothesis is that the relationship observed after impact is consistent with the same relationship observed before impact.)

The magnitude of impact in the 'after' period may be sufficient that there is no such dissimilarity-discharge relationship at all. This appears to be the case for the JJ1-JJ2 and JJ1-JJ3 site pairs, where samples exhibiting turbidity-related disturbance appear as 'outliers' in the regression plot of dissimilarity versus stream discharge, where non-impacted and impacted samples are plotted together (Figure 3C). The early reports noted that the JJ1-JJ2 and JJ1-JJ3 dissimilarities fell increasingly outside of the 95% confidence limits of the regression relationship with decreasing creek flow (or increasing time after the crossing opening), inferring impact.

Multivariate ordination, depicting the relationship of macroinvertebrate samples to one another, and ANOSIM (ANalaysis of SIMilarity), a multivariate equivalent of ANOVA, were used to examine more closely, the putative impact samples identified in the earlier studies, i.e. most of the JJ2 and JJ3 occasions from the After (road crossing opening) period shown in Figure 3C. This analysis aimed to better define thresholds or gradients of possible impact associated with elevated turbidity.

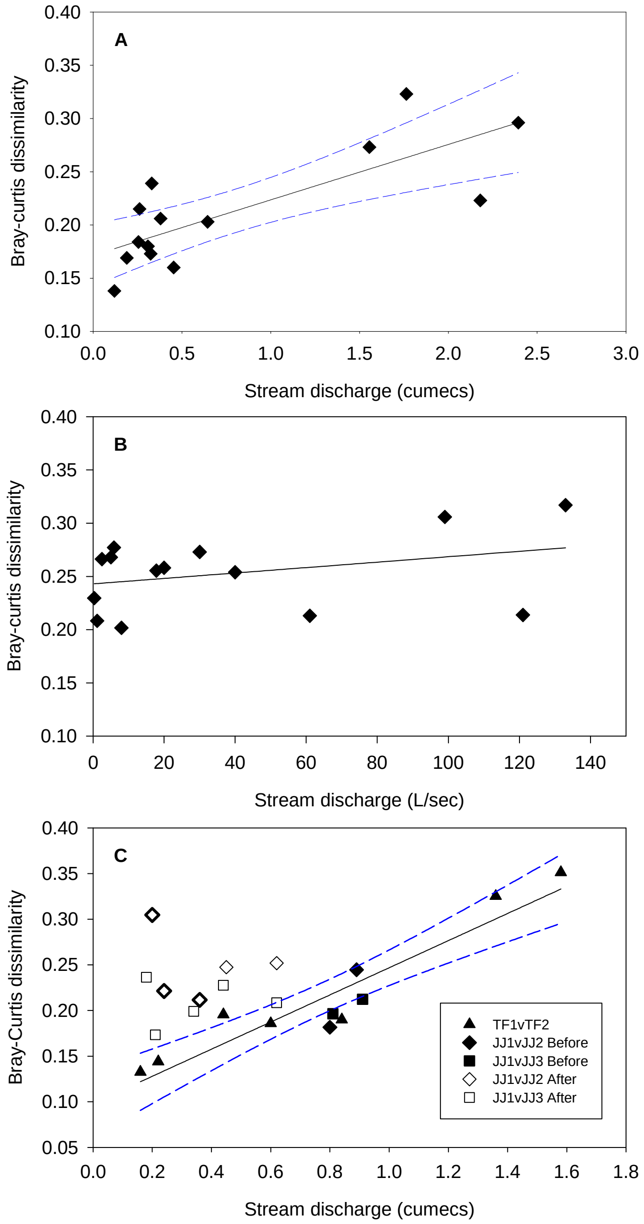

Figure 3 Relationship between discharge and dissimilarity of macroinvertebrate community structure between upstream and downstream sites for "A" the upper (permanent) reaches of the South Alligator River, "B" undisturbed portions of the seasonally-flowing Rockhole Mine Creek, and "C" Jim Jim (JJ) and Twin Falls (TF) Creeks using family level (log-transformed) data.

Before and After samples for Jim Jim Ck refer to before and after opening of the road crossing to traffic. The regression line and 95 percent confidence interval for Jim Jim/Twin Falls data (C) relate to all 'unimpacted' (black enclosed) samples and creek discharge at the time of sampling. R^2 values for regressions "A" and "C" are 0.61 and 0.8 respectively. Trend line in "B" is not significant.

The ordination representing all sampling sites and sampling occasions in the post-road-opening period is shown in Figure 4, with corresponding turbidity values depicted with each sample. Notable in the ordination is the interspersion of most Jim Jim site 3 samples (JJ3/5, JJ3/6 and JJ3/7 representing post-road opening) with corresponding control samples from upstream of the crossing (JJ1) and from Twin Falls Creek, indicating little if any impact. Conversely, Jim Jim site 2 samples from late in the tourist season (JJ2/5, JJ2/6 and JJ2/7), when turbidity reached highest values, are grouped together but are well separated from other sites and sampling occasions, indicating a distinctive community structure that reflects impact.

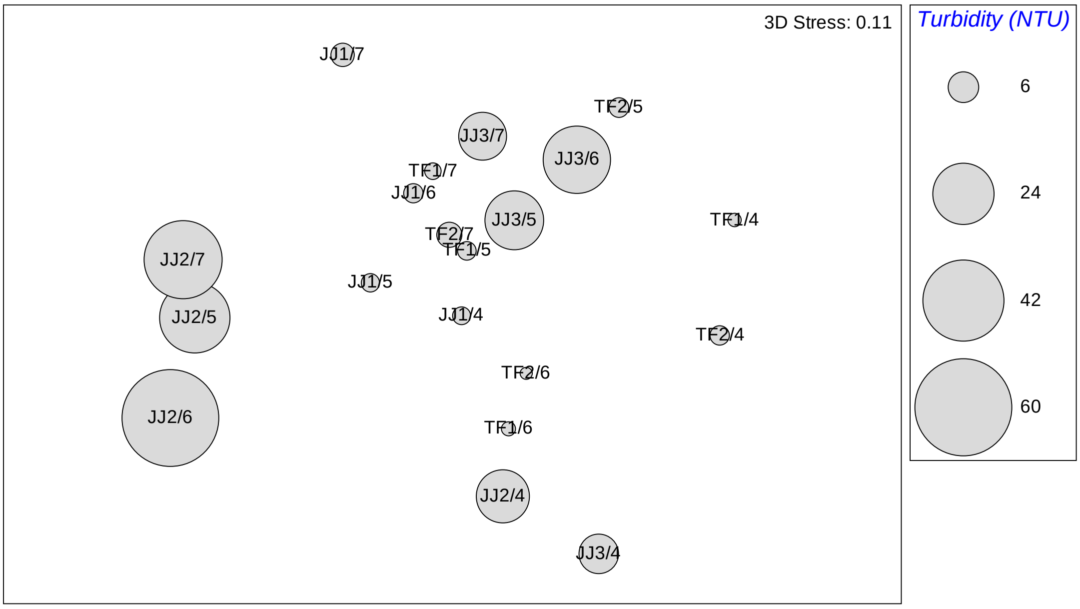

Figure 4 Axes 1 and 3 of Multi-Dimensional Scaling ordination of macroinvertebrate communities from Jim Jim (JJ) and Twin Falls (TF) Creeks using family level (log-transformed) data. Each data point represents the average of three replicate samples per site per sampling occasion. Numbers after creek abbreviations refer to site number / sampling occasion, e.g. JJ2/6 – Jim Jim Creek site 2, sixth sampling occasion etc. Bubble plot indicates turbidity values associated with each sample.

The ANOSIM test statistic was used to compare the observed community differences *between* groups – in this case the separate sampling occasions for 'exposed' JJ2 and JJ3 replicate samples, versus control samples for each corresponding time period, ie JJ1, TF1 and TF2 sites – with the differences among replicates *within* the groups. The degree of separation between groups is denoted by the R-statistic, where the ANOSIM program's authors provide guidance on the R-statistic as follows: > 0.75 = groups well separated, > 0.5 ≤ 0.75 = groups overlapping but clearly different, and < 0.25 = groups barely separable. The resulting ANOSIM R values are plotted with corresponding ambient turbidity measured at the downstream Jim Jim sites in Figure 5. With increasing time after opening of the road crossing, there is an increasing separation of the Jim Jim downstream sites from the corresponding controls sites. Only JJ2 macroinvertebrate communities for sampling occasions 5, 6 and 7 differed significantly (ANOSIM $P<0.05$) from respective control samples and the ANOSIM R values for these samples (>0.3) approach or exceed thresholds of separation consistent with the ANOSIM authors' guidance (from above).

On the basis of ANOSIM results (Figure 5), supported by the distinct separation of the sites in ordination space (Figure 4), JJ2 macroinvertebrate communities for sampling occasions 5, 6 and 7 can be regarded as impaired. These adverse effects appeared after turbidity had reached sustained values of 30 NTU (Figure 5).

JJ2 macroinvertebrate communities for sampling occasions 5, 6 and 7 are also depicted in the dissimilarity-discharge regression plot (Figure 3C) as emboldened open diamond symbols. Given that other JJ2 and JJ3 samples post-road opening also sit outside the 95% confidence limits for the regression plot, it indicates that a number of different analytical methods must be used to correctly identify ecologically impaired communities.

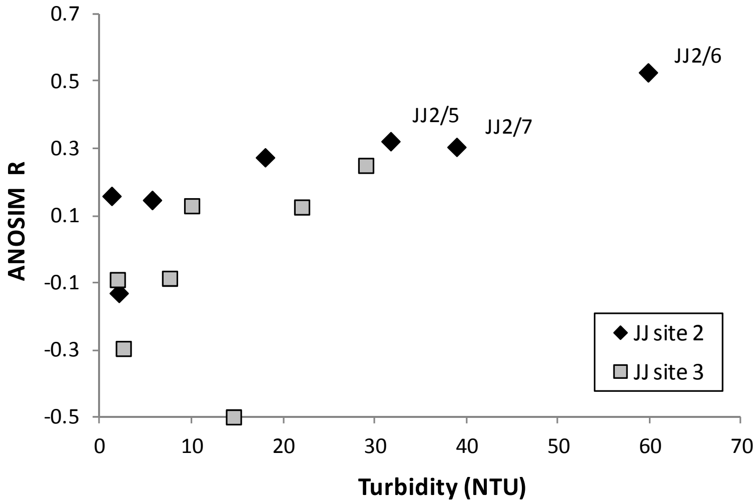

Figure 5 ANOSIM (Jim Jim downstream sites versus control sites) R values in relation to ambient turbidity at the 'exposed' JJ2 and JJ3 sites. Each data point represents one of the seven different sampling occasions. Site labels refer as follows: JJ2/5 – Jim Jim Creek site 2, fifth sampling occasion etc. Negative R values indicate high replicate variability within-samples (ie within JJ and/or control samples)

Types of ecological response noted with elevated turbidity

The SIMPER multivariate routine is used to identify the taxa discriminating between two observed sample clusters, such as observed in ordination space. SIMPER was used to identify the macroinvertebrate taxa influential in separating late season JJ2 samples (JJ2/5, JJ2/6 and JJ2/7) from control samples from the same time periods, ie JJ1, TF1 and TF2 sites. The top eight influential taxa, in decreasing order of influence, were adult dyticid beetles (+), adult corixid bugs (+), hydroptilid (cased) caddisfly larvae (-), protoneurid damselfly larvae (+), elmid beetle larvae (-), palaemonid shrimps (+), chironomid larvae (-) and baetid mayfly nymphs (-), where (+) and (-) refer to more and less abundant in JJ2 samples compared to control samples, respectively.

'Bubble' plots depicting the relative abundances of two influential taxa, corixid bugs and palaemonid shrimps, from sites represented in ordination space, are shown in Figure 6. As with the other influential taxa listed above that were more abundant in JJ2 samples compared to control samples, these two taxa are also predators. It is possible that the high proportion of predatory taxa found in these late-season JJ2 samples reflects the concept of 'ecological release', i.e. in the absence of the usual dominant predators, i.e. fishes (see below), a niche and opportunity is presented for invertebrate predators to fill. High levels of turbidity may also protect these taxa from visual fish predators. At least dyticid beetles and corixid bugs, moreover, use a bubble of air to respire and hence do not have external gills that would otherwise become clogged with suspended sediment particles. Thus these anatomical and physiological features would render these taxa relatively immune from turbidity effects.

The most significant observation made from the results of fish sampling (sites JJ1, JJ2, TF1 and TF2, before and after) was the dramatic (90%) decline of the numerically-dominant hardyhead, *Craterocephalus marianae*, from JJ2 late in the season. At all other sites, abundances of this species had increased from before to after the road opening (between 1.3 to seven-fold). This small-growing species is an active visual, predatory-feeder over benthic substrates of clear-water, sandy streams in the ARR. The significant decline in abundance of the species from JJ2 when turbidity was high is consistent with impairment of the fish's feeding behaviour. As noted above, this decline corresponded with an increase in abundances of predatory macroinvertebrate taxa. Because fish sampling after the road opening was only conducted once, and months after the opening, it was not possible to discern thresholds at which *C. marianae* declined at JJ2, only that it was sensitive to turbidity values as high as 60 NTU.

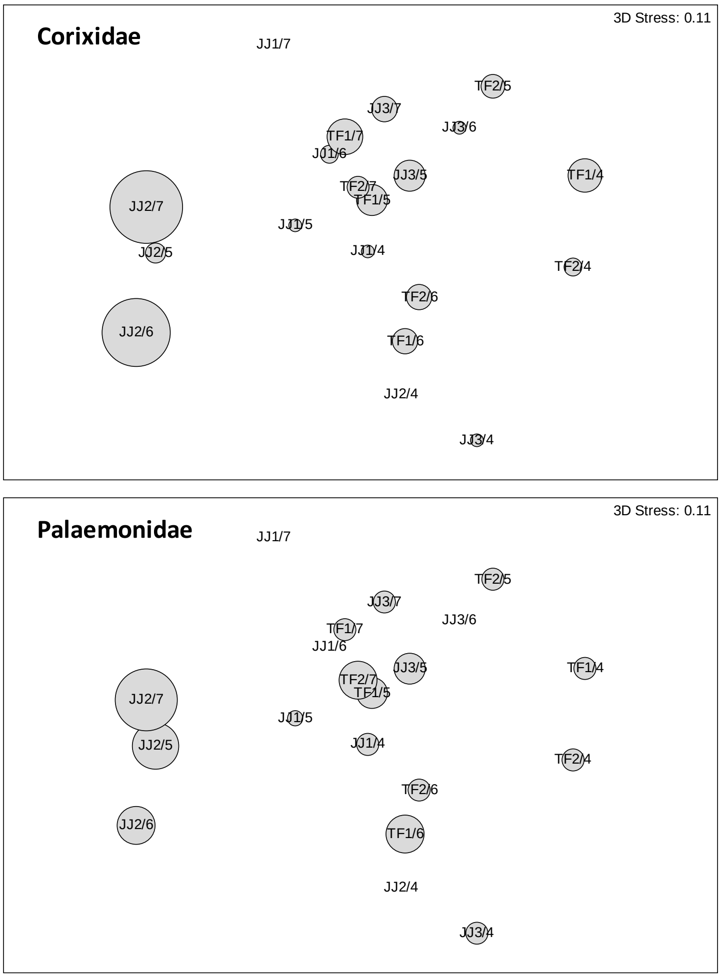

Figure 6 Axes 1 and 3 of Multi-Dimensional Scaling ordination of macroinvertebrate communities from Jim Jim (JJ) and Twin Falls (TF) Creeks using family level (log-transformed) data. Each data point represents the average of three replicate samples per site per sampling occasion. Numbers after creek abbreviations refer to site number / sampling occasion, e.g. JJ2/6 – Jim Jim Creek site 2, sixth sampling occasion etc. Bubble plots indicate relative abundances of indicated macroinvertebrate taxa associated with each sample.

Conclusions

Based on data collected to date, thresholds for biological effects of high and sustained turbidity in ARR ecosystems ranged between 50-70 NTU for phytoplankton in GTB, and 30 NTU for macroinvertebrate communities in the sandy Jim Jim Creek channel. Two caveats have been noted with respect to translating these values to minesite closure criteria:

1. The values are derived from dry season studies whereas erosion-induced, elevated turbidity associated with rehabilitation will occur during the wet season.
2. The sources of turbidity for the two studies reported above differ from those to be expected to arise from erosion of rehabilitated mine landforms, i.e. laterite and Cahill schists.

These factors require further assessment, together with decisions on: (i) relative risks to biota in the wet versus dry season; (ii) whether different criteria are derived for billabong and sandy stream channel ecosystem types or whether the more conservative (sandy creek channel) value from above is applied for all ecosystem types; and (iii) whether the ongoing billabong study can provide information on the effects of relatively short-lived turbidity peaks (cf sustained exposure that has characterised the studies to date). The appropriate forum for deliberation on these issues is a recently-formed, technical working group for the ERA Ranger Closure Criteria Working Group.

Future work will focus on refining the reported ranges in accordance with seasonal expectations for minesite erosion. A challenge for setting appropriate closure criteria is in understanding biological responses to rapid and unseasonal changes in turbidity. In addition to the ongoing field study, laboratory experiments are planned to help inform this issue.

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

C Humphrey

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. As reported in Humphrey & Jones (2012), GTB is being used as a case study to develop surface water quality closure criteria, in this case for the solutes, $MgSO_4$, EC (a reliable surrogate and correlate of $MgSO_4$) and U, for natural waterbodies within the mining lease.

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.

Progress to date

Results of macroinvertebrate sampling in the wet-dry season transition period of 1995, 1996 and 2006 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 (Jones et al. 2008) based on water quality and macroinvertebrate data acquired in 1995, 1996 and 2006. (Values are not shown here because additional data acquired for those years subsequent to publication in 2008 will result in changes to the values on finalisation of the closure criteria.)

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 to values above those observed in past sampling years (1995, 1996 and 2006). The values developed for those past three sampling years have been used as minimum quality values, so that any deviation above the derived values may trigger 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 (i.e. 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.

Antecedent water quality conditions in GTB for the wet-dry season transition periods since 2011 (based on EC) have deteriorated to the worst they have been since mining commenced. This is due to an increase in MgSO4 in the billabong, with no appreciable increase in U concentrations for the same period. Accordingly, macroinvertebrates in GTB and the previously-studied waterbodies were re-sampled in 2011 and again in 2013.

Sampling in 2011 revealed, for the first time, evidence of water quality impacts upon the macroinvertebrates of GTB, corresponding to a reduction in water quality observed during the preceding dry and wet seasons (Humphrey et al. 2013). Metrics quantifying the separation in multivariate space of GTB macroinvertebrate communities from those of reference waterbodies were regressed with the average antecedent wet and dry season Mg in GTB for all sampling years. From this relationship, a Mg concentration below ~3.5 mg/L was predicted to provide minimal GTB-reference waterbody separation in multivariate space, this value being consistent with the laboratory-derived TV for Mg (van Dam et al. 2010). Using the Mg threshold determined from 2011 and earlier sampling, only the ('no effects') water quality data from 1995, 1996 and 2006 were assessed as suitable for deriving potential water quality closure criteria for U, Mg and EC.

Further work

Macroinvertebrate samples collected in 2013, when antecedent water quality conditions were similar to those observed in 2011, are currenly being processed and too few data are available to assess possible water quality effects. Once 2013 results are available, a re-assessment of the Mg ecological threshold and relevant water quality record supporting development of closure criteria will be undertaken. The proposed closure criteria will be further assessed by the Ranger Closure Criteria Working Group. The implications of results for GTB will also be considered for deriving water quality closure criteria for natural (Coonjimba Billabong) or possible reinstated waterbodies elsewhere on the mining lease.

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

C Humphrey, P Lu[1] & G Fox

Background:

Revegetation research to inform future closure of the Ranger Project Area has used the concept of analogues. The work has been progressed by both 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. The sequence of research has shifted from broad regional Alligator Rivers Region (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 & Meek (2004) that early establishment focus just on this vegetation category. Work on trees and shrubs has drawn to a close with many of the research outcomes arising from the surveys and field work conducted at locations adjacent to Ranger, including intensive study 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, has provided much of the reference data about local vegetation communities. These vegetation data have been gathered separately by ERA and ***eriss*** but were 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 ARR (Humphrey & Fox 2010).

The primary objectives of the analogue work were to:

1. Characterise natural plant communities adjacent to Ranger and identify the key environmental determinants of those communities from a suite of chemistry, texture and structure variables for soils, as well as landscape terrain descriptors.
2. Use the findings from (1) to assist with:
    a. selecting the most appropriate species for revegetation of the Ranger mine landform post decommissioning, viz either:
        i. manipulating the environmental conditions in order to support key communities (i.e. through landform engineering or configuring) or
        ii. choosing the community types that best suit particular environmental conditions

[1] Energy Resources of Australia Ltd, Darwin

b. the development of revegetation closure criteria and a suitable post-closure, performance monitoring regime.

Progress to date

For ARRTC29, Humphrey (2013) summarized the vegetation analogue results accordingly:

1 Multivariate classification techniques identified four distinct vegetation communities on the Georgetown analogue area and other areas adjacent to Ranger that were undisturbed by mining activities: *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.

2 Modelling of plant-environment relationships was undertaken for communities and individual species from up to 54 analogue sites using soil chemistry, texture and structure, depth to groundwater and landform (terrain) variables. Both the community- and species-level modelling were consistent with one another in highlighting key – but obvious– differences between *Melaleuca* woodlands and the dominant mixed eucalypt woodland type. Amongst the three eucalypt woodland communities and constituent species generally, no strong 'preferences' were observed for particular environmental (soil/landform) conditions.

3 Outcome 2 above reflects the tolerance of position and general ubiquity of many of the ARR woodland tree and shrub species over a range of different environmental conditions. This suggested that recommendations for revegetation strategies at Ranger need not be overly prescriptive. Thus apart from placing *Melaleuca* woodland species in locations predicted to be seasonally-inundated / poorly draining, a possible strategy for eucalypt woodlands 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.

Applying the analogue findings

For the past 20 years, excess pond water from operational areas of the Ranger Project Area (RPA) has been irrigated onto areas of natural vegetation within the RPA. ERA will progressively rehabilitate these Land Application Areas (LAAs), commencing in early 2016 with remediation of two woodland plots located adjacent to Magela Creek, so-termed Magela and Djalkmara LAAs. Remediation of the LAAs must consider and develop closure criteria for radiation, soil quality and flora and fauna. Planning for revegetation and associated closure requirements for Ranger's LAAs, specifically, was provided by Addison (2011), while a general revegetation strategy for the RPA was most recently provided by Gellert (2012). The latter report provided a general framework for which revegetation of the LAAs would be carried out, comprising the following elements (not necessarily in priority or chronological order):

1 Adherence to a general revegetation strategy developed by ERA. The first articulation of the strategy was provided by Reddell & Meek (2004) who emphasized, amongst other recommendations, the need to first establish framework trees and shrubs (see 'Background' section above).

2 Use of reference or analogue sites for species selection and to provide a benchmark for development of closure criteria.

3 A clear articulation of stakeholder expectations.

4 Provenance and securing seed supply, i.e. agreement on the geographical range within which seed may be sourced and supplied.

5 Consideration of final land use (i.e. incorporation into Kakadu National Park and advice of Traditional Owners).

In addition, aspects for revegetation planning include site preparation, planting methods, site maintenance (including fire and weed control), monitoring and reporting.

The current project directly informs aspect 2 above, species selection and development of closure criteria. Magela and Djalkmara LAAs occur in areas dominated by the common mixed eucalypt woodland community identified in the current analogue study (see above, 'Progress to date') and hence this study can be used to advise on plant composition and densities.

The present analogue work is based on spatially-intensive, on-ground sampling, well suited to identification of particular plant community types. Two other complementary studies will also inform development of vegetation closure criteria in considering natural temporal changes and extrinsic (fire, weeds) factors not considered in the present study. These aspects are described in accompanying papers provided in this research summary (Bartolo et al. 2014, Whiteside et al. 2014).

The collective results from all studies, together with development of assessment criteria for monitoring and sign-off, will be considered by a Revegetation Technical Working Group, that will report to the Ranger Closure Criteria Working Group.

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Developing an integrated aquatic ecosystem research framework

A George & K Clark[1]

Introduction

In December 2012 at the 29th meeting of the Alligator Rivers Region Technical Committee (ARRTC), a new KKN was proposed to prioritise aquatic ecosystem rehabilitation research. The proposed KKN aimed to draw upon and integrate the aquatic ecosystem components embedded across various KKNs to ensure sufficient information is available for developing a rigorous rehabilitation framework for aquatic ecosystems.

An initial component of this project is a collaborative effort between SSD and Energy Resources Australia Ltd (ERA) aiming to identify key knowledge gaps for aquatic ecosystem establishment. Outcomes will directly support the revision of aquatic ecosystem KKNs and the identification of aquatic analogue sites. Outcomes will also support interdisciplinary strategic research across analogue and mine-affected sites.

Background

Biological and ecological research has been conducted on and around the Ranger minesite since the early 1980s. Both SSD and ERA have active research programmes which continue to address these areas of interest. A significant portion of this work is widely available through external peer reviewed journals and much of the information has been captured in internal reports and documents.

While these literature sources are readily available, the large volume of information, often fragmented and 'hidden' in larger reports, can make the dissemination of the key learnings from the studies difficult. Further, key learnings are often retained as 'corporate knowledge' which can be lost through staff attrition. Aquatic ecosystems benefit from a long-standing retention of corporate knowledge, but there is still a need to collate the existing information on this topic. A literature review ensures that any research programme developed for aquatic ecosystem establishment will address actual knowledge gaps, rather than attempting to direct work into areas which may have already been addressed, albeit not in the recent past.

This project will review existing aquatic ecosystem literature (primarily internal literature) to determine current and future research needs for aquatic ecosystems within the context of rehabilitation of Ranger mine. Concurrent to this, subprojects pertaining to the identification of aquatic analogue sites and the collation and re-analysis of existing aquatic vegetation datasets in particular, can also be initiated.

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Methodology

To address the literature review, available literature on aquatic ecosystems research within the ARR is being compiled into a simplified database intended to capture the extent of existing knowledge. The review will facilitate the identification of key knowledge gaps thus enabling a prioritisation of additional research which may be required.

Identification and selection of aquatic analogue sites will be delivered through collaboration with the teams addressing KKNs 2.6.1 and 2.6.2 (monitoring of rehabilitation success), and information provided by supporting research on aquatic ecosystems, both current and historical. The SSD Revegetation & Landscape Ecology Group (RLE) mapped and identified sites with current vegetation information. They will further work with the authors to populate attribute-based tables which will support criteria-based selection of aquatic analogue sites. This will provide both a rigorous, objective approach to selection of aquatic analogue sites across various spatial and temporal scales as well as a catalogue of existing knowledge about aquatic ecosystems. This work links to RLE projects on updating vegetation distribution across the region.

Targeted research at analogue sites is proposed for 2015, to assess seasonal and interannual variation of aquatic vegetation relevant to rehabilitation.

Progress to date

The literature review is in progress, though has been delayed by resource constraints. Research papers and reports from the late 1970s to 2012 on a range of topics across various trophic levels, from primary productivity to fish communities in shallow billabongs, have been reviewed. This preliminary evaluation suggests that the billabongs have undergone significant changes following the reduction of feral buffalo in the early-mid 1980s. While the formal literature review aspect of this project will conclude in 2014, regular updating of the literature database will be undertaken.

Field assessment of current biological monitoring sites was undertaken during the 2013 dry season to assess the suitability of the sites as aquatic analogues for taxa and attributes not currently studied. Additional information on these sites, available through various programmes, will be examined in 2014.

Initial compilation of aquatic vegetation data collected synchronously with billabong macroinvertebrate data has commenced, including preliminary analyses using multivariate techniques. Access to additional unpublished historical vegetation data in the region is being negotitated through Charles Sturt University.

Radionuclide transfer to terrestrial bush foods

C Doering, P Medley & A Bollhöfer

Introduction

Rehabilitation of the Ranger mine will result in the re-establishment of terrestrial ecosystem and eventual return of the land to traditional Aboriginal owners. The consumption of bush foods sourced from the rehabilitated Ranger site is one of the exposure pathways through which an Aboriginal person may receive an above background radiation dose. The magnitude of potential doses via this exposure pathway need to be assessed during rehabilitation planning to help inform the development of radiological closure criteria for soil quality. To make the dose assessment, information on radionuclide transfer to bush foods is needed.

The transfer of radionuclides to food items is typically quantified as the concentration ratio (CR) between the activity concentration in the food item and that in the surrounding environmental substrate (IAEA 2010). Soil is generally assumed as the substrate from where plants and animals in terrestrial environments obtain radionuclides. The use of CRs in dose assessment assumes equilibrium environmental conditions, which are the conditions expected to exist in natural (undisturbed) environments where any influx or outflux of radionuclides tends to be steady rather than strongly pulsed. Such conditions are expected to exist in the quasi-natural terrestrial environment of the rehabilitated Ranger site.

eriss has been measuring radionuclides in bush foods collected from around the Ranger mine and the broader ARR for more than 30 years. Sampling and measurement were initially focused on aquatic plant and animal species due to the importance of the aquatic transport pathway during the operational phase of mining (Johnston 1987, Martin et al. 1998). Recent emphasis has been on terrestrial plant and animal species due to the general paucity of data and the increased radiological importance of terrestrial bush foods during the post-rehabilitation phase when people can access the site. We have collated the existing radionuclide data with the aim of deriving reference CRs for bush foods. This paper presents preliminary results for terrestrial bush foods.

Method

Bush food and soil radionuclide activity concentration data were initially gleaned from ***eriss*** reports (e.g. Martin et al. 1995, Wasson 1992) and a large set of unpublished analysis results. The scope was then expanded to include data reported by the Ranger mine operator (ERA 1984, ERA 1985, ERA 1986, ERA 1987). Only measurement results reported for individual samples were collated from the source files and used in the data analysis. Summary statistics reported in source files were not collated or used.

A bespoke spreadsheet tool (Doering 2013) was used to calculate CRs on a fresh weight bush food to dry weight soil basis. The tool can be used to match bush food and soil samples collected within a user-specified distance and time period of each other and calculate CRs for the matched samples. Plants (fruits and yams) were matched to soils collected concurrent with the bush food. Animals were matched to soils collected within their typical home range and over a period of several years to give a representative estimate of the soil radionuclide activity concentrations to which the they were likely exposed during their lifetime.

Plant species were grouped as either fruits or yams to overcome impracticalities in calculating CRs for the wide variety of individual species sampled and difficulties in identifying exact species from some of the source files. This grouping was considered appropriate in the context of deriving CRs for rehabilitation planning to account for the various combinations of individual plant species that might be present on the rehabilitated Ranger site at any point in the future. No grouping was applied to animal species as they were substantially less in number and readily identifiable from the source files.

Results

Figure 1 shows CR geometric means and standard deviations for fruits and yams. Yam CRs were similar for all radionuclides and generally greater than the corresponding radionuclide CRs for fruit. Fruit CRs varied over two orders of magnitude and followed the approximate order ^{226}Ra ≈ ^{210}Pb > ^{210}Po > ^{232}Th > ^{238}U. It has previously been shown that some elements when absorbed from the soil through the root system are characterised by low mobility within the plant and are retained and accumulated in the roots (Pöschl & Nollet 2007). Translocation of uranium, thorium, radium, lead and polonium from the roots to the fruit is limited (Mitchell et al. 2013) which may explain the higher CRs for yam compared to fruit.

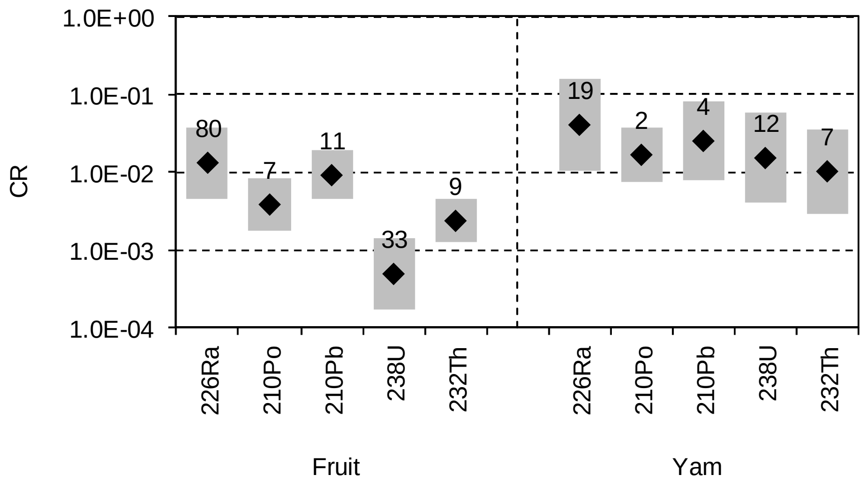

Figure 1 CR geometric means (black diamonds) and standard deviations (grey columns) for fruits and yams on a fresh weight bush food to dry weight soil basis (numbers indicate the number of bush food samples on which the geometric mean is based).

Figure 2 shows CR geometric means and standard deviations for flesh and organs of buffalo, pig and wallaby. The buffalo results (Figure 2(a)) indicate that ^{210}Po concentrates more in the organs than in the flesh, with the ^{210}Po CR for liver higher by one order of magnitude compared to flesh and that for kidney higher by two orders of magnitude compared to flesh. The ^{210}Pb CRs for organs were also higher than for flesh, typically by one order of magnitude. The liver and kidneys act as filters for substances penetrating or leaving the body of an animal and they have previously been shown to accumulate various pollutants, whereas radium preferentially accumulates in the bones (Pöschl &

Nollet 2007). The ^{226}Ra CRs were similar for flesh and liver, slightly higher for kidney and lower for heart. The ^{238}U and ^{232}Th CRs for organs were generally less than those of other radionuclides and the ^{238}U CR for flesh was similar to that for ^{226}Ra and ^{210}Po.

The pig results (Figure 2(b)) indicate that ^{210}Po strongly concentrates in flesh compared to other radionuclides, with the ^{210}Po CR higher by around two orders of magnitude. The other radionuclides all concentrate to a similar level in pig flesh. Organ data for pigs was generally lacking. Whether this lack of data is significant in the context of ingestion dose assessment will depend on whether or not the organs are consumed as part of a bush food diet. Anecdotal evidence suggests that they are not consumed.

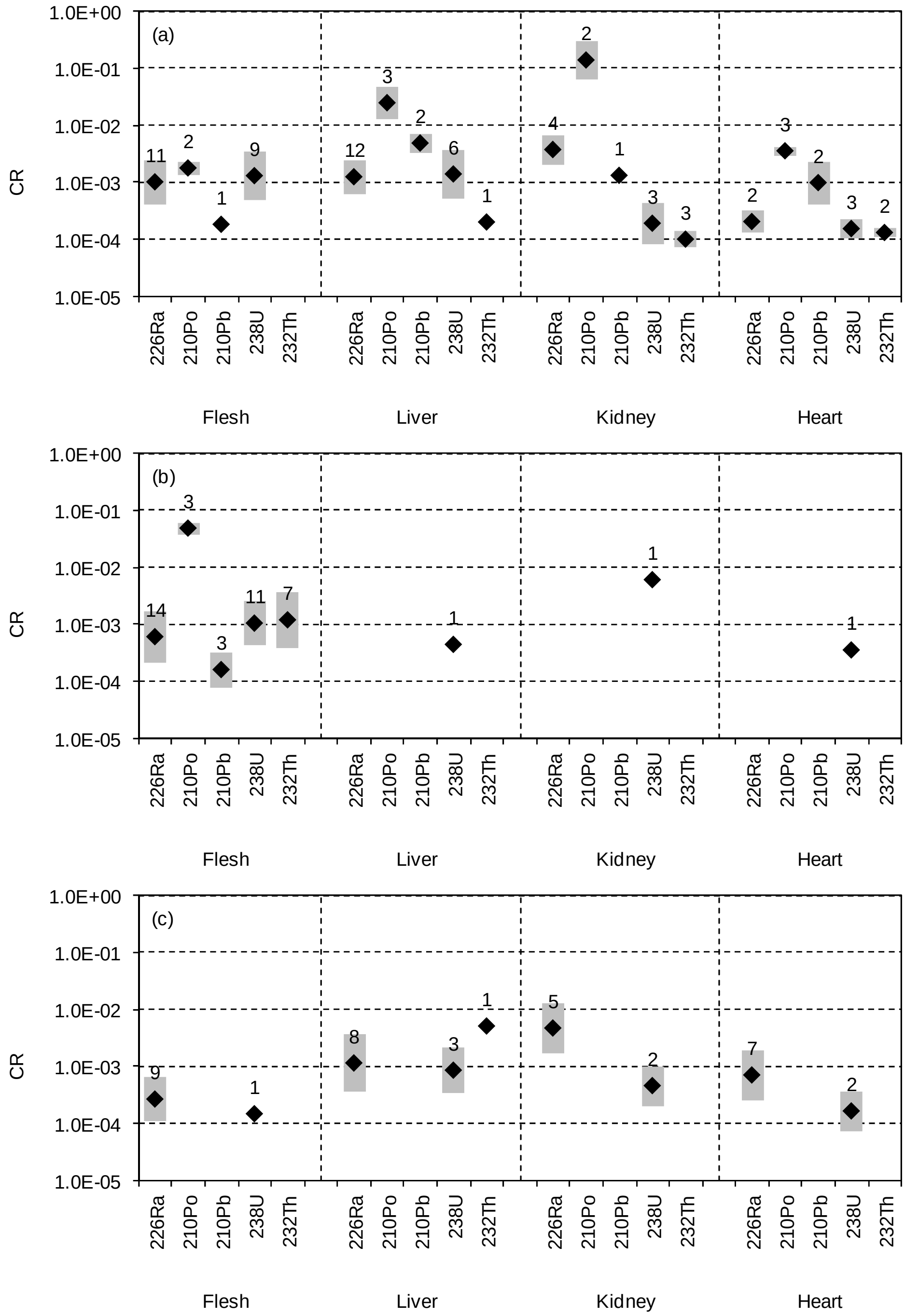

Figure 2 CR geometric means (black diamonds) and standard deviations (grey columns) on a fresh weight bush food to dry weight soil basis for (a) buffalo; (b) pig; and (c) wallaby (numbers indicate the number of bush food samples on which the geometric mean is based).

The results for wallaby (Figure 2(c)) indicate that ^{226}Ra and ^{238}U tend to concentrate more in the organs than in the flesh. The ^{226}Ra CR was highest for kidney and the ^{238}U CR was highest for liver. No ^{210}Po or ^{210}Pb radionuclide data was available for wallaby flesh or organs. This lack of data is significant as wallaby flesh and organs are consumed by Aboriginal people (Altman 1987). In addition, ^{210}Po has a high ingestion dose coefficient (ICRP 1995) and will likely be the radionuclide of highest radiological importance to ingestion doses from wallaby.

The product of the CR and ingestion dose coefficient can be used to indicate the potential radiological importance of the various bush food-radionuclide combinations. Table 1 ranks the ten combinations of highest potential radiological importance determined using the available CR values and adult ingestion dose coefficients from ICRP (1995). The ranking is dominated by ^{210}Po due to its relatively high CR in bush foods and high ingestion dose coefficient. The actual radiological importance of the various bush food-radionuclide combinations in the context of ingestion dose is also strongly dependent on quantities consumed. For this reason it is critical that a reference bush food diet for rehabilitation planning of the Ranger mine is developed.

Table 1 Bush food-radionuclide combinations of highest potential radiological importance

Rank	Bush food-radionuclide	CR	Ingestion dose coefficient (Sv Bq^{-1})
1	Buffalo-Kidney-^{210}Po	1.4×10^{-1}	1.2×10^{-6}
2	Pig-Flesh-^{210}Po	4.8×10^{-2}	1.2×10^{-6}
3	Buffalo-Liver-^{210}Po	2.4×10^{-2}	1.2×10^{-6}
4	Yam-^{210}Po	1.7×10^{-2}	1.2×10^{-6}
5	Yam-^{210}Pb	2.5×10^{-2}	6.9×10^{-7}
6	Yam-^{226}Ra	4.0×10^{-2}	2.8×10^{-7}
7	Fruit-^{210}Pb	9.1×10^{-3}	6.9×10^{-7}
8	Fruit-^{210}Po	3.8×10^{-3}	1.2×10^{-6}
9	Buffalo-Heart-^{210}Po	3.5×10^{-3}	1.2×10^{-6}
10	Fruit-^{226}Ra	1.3×10^{-2}	2.8×10^{-7}

Steps for completion

A reference diet indicating the types and quantities of bush foods consumed needs to be developed to enable ingestion dose assessments for rehabilitation planning purposes. This should be a priority task. ERA has advised that the Cultural Closure Criteria Working Group has been delegated the task of obtaining contemporary diet information for the development of a reference diet, although the timeframe for this is unclear at present. The reference diet should be developed and agreed among relevant stakeholders, including with traditional owners to gain the benefit of their knowledge on significant bush foods and diet composition.

eriss will continue with the measurement of radionuclides in terrestrial bush foods, particularly in animals. Pig flesh and organ samples were recently acquired from culls conducted on the Ranger project area and are currently being analysed for ^{226}Ra, ^{210}Po, ^{210}Pb, ^{238}U and ^{232}Th. Wallaby samples acquired recently will also be analysed for these radionuclides. Historical samples are being retrieved from the archive and analysed for missing radionuclides where possible. However, the analysis of historical samples precludes analysis for ^{210}Po due to half-life considerations. Any ^{210}Po now measured in historical samples would be due to ingrowth from ^{210}Pb rather than environmental transfer of ^{210}Po.

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Radionuclide transfer to wildlife

C Doering & A Bollhöfer

Introduction

The International Commission on Radiological Protection (ICRP) now recognises the environmental exposure of wildlife to ionising radiation as a distinct exposure category for which it has developed a radiological protection framework (ICRP 2007, ICRP 2008, ICRP 2009). Radiation doses to wildlife from radionuclides present in the environment from human activities should be considered and assessed under the famework where the potential for environmental effects exists or where otherwise required to do so. Nuclear fuel cycle activities (including uranium mining and site rehabilitation) can release significant quantities of radionuclides and may lead to environmental exposures of wildlife of potential concern.

The radiation dose to wildlife comes from radionuclides that are both internal and external to the organism. Radionuclides internal to the organism can account for a significant portion of the dose received. To calculate internal doses, information on radionuclide transfer from the environment to the organism is generally required.

The transfer of radionuclides to wildlife is typically quantified as the concentration ratio (CR) between the radionuclide activity concentration in the whole organism and that in the soil (terrestrial organisms) or water (aquatic organisms) that often function as reservoirs for nutrients and contaminants (Howard et al. 2013). Wildlife dose assessment tools, such as ERICA (Brown et al. 2008) and ResRad-Biota (USDOE 2004), use whole organism CRs to estimate radionuclide transfer in the calculation of internal doses.

Whole organism CRs for a range of organisms will be needed for wildlife dose assessments for site rehabilitation of the Ranger mine in order to demonstrate compliance with closure criteria for radiological protection of the environment. In this context, ***eriss*** has consolidated its data on radionuclide activity concentrations in biota and soil/water sampled from the ARR to facilitate the calculation of region-specific CRs. This paper presents preliminary whole organism CRs for ARR wildlife for ^{226}Ra and comparison with worldwide generic values recently derived by the International Atomic Energy Agency (IAEA).

Method

A bespoke spreadsheet tool (Doering 2013) was used to calculate CRs for ^{226}Ra in flesh for freshwater and terrestrial organisms grouped on the basis of suggested IAEA wildlife groups (Howard et al. 2013) (Table 1). The CRs were calculated relative to filtered water for freshwater wildlife and relative to soil for terrestrial wildlife. Only water and soil samples that were collected within one year of the organism being collected and from the same location were included in the CR calculation. The flesh CRs were converted to whole organism CRs using the whole organism to tissue conversion factors derived by Yankovich et al. (2010). The assumption is that the distribution of ^{226}Ra in the flesh and tissues of ARR wildlife would yield the same whole organism to tissue conversion factors.

Table 1 Wildlife groups and ARR species included in each group

Wildlife group	Species
Freshwater	
Bird-Herbivorous	Magpie goose (*Anseranas semipalmata*)
Crustacean	Shrimp (*Macrobrachium rosenbergii*)
Fish-Benthic	Black catfish (*Neosilurus ater*); Ell-tailed catfish (*Neosiluris hyrtlii*); Mouth almighty (*Glossamia aprion*); Sharp-nose grunter (*Syncomistes butleri*); Toothless catfish (*Anodontiglanis dahlia*)
Fish-Forage	Archer fish (*Toxotes chatareus*); Banded grunter (*Amniataba percoides*); Black bream (*Hephaestus fuliginosus*); Black-banded rainbowfish (*Melanotaenia nigrans*); Blue catfish (*Arius graeffei*); Bony bream (*Nematalosa erebi*); Chequered rainbowfish (*Melanotaenia splendida inomata*); Fork-tailed catfish (*Arius leptaspis*); Saratoga (*Scleropages jardini*); Sleepy cod (*Oxyeleotris lineolatus*); Spangled grunter (*Leiopotherapon unicolour*)
Fish-Piscivorous	Barramundi (*Lates calcarifer*); Long tom (*Strongylura kreffti*); Tarpon (*Megalops cyprinoides*)
Insect	Chironomid midge; Dragonfly
Mollusc-Bivalve	Mussel (*Velesunio angasi*)
Reptile	File snake (*Acrochordus arafurae*); Crocodile (*Crocodylus johnstoni*)
Terrestrial	
Mammal-Herbivorous	Buffalo (*Bubalus bubalis*)
Mammal-Marsupial	Wallaby (*Macropus agilis*); Bandicoot (*Isoodon macrourus*)
Mammal-Omnivorous	Pig (*Sus scrofa*)
Reptile-Carnivorous	Olive python (*Liasis olivaceus*)

Results

Figure 1 shows whole organism geometric mean CRs for ^{226}Ra transfer to ARR freshwater and terrestrial wildlife and comparison with the IAEA worldwide generic values.

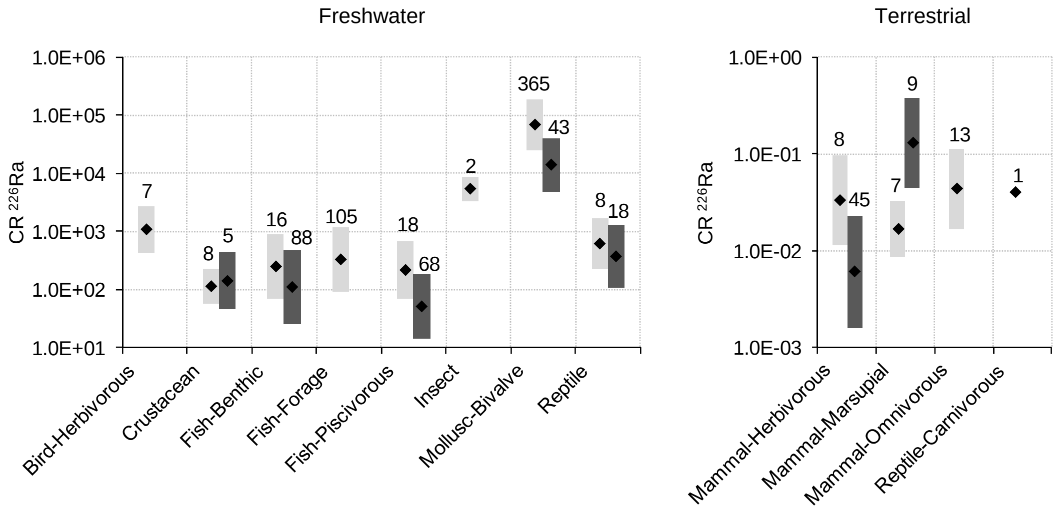

Figure 1 Whole organism CRs for ^{226}Ra transfer to freshwater and terrestrial wildlife in the ARR and comparison with IAEA worldwide generic values (black diamonds indicate the geometric mean value, light grey vertical bars indicate one geometric standard deviation in ARR values, dark grey vertical bars represent one geometric standard deviation in IAEA values, the number above the vertical bar indicates the number of samples on which the geometric mean is based)

^{226}Ra CRs for ARR freshwater wildlife were generally greater than the IAEA values, particularly for the fishes and mollusc-bivalve (freshwater mussel) wildlife groups. This is believed to be the result of low calcium and magnesium concentrations in water in the ARR. Radium has similar chemical properties to calcium and magnesium, meaning that wildlife 'see' ^{226}Ra as an essential nutrient and preferentially take it up from the water column if calcium and magnesium concentrations in water are low.

^{226}Ra CRs for ARR terrestrial wildlife were one order of magnitude greater than the IAEA value for mammal-herbivorous and one order of magnitude less than the IAEA value for mammal-marsupial. The mammal-marsupial wildlife group is effectively uniquely Australian. The IAEA value for ^{226}Ra for this wildlife group comes from studies of kangaroos in semi-arid and desert areas in Australia (Johansen & Twining 2010). Higher dust loads in these dry environments is likely to enhance radionuclide transfer via the inhalation and ingestion pathways, resulting in a higher CR than for marsupials (wallaby and bandicoot) in the ARR.

Overall, the comparison of ARR and IAEA whole organism CRs for ^{226}Ra for selected wildlife groups shows that worldwide generic values can either under- or over-estimate radionuclide transfer to ARR wildlife. The implication is that region-specific CRs should be used in wildlife dose assessments for rehabilitation planning of the Ranger mine, as this will help to ensure the integrity of the assessment.

From Figure 1 it is apparent that ***eriss*** has CRs for ^{226}Ra for some wildlife groups not included in the IAEA data summary. This includes bird-herbivorous, fish-forage and insect from the freshwater wildlife groups and mammal-omnivorous and reptile-carnivorous from the terrestrial wildlife groups. ***eriss*** participates in the IAEA Modelling and Data for Radiological Impact Assessments (MODARIA) programme to help improve global knowledge of radionuclide transfer in the environment and will continue contributing its CR results for radionuclide transfer to ARR wildlife to this programme.

Steps for completion

The next step is to identify a set of representative organisms on which to base wildlife dose assessments for rehabilitation planning of the Ranger mine. Representative organisms are real organisms typically found living in or utilising the impacted environment. They generally have characteristics and perform ecological functions that are representative of the range of diverse organisms present. Their selection is a critical step in the wildlife dose assessment that should be done in consultation with relevant stakeholders, including with traditional owners to gain the benefit of their knowledge on the cultural significance of various plant and animal species.

After an agreed set of representative organisms has been established, an evaluation can be made of whether the existing radionuclide data adequately encompasses the selected representative organisms or whether additional targeted sampling is required.

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2.6 Monitoring

A remote sensing framework for environmental monitoring within the Magela Creek catchment

RE Bartolo, T Whiteside, K Pfitzner & A Esparon

Introduction

Remote sensing is a viable and cost effective tool for monitoring the rehabilitated landform and offsite area in the rehabilitation phase of Ranger mine site and into the future. The Ranger Project Area will have reduced access over time as the area progresses through the rehabilitation phase and is relatively large in areal extent (with the inclusion of offsite monitoring). Therefore, an *operational* monitoring programme using remote sensing at suitable spatio-temporal scales will be a fundamental component to an integrated monitoring programme. A strategic direction for research and review of ***eriss***' remote sensing programme has not previously been undertaken.

The main objective of this project is to document and communicate the strategic direction for SSD's remote sensing programme in the context of the KKNs and environmental monitoring requirements in the Magela Creek catchment. Specific aims are to:

- undertake a review of the remote sensing projects undertaken to date by eriss.
- review the current KKNs and assess which science questions require remote sensing, either to answer research questions or to provide information.
- develop a conceptual framework for environmental monitoring in addressing KKNs using remote sensing.

This is not a new project and has previously been proposed under KKN 5.1.1. This project underpins the remote sensing monitoring programme and should be reported under KKN 2.6. The main output of this project will be a conceptual framework that delivers a routine monitoring and *ad hoc* data capture capability based on remote sensing. The routine monitoring will cover mine site rehabilitation (landform and off-site) and landscape scale monitoring. The *ad hoc* data capture will account for unplanned events where information may be required (e.g. extreme events, cyclones, landslips, mine incidents). In developing a conceptual framework, a landscape ecology approach will be used, whereby spatio-temporal characteristics of disturbance agents (e.g. uncontrolled fire, feral animals, floods, weeds) will be defined for the Magela Creek catchment. The characteristics of the disturbance agents will then be mapped against the spatio-temporal characteristics of remote sensing tools at specific scales (spatial and temporal) so that monitoring can be conducted in an integrated manner and will be fit for purpose. Site specific components will be accounted for (e.g. monitoring of vegetation health on the rehabilitated landform).

Monitoring and research in the Magela Creek catchment, using remote sensing requires the development of a remote sensing monitoring framework. This framework needs to take into account the following:

- Due to the highly seasonal environment characterised by a wet season dominated by ubiquitous cloud cover and a dry season dominated by atmospheric smoke resultant from wildfires, the use of optical data can be limited.
- The scales of interaction, in terms of spatio-temporal characteristics of disturbance agents, such as introduced weeds, feral animals, fire, climatic agents (e.g. cyclones, floods and climate change impacts such as sea level rise) need to be defined in relation to remote sensing data sources.
- Cross-institutional participation in developing and reviewing such a framework on a long-term basis and the potential monetary cost.

A robust framework will be developed and implemented through this project that will have the capacity to incorporate all of SSD's remote sensing activities in the future.

Other outcomes for the project are:

- Baseline data for a number of biophysical parameters that SSD does not currently have (e.g. biophysical characterisation of environments over a time period to enable seasonal characterisation compared with inter-annual variability).
- That SSD will have the capability to routinely provide assurance in environmental monitoring through maps and remote sensing imagery that are easily understood by a number of stakeholder groups within the region.
- A reduction in long-term fieldwork, information provision and management costs.

Progress to date and steps for completion

A review of remote sensing projects undertaken at ***eriss*** to date and a review of the KKNs, including identification of which science questions can either be answered by or benefit from the use of remote sensing, has been completed. Work has commenced on the development of a conceptual framework for environmental monitoring in addressing KKNs using remote sensing. The review will be documented as an internal report and be made available to stakeholders in early 2014.

Demonstrating the utility of unmanned aerial vehicles (UAVs) for monitoring rehabilitation and revegetation of the Ranger mine site

T Whiteside & RE Bartolo

Background

A critical component in assessing the success of mine site rehabilitation is a representative monitoring programme that can detect changes in ecosystem condition as they relate to mine closure criteria. At present, no such monitoring programme has been designed, for the impending closure of the Ranger mine site. To be effective, research and development of a remotely sensed monitoring programme for minesite rehabilitation needs to commence in advance of the rehabilitation so the programme can commence concurrent with the rehabilitation activities.

Monitoring the success of mine site rehabilitation requires the assessment of numerous biophysical indicators within the landscape which may including, but is not limited to, tree and canopy density, ground cover (vegetative and bare ground), tree height, vegetation rigour and health, hydrological change and erosion. For the rehabilitated Ranger mine site, the impacts of landscape disturbances such as fire, weeds and extreme events also need to be measured and modelled. In order to make such assessments, data and information on these indicators is required at suitable temporal scales (frequency) and within an appropriate spatial sampling resolution and design. This will enable landscape change and changes in status of landform processes over time to be identified and quantified, and overall landscape condition (including vegetation health) to be monitored. Early detection of trends in long-term landscape condition can allow for targeted management activities to address potential negative environmental impacts and/or inappropriate management methods. Such detection also allows for early intervention to mitigate potential environmental impacts.

Design proposals for a successful monitoring programme for mine closure also need to be cost effective and have the capability to be responsive to events that may impact on mine closure success. Field sampling in remote tropical forests has been shown to be time-consuming, financially expensive, and logistically challenging (Gardner et al. 2008). Therefore, due to resource limitations and logistic challenges, field sampling, while able to inform or assess the accuracy of remotely sensed data products, may not be able to be conducted at the necessary spatial and temporal frequency to analyse variation of vegetation growth and condition. In addition, plot-based field sampling is only representative of portions of the cover and is limited by size and location, and lacks the temporal and spatial coverage possible with remote sensing. So, although more information may be obtained from plots than from a remotely sensed product, there is no information provided between plots compared with the continuous data produced by remote sensing. Fundamental to any proposed monitoring scheme is to ensure that the sampling design is capable of detecting change over and above natural variability. Spatial and temporal variation in vegetation communities may exhibit superficial change in condition that can mask longer term trends. Measurement accuracy, along with sufficient sampling frequency and intensity can help differentiate between the two.

There are limitations associated with both the use of satellite and traditional airborne remote sensing for this type of work. Satellite remote sensing data at the highest spatial resolution (2 m for multispectral data) are costly and not always available due to previous tasking and cloud cover. This limits the ability to use satellite based imagery at the temporal resolution required for monitoring rehabilitation. Additionally, the spatial scale of the highest spatial resolution satellite data available will not be able to detect small changes in surface condition. Airborne remote sensing data is also costly with the current requirement for deployment from interstate for monitoring Ranger limiting its application for temporal analysis at this site. Therefore the monitoring of vegetation condition (i.e. phenology and seasonal water stress) via satellite or more traditional manned airborne remote sensing requires a temporal frequency of data capture that is difficult to achieve at the Ranger mine due to cost and availability.

Sensors onboard unmanned aerial systems (UASs) capture very high spatial resolution data that not only includes imagery of sub-decimetre resolution but also are the basis for very high resolution photogrametrically derived products including point cloud data, digital surface models (DSMs) and digital elevation models (DEMs). Some of the advantages UASs have over satellite derived imagery and traditional aerial photography include the higher spatial resolution, lower costs for high temporal resolution sensing, the ability to operate below cloud, and a rapid response time to events. In a minesite closure context, data captured by a UAS will be of a spatial and temporal scale relevant to the monitoring of minesite rehabilitation (with particular emphasis on monitoring surface conditions, landform changes and vegetation growth).

Introduction

The objective of this project is to trial the use of UASs for obtaining very high spatial resolution imagery that can be used to monitor mine site rehabilitation in a time and cost effective manner. The key outcomes of the project are:

1. Demonstration that a UAS can be used to successfully capture data required for environmental monitoring in the Ranger context.
2. Collection of data for focus areas that can be analysed to demonstrate applications for environmental monitoring.

The project will address the following KKNs:

- *KKN 2.1.1- Ranger rehabilitation- defining the reference state and baseline data:* This project will demonstrate the utility of UAS derived data in producing information concerning the surrounding environment that will assist in the characterisation of the reference state.
- *KKN 2.6.1- Monitoring of the rehabilitated landform:* This project will demonstrate the utility of UAS derived data in monitoring revegetation success in the first instance. It is anticipated the UAS data can also provide information on the landform and surface.
- *KKN 2.6.2- Offsite monitoring during and following rehabilitation:* This project will demonstrate the utility of UAS derived data in the future monitoring of erosion, flora, weeds and fire surrounding the rehabilitated landform.

The project is a collaboration between ***eriss*** and the University of Queensland's Centre for Mined Land Rehabilitation (CMLR) which is researching UAS applications for monitoring minesite impacts and rehabilitation.

Methods

Demonstration sites and imagery capture

Between 8th and 10th October 2013, UAS flights were conducted at four sites (Figure 1). Two of the sites were located within the Magela Creek floodplain: (1) an area including Buffalo Billabong and northern section of Mudginberri billabong and (2) the Djarr Djarr locality within the Jabiluka Mineral lease where there is a number of para grass plots located. Imagery was additionally captured for two of the vegetation reference plots established by ERA and SSD, located within the savanna south of Jabiru, and for the southern extent of the Georgetown analogue area. All areas flown with the UAS were outside the 3 nautical mile (nm) restriction around the Jabiru airstrip. Areas within the 3 nm boundary can be flown with CASA approval, which was not obtained in time for this demonstration project.

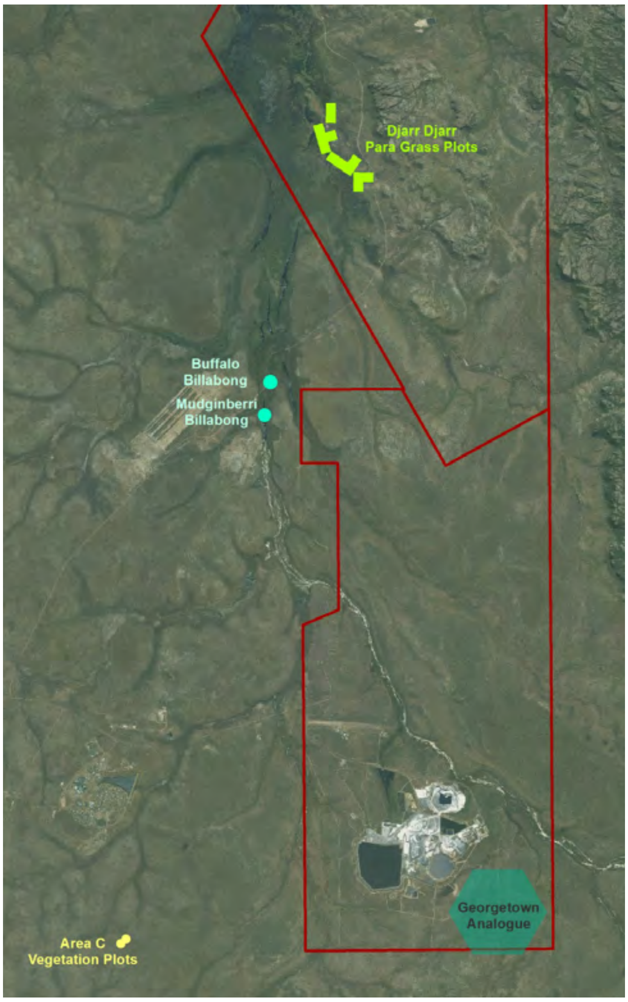

Figure 1 Location of the demonstration sites for which imagery was acquired.

UAS: platforms and sensors

The Swampfox fixed-wing UAS (Figure 2) from Skycam NZ (www.kahunet.co.nz) was used as the platform for image capture at all of the sites except the Georgetown Analogue area. The UAS, owned and operated by the CMLR at the University of Queensland, contains a flight control system (FCS) that is global positioning system (GPS) guided and controlled via radio telemetry from a ground control station. Flights were conducted at an average 400 ft altitude and at a cruising airspeed of 60 km/hr. The site was covered by flying overlapping transects. The Swampfox's payload was two modified Sony NEX5 cameras set up for aerial photography capture; one standard Red Green and Blue (RGB) and one converted to capture Near Infra-Red (NIR) (≥720 nm) (Figure 3). The cameras were triggered simultaneously by the autopilot at a rate of one frame per 1.8 seconds. The GPS location, date and time from the FCS's GPS are recorded in the image metadata. The capture ensured that for each flight there was at least 75% forward overlap and 60% side overlap between images.

Due to increased tree cover and less available open space for launching at the Georgetown analogue site, a smaller UAS was deployed. The platform based on the TekSumo flying wing (www.hobbyking.com) was used due to its much lighter weight (1.1 kg) and corresponding increased takeoff angle (Figure 4). The platform contains a GPS guided autopilot with telemetry. The sensor on this UAS was a Canon IXUS 135 compact digital camera capturing RGB standard photography.

Figure 2 The Swampfox UAS platform.

Figure 3 The modified digital cameras carried onboard Swampfox.

Figure 4 The UAS based on the Teksumo flying wing.

Preliminary imagery

Imagery was captured at an average Ground Sampling Distance (GSD) of 5cm. The number of images captured depended on the areal extent of the demonstration site. For example, for the vegetation monitoring plots 1538 images were captured compared with 741 images captured for a small portion of the Djarr Djarr para grass monitoring plots. Figure 5 shows some preliminary imagery for Buffalo Billabong.

Progress and steps for completion

The team from the CMLR are currently processing the imagery into orthophoto mosaics and DTMs. There have been challenges in processing the data, mostly due to a lack of overlap between photos. A combination of the extreme conditions and an inefficient voltage regulator resulted in the cameras overheating and stopping during the first mission, reducing the number of photos available with sufficient overlap. On other flights there was also insufficient overlap between photos due to the frame capture rate being 1.8 frames per second (fps) also attributable to the voltage regulator. The consequence of insufficient overlap is the automatic algorithms in the photogrammetric software are not able to find common features in enough photos. The overheating and stopping issue was remedied by increasing air flow through the fuselage and camera module. The CMLR team, in combination with the UAV vendor, are now working with a new voltage regulator to achieve a frame rate of 1 fps, which will ensure sufficient overlap in future missions. Once these issues are resolved, there are a number of mapping applications we will undertake to demonstrate the application of data obtained from a UAS, including: vegetation species identification across the range of habitats surveyed; measuring the extent of aquatic weeds; and quantifying damage from feral animals. These mapping applications will be undertaken in collaboration with the CMLR team.

Figure 5 Preliminary imagery captured with the Swampfox UAS for Buffalo Billabong.

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Geological province of sediment within the Magela Creek catchment

A Esparon, K Pfitzner & RE Bartolo

Background

The identification of minerals and rocks using spectroscopy is well documented, and there have been many in situ and laboratory studies (Clark et al. 1990) analysing the reflectance of sediments (see Pfitzner et al. 2013). Spectral discrimination underpins all remote sensing approaches including contextual (object-based) methods. Spectral-reflectance-based fingerprinting for documenting suspended sediment sources is an emerging application (e.g. Martínez-Carreras et al. 2010). Suspended sediments samples have been collected routinely from ***eriss'*s** continuous surface water monitoring programme located both upstream and downstream of Magela and Gulungul creeks, including the 2007 landslide event. The origin of the suspended sediment is from water samples that have been filtered and deposited onto filter paper. It is the filter paper with the sediment deposited that is spectrally measured.

The spectra of sediments will be measured from 400 to 2500 nm using a very high resolution portable spectrometer (spectral resolution between 3–12 nm full-width-at-half-maximum in the visible-near infrared and shortwave infrared, respectively). The purpose of these measurements is to determine the separability of spectral end-members based on reflectance magnitude, absorption position, width and depth. These data may then be used to quantify minesite derived sediment loads on both Magela and Gulungul creeks after construction of the final rehabilitated landform at Ranger Uranium Mine (RUM).

Introduction

A database is being developed for capturing data from spectral characterisation of sediments sourced from the trial landform (TLF), RUM and the surrounding environment of the Ranger Project Area (RPA). Spectral measurements of varying grain sizes, including suspended sediments, combined with Physico-chemical analyses are being used to obtain high resolution spectral measurements across the visible near-infrared (VNIR) and short-wave infrared (SWIR). The purpose is to investigate the spectral response of sediments at varying grain sizes to determine the spectral separability of source sediments (e.g. waste rock from laterite mix). This project will also enable us to develop an integrated reference library for sediments which will be used as a reference for remote sensing data analysis. Ultimately, a mathematical model to quantify sediment loads derived from the rehabilitated landform in both Magela and Gulungul creeks will be developed. This model will quantify mine derived sediment loads in local creek systems due to the construction of the final rehabilitated landform at RUM. This work complements the vegetation component of KKN 2.6.1 in developing a spectral reference library.

The research aims are to determine:

- If *in situ* and laboratory measured TLF sediments from the waste rock and laterite mix are spectrally separable.
- Whether mine-derived suspended sediments from the TLF are uniquely identifiable and if these spectral signatures can be used as a surrogate to identify sediment source in down water streams.
- If there is a difference in the spectral features of suspended sediment from upstream and downstream of Ranger uranium mine due to the construction of the TLF (Magela and Gulungul upstream and downstream continuous monitoring gauging stations).
- What the spectral responses of the soils of the vegetation analogue sites are and whether these are spectrally separable.
- If a gap analysis is required to potentially reveal any missing sediment source provinces that should be accounted for in the spectral database.

Progress to date and steps for completion

In situ spectra (~320 survey points) have been measured on the trial landform surface. Laboratory spectra of suspended sediment from Magela Creek, Gulungul Creek, and the erosion plots on the trial landform using the method outlined in Pfitzner et al. (2013) are being collected. The resulting spectra and metadata, including photos, will be stored in SSD's spectral library (Pfitzner et al. 2008). Spectra will be compared to the Physico-chemical parameters measured for water quality and representative samples of sediments will be analysed for chemical and mineralogical content by X-ray diffraction (XRD) and X-ray fluorescence (XRF). The systematic comparison and assessment of mathematical algorithms for varying weighted contribution of the end-members will be undertaken. Feasibility studies will be performed to determine the spectral separability of sediment sources to determine if the minesite rehabilitated landform can be isolated from other sediment sources to allow assessment of minesite contribution to sediment load.

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Development of a spectral library for minesite rehabilitation assessment-vegetation components

K Pfitzner, A Esparon & RE Bartolo

Work on this project resumed in April 2013 after being inactive for over two years. A brief history of work undertaken is provided as well as the current activity.

Introduction

A spectral database of land cover end members pertinent to remote sensing for minesite rehabilitation assessment is being developed, which specifically includes exotic and native vegetation groundcovers. The purpose of the database is to investigate the spectral response of the phenological condition when data was captured, and, analyse the change in phenology both within and between species over time. Spectral data collated in the database may be used to determine the most appropriate remote sensor for assessing minesite rehabilitation at different times of the year.

The research questions were: How do spectral responses of ground cover species change over time? Can groundcover species be distinguished using ground-based reflectance spectra and, if so, what spectral resolution (spectral selectivity or full-width-half-maximum and spectral sampling interval) is required? Can we distinguish species by their spectral response at differing phenological stages? And, Can we use spectral response to inform the use of remote sensing in monitoring revegetation during minesite rehabilitation?

Background

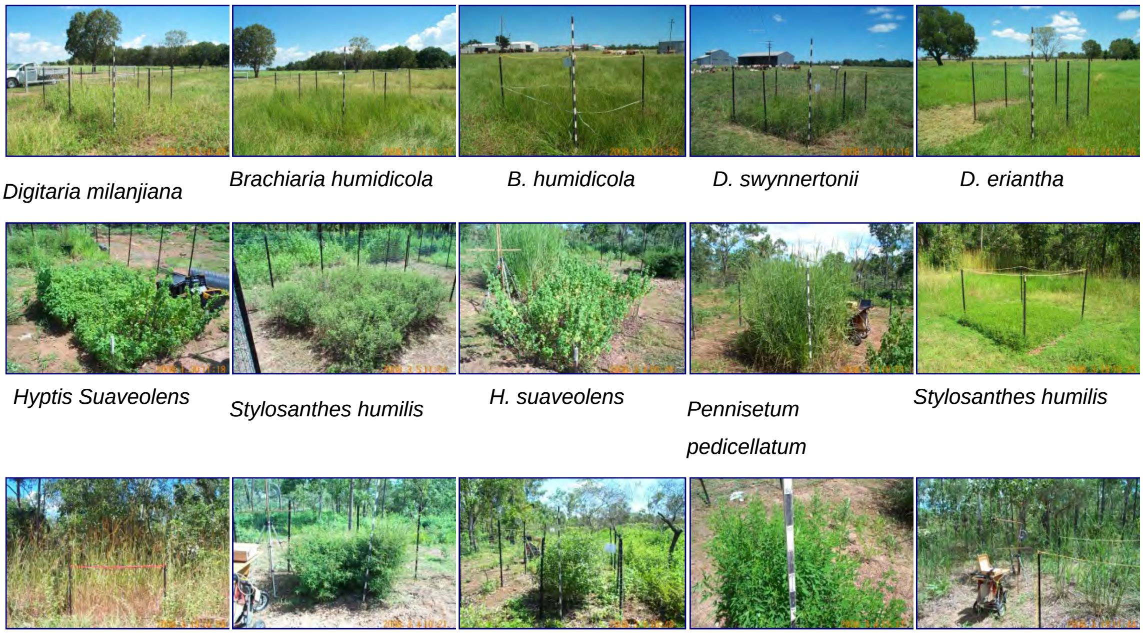
Exotic and native ground cover species including small shrubs, grasses, herbs and vines were identified for spectral sampling. Suitable ground covers unlikely to be disturbed and located nearby to reduce travel time for frequent sampling were located at Berrimah Farm, CSIRO Berrimah and Crocodylus Park. Over thirty species including the weed species: *Aeschynomene Americana, Andropogan gayanus, Chloris inflata, Hyptis suaveolens, Passiflora foetida, Pennisetum pedicellatum, P. polystachion, Urochloa maxima* and *U. Mutica,* and native grasses including *Panicum Mindanese, Schizahrium* spp. *and Sorghum stipodeum* were identified. Emphasis was placed on selecting exotic species found at or around Ranger and Nabarlek mines (Pfitzner & Bollhöfer 2008). Examples of these plots are illustrated in Figure 1.

Method

Standards for reflectance spectral measurements of temporal vegetation plots were developed and published (Pfitzner et al. 2011). These standards, including protocols to acquire robust in situ spectral data, metadata and laboratory calibrations were used to provide a consistent and repeatable method for recording spectra that minimized the influence of extraneous factors in spectral measurements (Pfitzner et al. 2005 & 2006).

Spectral sampling was undertaken in 2006 and 2007. The frequency of measurement varied but each plot was typically measured fortnightly or monthly. For each plot, three subsamples (a, b and c) were measured for each date of measurement to capture any spectral variation within the species plot. For each sample, between three and six replicates per sample was measured. Metadata and photographs were recorded.

The data (spectra and metadata) were stored in SSDs Spectral database (Pfitzner et al. 2008, Pfitzner et al. 2010). Spectra (from Berrimah Farm and CSIRO to date) were imported into a SAMS (Spectral Analysis and Management System) database into folders by date.



Digitaria milanjiana | *Brachiaria humidicola* | *B. humidicola* | *D. swynnertonii* | *D. eriantha*

Hyptis Suaveolens | *Stylosanthes humilis* | *H. suaveolens* | *Pennisetum pedicellatum* | *Stylosanthes humilis*

Sorghum stipodeum | *Aeschynomene americana* | *S. Australis* | *Crotalaria pallida* | *P. polystachion*

Figure 1 Examples of vegetation plots

Each group of replicates within a subsample was averaged and the standard deviation computed (equation 1 and equation 2). Referring to Figure 2 as an example plot in time, there were five replicates measured in subsample a (Figure 2a), six replicates in subsample b (Figure 2b) and six replicates in subsample c (Figure 2c) that were used to calculate the average and standard deviation of a, b and c (Figure 2d). The Coefficient of Variation was also calculated (equation 3). The averages of a, b and c were then computed (equation 4) to provide the endmember (single spectrum) for a given site and sampling time (Figure 2e).

$$Average = \bar{s_x}(\lambda_k) = \frac{1}{n}\sum_{i=1}^{n} s_i(\lambda_k) \qquad (1)$$

Where : n = number of replicates in each subclass

x = subclass {a,b,c}

$s_i(\lambda_k)$ = Are the weighing factors for each spectral element k for the n^th replicate.

$\bar{s_a}(\lambda_k)$ = Average of subclass a

$\bar{s_b}(\lambda_k)$ = Average of subclass b

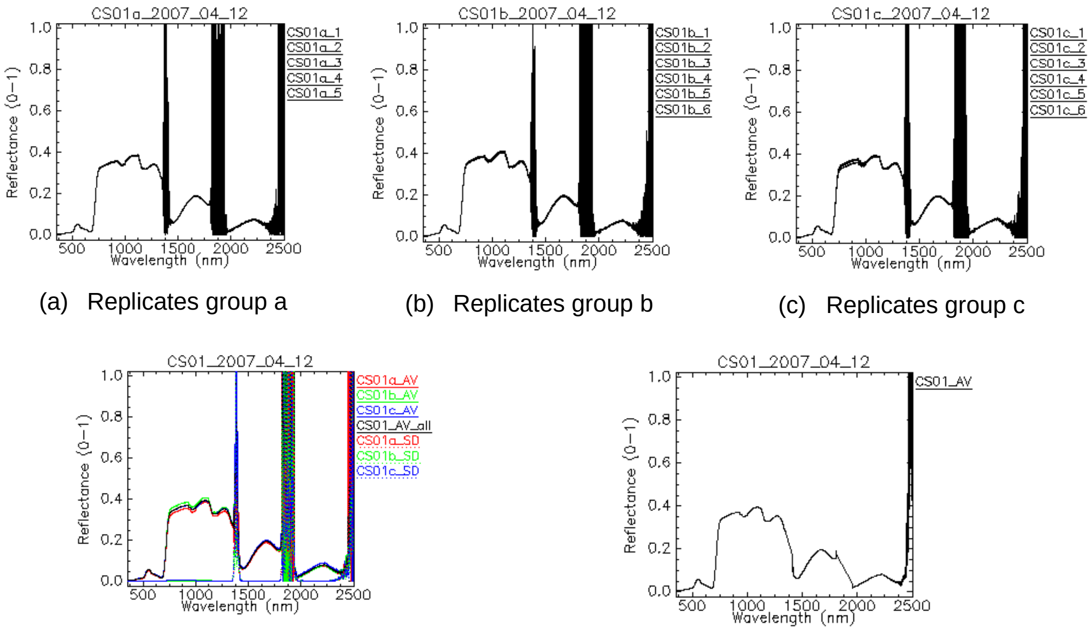
$\bar{s_c}(\lambda_k)$ = Average of subclass c

$$Standard\ Deviation = s'_x(\lambda_k) = \sqrt{\frac{1}{n-1}\sum_{i=1}^{n}[s_i(\lambda_k) - \bar{s}_i(\lambda_k)]^2} \qquad (2)$$

Where: $s_a^{'}(\lambda_k)$ = Standard deviation of subclass a
$s_b^{'}(\lambda_k)$ = Standard deviation of subclass b
$s_c^{'}(\lambda_k)$ = Standard deviation of subclass c

$$Coefficient\ of\ Variation = \frac{s_a'(\lambda_k)+s_b'(\lambda_k)+s_c'(\lambda_k)}{\overline{s_a}(\lambda_k)+\overline{s_b}(\lambda_k)+\overline{s_c}(\lambda_k)} \quad (3)$$

$$Endmember = \overline{s_a}(\lambda_k) + \overline{s_b}(\lambda_k) + \overline{s_c}(\lambda_k) \quad (4)$$



(a) Replicates group a
(b) Replicates group b
(c) Replicates group c
(d) Averages and SD of a, b and c
(e) EndmemberCS01-2007-04-12

Figure 2 The processing of spectra for one species of one date.

Spectra were exported into ENVI spectral library and resampled to eliminate atmospheric water noise between 1354-1409 and 1811-1952 nm. The endmember signature (example (e) CS01_2007_04_12_AV_all in Figure 2), representing a given species for a single date in phenological condition was then saved as a spectral library.

Results

The processing of spectra from Berrimah Farm (*Digitaria milanjiana*, *Brachiaria humidicola, D. swynnertonii, D. eriantha* and CSIRO Berrimah (*Stylosanthes humilis, S. stipodeum, P. polystachion, P. foetida* (vine), *P. pedicellatum , U. maxima, B. humidicola, and Melinis repens*) are being finalised. It was expected that the averages of subsamples a, b and c would be used to look at intra-species variability, but the method of replication has shown that very few species processed to date show variability at a single time, and when they do, variability may be attributed to windy conditions changing the field-of-view.

The four grasses at Berrimah Farm were not a high priority in terms of a threat to mine rehabilitation success, but these plots were established opportunistically as they represent dense and homogenous stands of introduced pastoral grasses that are also present in the

Alligator Rivers Region. Further, they are floristically similar, being flowering angiosperms from two genera, and the selected species have a similar structure including the same growth form. They were expected to be spectrally similar, but a visual analysis indicates some differences.

Examples of six species over four sampling dates from CSIRO Berrimah are provided in Figure 3. This subset of data illustrates the phenology of species over time. Vegetation reflectance is primarily influenced by the optical properties of plant materials (including proteins, lignin, cellulose, sugar and starch) which are composed largely of hydrogen, carbon, oxygen and nitrogen. The absorption bands observed in vegetation arise from vibrations of C-O, O-H, C-H and N-H bonds as well as overtones and combinations of these vibrations. Changes in spectral response of vegetation can be attributed to the activity of photosynthesis (400-700 nm) as a result of leaf pigments, particularly chlorophyll a and b, carotenoids and xanthophylls. Cell structure controls reflectance in the near-infrared primarily from the internal structure of plant leaves as a function of the number and configuration of air spaces that form the internal leaf structure. The near-infrared spectra of leaves results from a combination of scattering processes and overlapping absorptions arising from water and biochemical components. Water content controls reflectance in the mid infrared. Although the general shape of the spectral curve may be similar for all green vegetation, changes in reflectance occur through variations in amplitudes of the curve. Differences in chlorophyll and water absorption positions and reflectance magnitude changes across regions of the spectrum may occur both between and within species, and of course, vegetation stress, senescing and desiccation all produce changes in the spectrum.

Referring to Figure 3, not all species senesce at the same time, despite all species experiencing similar environmental conditions at the CSIRO site. *Passiflora* remains green over the growing seasons illustrated, while the other species all show some senescence and drying. *P. polystachion* (perennial mission grass) still shows photosynthetic activity in August and is only beginning to senesce in July, whereas *P. pedicellatum* (annual mission grass) does not. A statistical, rather than visual analysis will be undertaken in the near future.

These preliminary data and results show that with a well-designed approach to collecting field spectral measurements and metadata, extraneous factors can be accounted for, accurate post-processing of spectra can be performed and the first database of northern Australian spectra relevant to the mine environment can be developed.

Further work

The results of the spectral sampling from Berrimah Farm and CSIRO are being documented in Internal Reports and will be published.

The species sampled at Crocodylus Park will be analysed in SAMS and endmembers exported. Species to be analysed include: *Hyptis suaveolens*, *Stylosanthes humilis*, *P. pedicellatum*, *Calopogonium mucunoides* (Calopo vine), *U. mutica* (Para grass), *Stachytarpheta cayennensis* (Snake weed – white flowers), *Sida cordifolia* (Flannel weed), *S. australis* (Snake weed – purple flowers), *P. mindanese*, *Chloris inflate*, *Cynodon dactylon*, *Heteropogon contortus, Schizachyrium* sp, *Crotalaria goreensis* (Gambia Pea), *Stylosanthes hamata* (Caribbean stylo), *Senna* sp (Sicklepod), *Lemna* sp. (Duckweed), and, *A. gaya*nus (Gamba grass).

A statistical approach to analyse the temporal change of a given species and variation between species is being developed. Spectra will also be resampled to hyperspectral and multispectral platforms and the same statistical approach applied with the spectrally subsampled data to determine the implications for remote sensing of species present during the rehabilitation phase of Ranger mine site.

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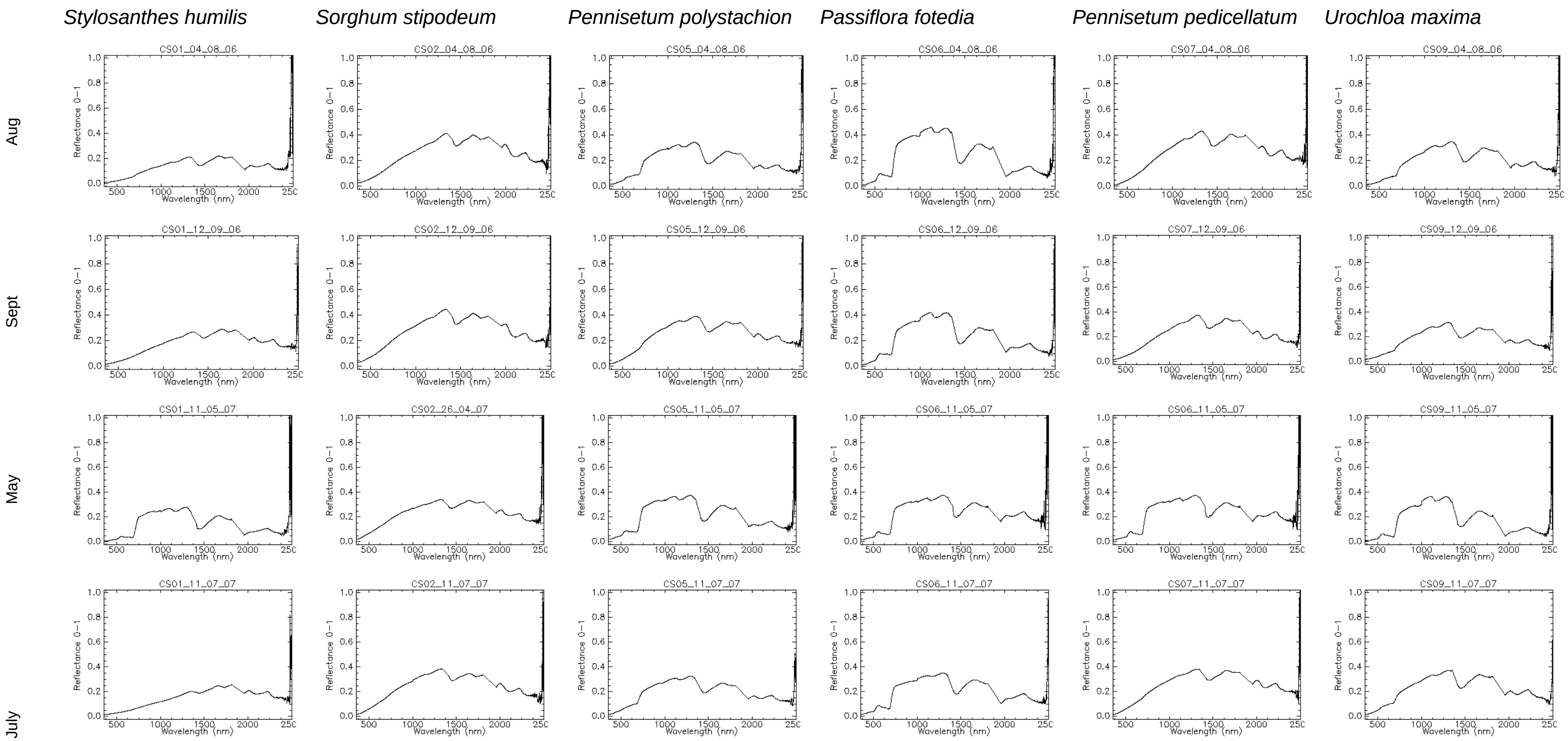

Figure 3 Examples of temporal spectral endmembers for selected ground cover species

Development of a low cost method and device for continuous monitoring of vegetation regrowth on a rehabilitated minesite.

A Esparon & RE Bartolo

Introduction

The process of decommissioning and rehabilitation of a uranium mine is a relatively new activity, therefore sustainable rehabilitation strategies and design options need to be developed according to geographical and climatic conditions. Ideally, the goal is to establish a plant community in line with the natural surrounding bushland which develops into a mature self-sustaining ecosystem (Daws 2008).

Results exist that demonstrate tree composition and size are useful traditional indicators of successful vegetation development on a rehabilitated landform (Ludwig et al. 2003). The method of re-vegetation of the Ranger Trial Landform (TLF) will form the model for the rehabilitation of the entire Ranger project area. The success of this rehabilitation model will also assist in final closure criteria of legacy first generation uranium mines.

ERA has a legal obligation to revegetate the Ranger Project Area (RPA) using local native plant species in a density and abundance similar to those existing in nearby areas. According to conditions of authorisation for Ranger mine (http://www.environment.gov.au/system/files/resources/9cc1f5c4-18ad-4604-93bf-e9efd8390ee0/files/ranger-ers.pdf) ERA must "*revegetate 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*".

It has been identified (Gellert 2012) that species such as *Eucalyptus tetrodonta* and *E. miniata* are of critical importance to ongoing revegetation of the mine site and the establishment of a sustainable eco-system.

In support of these objectives, extensive research has been undertaken to identify effective methods of rehabilitating the disturbed foot print of the Ranger Project Area (RPA). To date, key differences in whole plant water relations between *E. tetrodonta* and *E. miniata* and the implications of these for rehabilitation on a reconstructed landform have not been studied.

The literature indicates that a number of research studies have been undertaken on the effects of north Australian climatic variables and the response of *E. tetrodonta* and *E. miniata* within the natural environment. However, there is very limited research on seasonal climatic changes and the eco-physiological responses associated with these species on landscapes disturbed by mining.

A traditional method of remote monitoring of plant status involves using a field spectrometer and calculating indicators such as the commonly used Normalized Difference Vegetation index (NDVI) (Huete & Liu 1994).

$$NDVI = \frac{NIR - Red}{NIR + Red} \quad (1)$$

where NIR and Red are reflectance in the electromagnetic spectrum near the infrared and red bands respectively. NDVI is a common measurement of the status of plants. Healthier plants have more chlorophyll and will absorb more red light and reflect an increase in light in the near infrared region, therefore having a higher NDVI (Tucker 1979).

Light emitting diodes (LEDs) are semiconductor devices that emit incoherent light in narrow spectral bands when a sufficient voltage is applied across its terminals. Conversely, when exposed to light the LED is most sensitive to light with similar energy to the LED peak wavelength. This induces an electric current across the LED terminals that corresponds to the intensity of incident light within this narrow spectral band (Figure 1). LEDS are cheaply available with a variety of over 180 different peak sensitivities over the spectral range 400nm to 950nm.

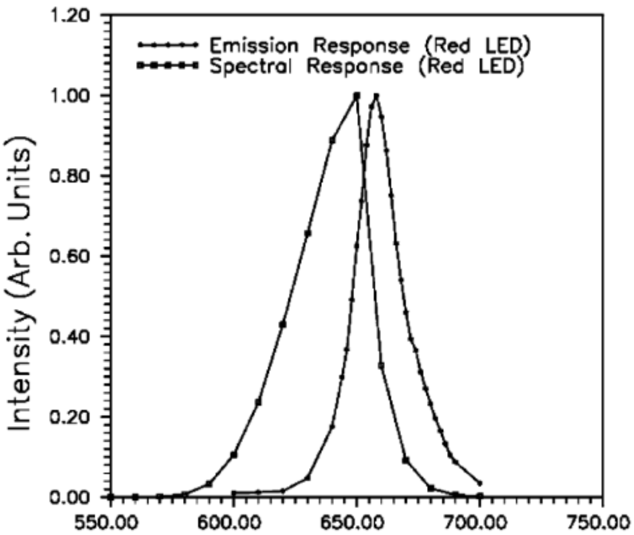

Figure 1 Emission and spectral sensitivity response for a typical LED

In relation to plants, loss of water through transpiration can be facilitated by the opening and closing of the stomata depending on environmental conditions. Leaf water deficits also arise where plant water loss is not compensated by root water uptake. Continuously monitoring of stomatal conductance can therefore assist in detecting early signs of water stress as the period when stomata pores are open will be reduced during water stress to decrease water loss through transpiration. Continuous measurements using a porometer are not usually practical in the field, thus this project will determine the viability of using remote sensing techniques to indirectly measure stomatal conductance.

This project will contribute to the research/development of a remotely sensed monitoring programme for minesite rehabilitation. This new programme needs to be finalised in advance of rehabilitation so the monitoring can commence concurrent with the rehabilitation activities.

This work also complements the vegetation component of KKN 2.6.1 in developing a spectral reference library to determining what seasonal stage gives maximum spectral separability detected between species.

Background

For a site like Ranger, it is important to continuously monitor plant growth in a cost and time efficient manner, thereby enabling the vegetation response to changing environmental conditions to be characterised throughout the year. A traditional remote sensing method for monitoring plant vigour and health involves using a field spectrometer to calculate Spectral Vegetation Indices (SVIs) such as the commonly used NDVI. These measurements can be taken in the field with an ASD spectrometer, but the advantage of a simple spectroradiometer is low cost, stability, and enablement of continuous measurement of the NDVI and other SVIs. Another advantage is that a number of devices can be deployed to monitor various species growing on different landscapes at the same time. These measurements will determine the health of the vegetation and the fraction of vegetation green cover within the Field of View (FOV) of the sensor. More importantly, the continuous measurements will enable investigation of diurnal and annual variations in vegetation structure and metabolism allowing early detection of water stress on the final rehabilitated minesite.

To complement the spectral measurements taken by the LEDs, an infra red temperature sensor will also be built into the device to monitor leaf temperature. It has been shown that leaf temperature is correlated to stomatal conductance and can therefore also be used to detect early signs of water stress (Hamlyn 2004). During plant transpiration water is lost and when a plant starts to experience water stress the period of transpiration during the day is reduced. Periods of transpiration can be measured using a porometer to measure stomatal conductance, but this can also be achieved by measuring the leaf temperature. Closure of stomatal pores in the leaf causes the temperature to increase and this can be detected with an infrared temperature sensor. This relationship is influenced by parameters such as soil moisture and vapour pressure deficit (VPD).

Aims

The knowledge gained from this project will contribute to assessing the success of mine site rehabilitation by detecting changes in ecosystem condition as they relate to mine closure criteria and will contribute to the research and development of a remotely sensed monitoring programme for minesite rehabilitation.

This research project aims to expand on preliminary investigations conducted on the Ranger TLF by measuring eco-physiological data from these *Eucalyptus* species in an attempt to establish specific responses to changing soil moisture and vapour pressure deficit (VPD) in response to seasonal climatic changes within a disturbed landscape.

It is hypothesised that while *E. tetrodonta* displays an ability to optimise available water resources, where both species co-occur, they are both at risk of dieback due to the poor physiological regulation of water use of *E. miniata* in drying conditions which will ultimately deplete the soil water reserve to dangerously low levels for their continued survival.

The aim of this research project is to establish the extent of these eco-physiological differences and whether these differences will potentially compromise the growth and survival of *E. tetrodonta* and *E. miniata* when planted in close proximity to one another. This research will also inform future revegetation strategies in regards to species composition and density on a rehabilitated landform in a tropical environment.

This project specifically aims to:

1. Determine spectral changes in both *E. miniata* and *E. tetrodonta* due to water stress using an ASD spectrometer.
2. Develop a low cost LED device for continuous monitoring of vegetation regrowth on the Ranger TLF.
3. Determine the relationship between stomatal conductance and leaf temperature. Specifically, how to remotely monitor *E. miniata* and *E. tetrodonta* stomatal conductance via plant/leaf temperature.
4. Deploy a number of LED devices to continuously monitor both *E. miniata* and *E. tetrodonta* on the trial landform for at least one year to enable investigation of diurnal and annual variations in vegetation structure and metabolism to contribute to the development of an eco-physiological model for revegetation on the final landform.

Methodology

The work programme of this research will include:

- determining the spectral changes in *E. miniata* and *E. tetrodonta* for selection of the most appropriate LEDs to detect water stress.
- measure plant/leaf temperature changes with an infrared temperature sensor while measuring stomatal conductance, soil moisture, and VPD.

Steps for determination of spectral characteristics of plants experiencing water stress

To determine the spectral bands required to be monitored using LEDs a number of *E. miniata* and *E. tetrodonta* will be setup in a controlled experiment at SSD Darwin where water supply is regulated and soil moisture content is continually being measured (Figure 2).

An ASD spectrometer will be used to measure the spectral reflectance of both the plant/leaf and will be analysed to determine more precisely the spectral bands of interest.

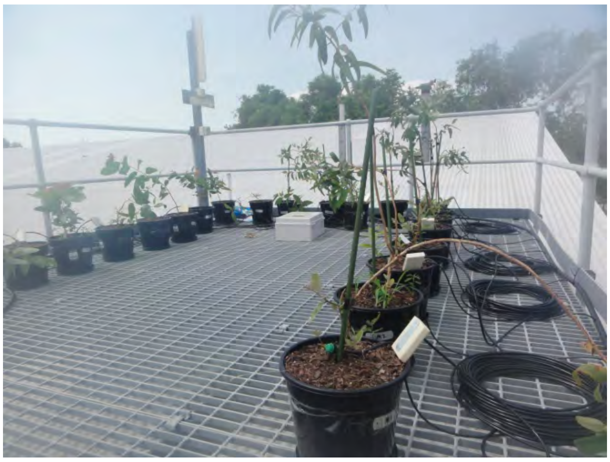

Figure 2 Setup of controlled experiment to detect water stress

Development of the LED continuous monitoring device

Once the spectral bands have been defined from published work and experimental testing, LEDs will be selected with peak emission wavelengths near the centre of the spectral bands of interest. This is done because LEDs are most sensitive to light near this wavelength.

Once all LEDs of various wavelength sensitivities have been selected, they need to be spectrally characterised using a tuneable light source (TLS) and a picoammeter to measure current. A TLS emits light in very narrow spectral band and by varying the light source across a range of wavelengths and measuring the current induced in the LED using a picoammeter the LED sensitivity curve can be constructed (Figure 3).

Once the LEDs have been spectrally characterized the next step is to interface the LEDs to a microcontroller unit (MCU) that records the signal from the LED when it is exposed to light. The MCU used in this study will be able to run continuously in the field with a rechargeable battery and a solar panel. The MCU performs both analogue and digital operations on a single chip with minimal peripheral circuitry. In essence, this means that MCU is able to both amplify the small current induced in the LED which is in the range of pico amps(analogue); and log recorded measurements including date and time (digital). These values are stored locally on a SD memory card. In addition the data is sent wirelessly to a base unit that transmits the information back to ***eriss*** via the mobile phone network. This will allow early notification of device failure and facilitate rapid corrective actions.

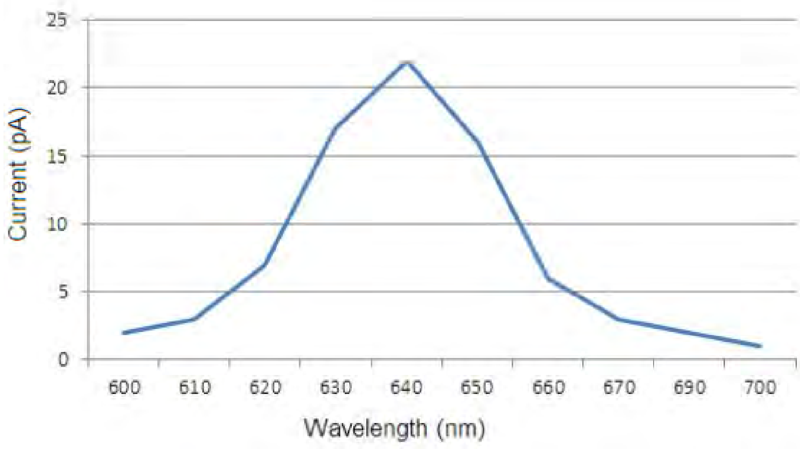

Figure 3 LED sensitivity curve

NDVI will be measured by selecting two LEDS that are sensitive in the near infrared and red bands. A value for NDVI can then be calculated from these sensors using equation [1] (refer introduction).

When taking measurements in daylight the conditions will vary and this needs to be taken into consideration when calculating the reflectance. Put simply the reflectance is calculated as:

$$\text{Reflectance} = \frac{\text{Energy leaving the sample by reflection}}{\text{Energy incident on sample}} \qquad (2)$$

Therefore to calculate a reflectance value a measurement must be taken of both the lighting conditions and the light reflected off the target (equation 2). This will be implemented in the design by having LEDs measuring solar radiation intensity while the plant reflected radiation is being measured (Figure 4). This is similar to taking a white reference with an ASD spectrometer.

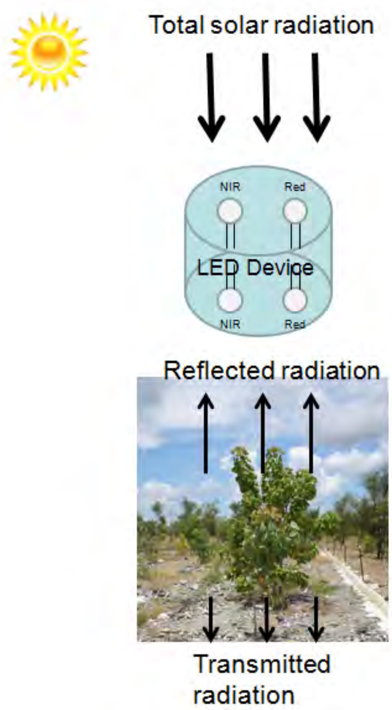

Figure 4 Illustration of how the LED device is able to measure both incident and reflected radiation

Initial testing of the performance of the LED spectroradiometer will be conducted at SSD prior to deploying in the field.

Determination of correlation of stomatal conductance changes with leaf temperature and effect of water stress on plant response

The same experimental setup as shown above (Figure 2) will be used. Measurements of plant/leaf temperature using an infrared temperature sensor will be taken hourly throughout the day while concurrently taking stomatal conductance measurements using a porometer (Li-cor 6400). Continuous measurements of VPD and soil moisture will also be collected.

Deployment on the trial landform.

Once constructed and tested the continuous monitoring devices will be deployed on the trial landform to monitor *E. miniata* and *E. tetrodonta* which are also being monitored for sap flow by Ping Lu (ERA). The continuous monitoring will continue for at least one year and be analysed regularly to assess diurnal and annual eco-physiological changes in *E. miniata* and *E. tetrodonta*.

Progress to date

Activity to date has focussed on the following:

- Procure and setup controlled experiment with E. tetrodonda and E. miniata in pots to assess the spectral characteristics to remotely determine water stress using the ASD spectrometer. This is being conducted at ***eriss***.
- Baseline spectral measurements have been collected.
- Construction of prototype device complete and communication network setup for telemetry of data.

Steps for completion

The completion of this project includes the following activities:

- finalise controlled experiments with *E. tetrodonda* and *E. miniata* in pots and analyse results (Q4 2013)
- construct 4 spectroradiometers for water stress testing and stomatal conductance measurements (Q4 2013)
- deploy monitoring devices on the TLF (Q1 2014)
- submit journal article detailing the construction of the LED device and methods of measurement (Q1 2014)
- submit second journal article describing the results measured in the field (Q1 2015)

In conjunction with ERA (Ping Lu) contribute to the development of a model for eco-physiological responses for *E. miniata* and *E. tetrodonta* growing on a rehabilitated landform in a tropical environment.

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Establishment of long term vegetation monitoring plots

T Whiteside, RE Bartolo, P Lu[1] & G Fox

Background

The project has arisen from Energy Resources of Australia Pty Ltd's (ERA) intention to implement a long-term fauna and flora monitoring programme on the Ranger Project Area (RPA) and, in agreement with Mirrar and Kakadu National Park (KNP) management, in the adjacent Kakadu National Park (KNP). The project's aims are to monitor and quantify the spatial and temporal variability of flora and fauna communities, thereby, a) providing essential data for the development of realistic closure criteria for the final rehabilitated landform and b) guiding future land and revegetation management on the RPA. In relation to item a), closure criteria, this project is the only targeted study that informs development of faunal communities. It complements development of closure criteria for vegetation, as reported under KKN 2.5.2, Vegetation analogues, in considering natural temporal changes and extrinsic (fire, weeds) factors, not considered in the former spatially-focused study – see below for primary objectives of the present study. It was intended that the monitoring programme be undertaken in close collaboration with ***eriss***. Initial phases of the project included site selection, development of the monitoring methodology, site establishment and initial monitoring.

The overall objective of ERA's monitoring programme is to establish baselines for the dynamics and variability that can be expected in the surrounding natural woodland ecosystem and the responses of these vegetation and faunal communities to natural disturbances such as fire or cyclones. This information assists development of closure criteria (see above) and is crucial to overall monitoring of revegetation success and management. The focus of this project is to monitor:

The impact of different fire regimes (low and high frequency burning) on woodland vegetation communities and fauna.

Through measurement of community structure, habitat complexity and landscape function over time, quantify natural variability due to seasonal (wet and dry season), interannual and broader global weather pattern changes (e.g. El Nino and La Nina).

The criteria for selecting vegetation communities deemed suitable for monitoring are:

- Woodland communities similar to those predicted to be established in the revegetation on the final landform (see KKN 2.5.2, Vegetation analogues).
- Natural/undisturbed status (e.g. no impact from mining or other activities, minimum presence of weeds).
- Under Ranger Mine land management or KNP land management.

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[1] Energy Resources of Australia Ltd, Darwin

While initially an ERA project, ***eriss*** is now the lead agency. The project will primarily inform the environmental monitoring key knowledge needs, KKN 2.6.1 and KKN 2.6.2. The project and the data collected will be integral to the project, Establishment of remote sensing method for monitoring the rehabilitated landform and off-site area listed under KKN 2.6.1, Monitoring of the rehabilitated landform.

Consequently, this project will be expanded to include monitoring sites that also represent the range of habitats on the RPA and off-site. The information from these monitoring sites will be used to link on ground biophysical properties to the spectral and spatial characteristics of remotely sensed data. The information will also be used as reference to verify analyses of the remotely sensed data used for on and off site monitoring.

Work to date

In October 2012, 11 monitoring sites were selected in the RPA. Field work was conducted during the week 3–7 June 2013 to select six monitoring sites in KNP. The purpose of the field work was to:

- determine whether identified sites were suitable as long term monitoring plots
- establish the plots
- collect and record preliminary biophysical data in the plots.

During the field work, data collected at the sites by the consultants from Eco Logical Australia Pty Ltd included site descriptions, species composition, percentage cover and height data for each strata, and vegetation classes and codes according to the Australian Soil and Land Survey Handbook (Hnatiuk et al. 2009). At each plot, ***eriss*** staff recorded plot location using a differential GNSS (Global Navigation Satellite System) receiver, the foliage projection cover (the percentage of a sample site occupied by the vertical projection of foliage only) and fractional ground cover.

Eco Logical Australia have produced a report for ERA, including the rationale for site selection and collating the field data.

Since conducting the field work, aerial photography over two plots was captured on 9^{th} October 2013, as part of the Unmanned Aerial System (UAS) demonstration project.

Future work

Due to the change in lead agency and revision of the focus, the project objectives and outcomes will be updated. This will include a determination of the temporal frequency at which sampling will be undertaken at the monitoring sites.

For each of the monitoring sites established in KNP in 2013, there will be comprehensive biophysical inventories undertaken in May 2014.

In addition, work will be undertaken to identify and establish additional sites in different vegetation communities, as well as data collection in these sites. This will ensure complete coverage of the primary vegetation communities identified in KKN 2.5.2, Vegetation analogues, study.

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Mapping of vegetation and annual variation for 2010–2013 using high spatial resolution multispectral satellite data

T Whiteside & RE Bartolo

Background

The Magela Creek floodplain is a down-stream receiving environment for the Ranger minesite. Off-site monitoring of this area will become increasingly important in the years following closure and rehabilitation of the minesite (post 2026), as part of an integrated environmental monitoring framework. Monitoring the success of mine site rehabilitation requires the assessment of numerous biophysical indicators within the landscape. The vegetation communities within the floodplain are spatially, seasonally and annually dynamic (Finlayson et al. 1989), but there has been little research of the drivers behind these dynamics. Such spatial and temporal variation exhibits superficial change in condition that masks longer term trends. Therefore, a robust methodology for mapping wetland vegetation at scales that can differentiate between variability and long term trends is required. Therefore, time series mapping of floodplain vegetation will provide a contemporary baseline of annual vegetation dynamics on the floodplain to assist with analysing change during and after rehabilitation. The study will also estimate the temporal scale (frequency) of mapping for monitoring the wetland during the rehabilitation and post-rehabilitation of the mine. High spatial resolution (HSR), multispectral, remotely sensed imagery (such as WorldView-2 data) provides spatially continuous data that is suitable for creating vegetation maps of the floodplain. This project involves the mapping of the vegetation communities within the Magela Creek floodplain for four consecutive years (2010–2013). The early detection of changes in long-term condition allows for early intervention to mitigate potential negative environmental impacts. Detection of change can also assist in targeting management activities and addressing inappropriate management methods. Due to the finer spatial scale of HSR data, there is an inherent increase in spectral heterogeneity within land covers. Consequently, the HSR imagery required data aggregation to assist with classification at a vegetation community level. This project used GEographic Object-Based Image Analysis (GEOBIA) (Hay & Castilla 2008) to partition and classify the data to map floodplain vegetation.

Method

WorldView-2 multispectral imagery (8 spectral bands) covering the study area was captured in the early dry season for 2010 (11 May), 2011 (14 May), 2012 (9 June) and 2013 (2 June) at the same time of day (±6 min). To enable accurate change detection, the 2010 imagery was geometrically orthorectified to sub-pixel accuracy (Whiteside et al. 2013) and the 2011, 2012 and 2013 imagery were registered to the 2010 base image. To correct for atmospheric effects and differing off-nadir view angles (including sensor orientation, satellite azimuth and elevation), radiometric calibration of the imagery was undertaken using the Fast Line-of-Sight Atmospheric Analysis of Spectral Hypercubes

(FLAASH) atmospheric correction algorithm (Matthew et al. 2003) to convert the at-sensor digital numbers to surface reflectance. The 2013 imagery has been partially analysed and is not reported on here butwill be presented future reports.

Images were classified using a geographic object-based image analysis (GEOBIA) approach, which involved segmenting the images into objects and the classifying these objects based on a series of class rules. Object-based methods have been shown to perform better with the increase in spatial heterogeneity associated with high resolution multi-spectral satellite imagery, compared with traditional per-pixel spectral based classification techniques (Blaschke 2010). The analysis was undertaken using eCognition® image analysis software package. Areas affected by cloud and cloud shadow (only visible in 2010) were eliminated from the image analysis and open water (visible in imagery from all years) was classified by the application of threshold masks based on the near infra-red (NIR) bands. The floodplain boundary was then delineated based on a 6 m threshold applied to a digital elevation model (DEM) of the region and analysis was confined within this boundary. Further image segmentation and classification was based on a series of multi-threshold segmentation algorithms. Objects representing floodplain vegetation community classes were created using a series of rules based upon four spectral indices (Table 1). A series of iterations involving segmentation, classification and reshaping were undertaken to create the class objects, whereby objects were iteratively partitioned into increasingly spectrally homogeneous objects and classified until an object's index threshold was met . This was achieved by analysing object values for the indices and establishing thresholds that achieved maximum separability between classes. Objects that were still spectrally variable were considered to contain two or more classes and were subsequently re-segmented and classified. The final classes derived from the index thresholds where then assigned to vegetation community classes based on expert knowledge and field data.

Table 1 Spectral indices used in this project.

Index	What the index is sensitive to
$NDVI = \frac{(NIR2 - Red)}{(NIR2 + Red)}$	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.
$EVI^{*} = \frac{G \times (NIR2 - Red)}{(NIR2 + (C1 \times Red) + (C2 \times Blue) + L}$	The Enhanced Vegetation Index (EVI) is strongly correlated to evapotranspiration. *Within EVI, G=2.5, C1=6, C2=7.5 and L=1
$FDI = NIR2 - (RE + Blue)$	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.
$LI = (NIR2 + RE) - Blue$	This Near infrared/Red edge/Blue index (LI) distinguished non-PS vegetation that is highly reflective; in this case communities dominated by *Nelumbo*, *Leersia*, and *Salvinia.*

For accuracy assessment purposes, reference data for each year was collected using field and aerial surveys coincident with the image capture dates. Due to accessibility issues and on ground/water safety, vegetation surveys over the floodplain were conducted using a helicopter and airboat for 2010, and helicopter for 2011–2013. Data collected included visual observations and photography, GPS-tagged digital photography (2011–2013) and GPS-tagged HD video (2012–2013). The vegetation classes determined at each sample site were compared to the classified imagery using confusion matrices (Congalton & Green 2009) and thematic accuracy measures calculated.

Results

Each of the final vegetation community maps (for 2010, 2011 and 2012) consisted of 11 vegetation classes labelled based on the dominant *Genera* for the community (Figure 1): *Eleocharis* sedgeland, *Hymenachne* grassland, *Leersia* grassland, *Melaleuca* open forest, *Melaleuca* woodland, *Nelumbo* herbland, *Oryza* grassland, para grass (*Urochloa mutica*), *Pseudoraphis* grassland, *Pseudoraphis*/ *Hymenachne* grassland and *Salvinia* floating mats. These class types are consistent with the classes identified and mapped previously and the overall accuracies (based on comparison with the reference data) are over 70% for each of the maps. Overall accuracy is calculated by dividing the total number of correctly identified sample sites for all classes by the total number of sample sites.

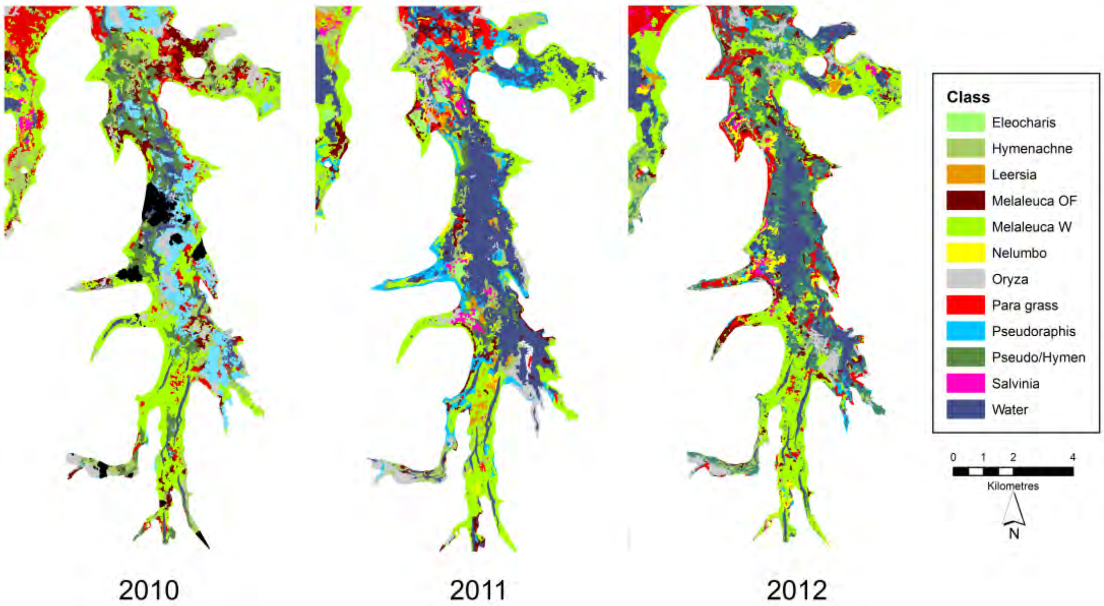

Figure 1 Vegetation maps for the southern portion of the Magela Creek floodplain for the early dry seasons (May-June) of 2010, 2011 and 2012. Black objects in the 2010 map are cloud or cloud shadow.

There is a measurable increase in open water from 2010 to 2011, and a reduction from 2011 to 2012 (Table 2). These changes in the area of open water correspond to annual rainfall records for the region and are coincident with reductions in area for the classes: '*Pseudoraphis*', '*Pseudoraphis/Hymenachne*', '*Melaleuca* woodland' and '*Melaleuca* open forest'. The area for para grass has also increased from 2010–2012 as patches have expanded.

There is a level of uncertainty associated with this analysis. While differences in reflectance can be used to analyse change in vegetation communities' structure and composition, changes in reflectance are also associated with the relative positions of sun and sensor, the sensor view angle, the amount of aerosols (e.g. water vapour and smoke) in the atmosphere, the extent of open water and plant phenology. Sun angle probably has minimal influence due to each year's imagery having been captured at the same time of day within a three week bracket. Sensor view angle appears to have more of an influence. There is also a visible haze in the 2011 image which may have affected the atmospheric correction algorithm. In addition, communities may not have changed in structure or composition, however, the plants within the community may be in different growth phases due to water availability thus displaying spectral differences.

Table 2 Total Area (ha) of CLASSES for Region displayed in Figure 1

Vegetation class	2010	2011	2012
Eleocharis	0.0	0.0	0.0
Hymenachne	557.0	518.8	348.4
Leersia	32.6	172.9	40.0
Melaleuca open forest	335.8	242.8	205.0
Melaleuca woodland	1560.3	1250.0	1345.3
Nelumbo	26.3	62.8	143.7
Oryza	395.7	285.2	287.6
Para grass	464.8	339.7	499.1
Pseudoraphis	661.9	450.4	117.1
Pseudoraphis/Hymenachne	626.5	150.3	1175.7
Salvinia	32.8	79.5	65.4
Open water	382.4	1674.4	1006.8

Preliminary results of the change analysis indicate that for large areas there was minimal community change; however there are areas that changed in species composition each year. Based on this the data set, much of this change is attributable to varying extent (and potentially depth) of open water, which is associated with inter-annual rainfall variability, while some change might be attributed to fire disturbance history.

Conclusions and future work

By applying the use of relative measures (such as ratios) as opposed to absolute (such as individual wavelength bands), it was anticipated the segmentation and classification process would be transferrable from one year to the next. On the whole , the described methodology was transferrable with only minor threshold adjustments required. Comparison with reference data indicate the rule set was able to distinguish the majority of floodplain classes. This method is currently being applied to mapping the floodplain vegetation using the imagery acquired for June 2013. The complete 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. Once vegetation maps are available for all dates, an object-based change detection algorithm will be used to identify objects that had changed classes between dates. Objects can then be identified as either unchanging or dynamic. Further analysis of the dynamic objects will be undertaken to distinguish between objects displaying actual change (ie a change in class associated with a change in community composition) and change due to either geometric mis-registration between the images, spectral differences associated with differing viewing geometries or phenology, or the presence or absence of water. From the analyses, three change maps will be created: 2010–2011, 2011–2012 and 2012–2013. These maps include objects belonging to consistent classes through the dates and objects where classes had changed (from-to classes) and will be shown in subsequent Annual

Reports as well as published in peer-reviewed literature. This change could then be matched to external variables such as annual and seasonal rainfall variation, stream flow variability and fire scar mapping for the time frame.

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2.7 Risk assessment

Stressor pathway conceptual models for the operational, rehabilitation and post-rehabilitation phases for Ranger uranium mine (collaborative project with ERA)

RE Bartolo, S Paulka, R van Dam, S Iles & AJ Harford

Background

Conceptual models of potential stressor pathways associated with mining at Ranger uranium mine have been developed as part of the evolving ecological risk assessment framework that was started by the Supervising Scientist in the early 1980s. Stressor pathway conceptual models for the operational phase of mining were revisited during the mid to late 2000s, and were subsequently completed during 2012–13 (Bartolo et al. 2013). Also during 2012–13, an ecological risk assessment commenced for the rehabilitation and post-rehabilitation phases of Ranger mine (referred to herein as the rehabilitation risk assessment). The rehabilitation risk assessment is a collaboration between ERA, SSD and other key stakeholders, and represents a key activity under the ARRTC Key Knowledge Needs (KKN 2.7.1 – *Ecological risk assessments of the rehabilitation and post rehabilitation phases*). The main activity during 2012–13 was the development of contaminant cause-effect conceptual models for the various key values and phases of closure. This paper provides an overview of the completion of the operational phase conceptual models and the initial development of the rehabilitation conceptual models.

Operational phase conceptual models

The models and associated narrative for the operational phase conceptual models have been completed and compiled in an Internal Report (Bartolo et al. 2013). A total of 31 stressor-pathway models were identified and developed, with a screening analysis indicating that five of the stressor-pathway combinations were of high importance. An example of a completed stressor-pathway conceptual model is shown in Figure 1, for the transport of inorganic toxicants from on-site water bodies to off-site water bodies. These models have been used to identify knowledge gaps and are also informing the recently commenced rehabilitation risk assessment (see below).

Ecological risk assessment for rehabilitation and closure

ERA is required to rehabilitate Ranger uranium mine by January 2026 and, thus, a large number of research and assessment projects are underway by both ERA and SSD to ensure the necessary knowledge is available to inform the rehabilitation and closure strategy. The rehabilitation risk assessment provides a structured and comprehensive framework for confirming that all the key issues related to ensuring the protection of the off-site environment and successful rehabilitation of the on-site environment are identified.

The rehabilitation risk assessment has been separated into two distinct phases in the first instance:

1. problem formulation
2. risk analysis.

The problem formulation phase was largely undertaken in 2012–13, through a stakeholder workshop aimed at developing causal conceptual models for the key stressors and values that can be used in the subsequent phases of the risk assessment. The workshop was facilitated by CSIRO. The temporal and spatial scope (see Figure 2) of the risk assessment was constrained as follows:

Temporal scales

- decommissioning – now to 2026
- stabilisation – decadal: 30–50 years; considering incorporation into KNP
- long-term post-closure monitoring – shorter term, ~300 years, and longer term, 10,000 years, as per the requirement for tailings to be contained for at least this long.

Spatial scales

- the Ranger Mine site (disturbed footprint)
- the Ranger Project Area
- Magela sub-catchment
- Kakadu National Park and the Alligator Rivers Region.

Conceptual models (causal maps) of potential stressors and their pathways have been developed as part of the problem formulation phase of the ecological risk assessment focused on closure and rehabilitation of Ranger Uranium Mine. The conceptual models were drafted during a two day workshop with stakeholders who formed breakout groups around four themes (aquatic ecosystems; terrestrial ecosystems (Ranger Project Area); terrestrial ecosystems (landscape); and people) and were subsequently reviewed by a small review group and finalised by the breakout groups.

Components of the problem formulation that were undertaken include:

- Identify the key sources, stressors and ecological assets that will be examined for the decommissioning, stabilisation and monitoring, and the post-closure phases of Ranger uranium mine's closure. The values are documented in Pollino et al. (2013).
- Determine the ecological assessment endpoints. Numerous assessment endpoints were developed by the four breakout groups as shown in Table 1. The biodiversity assessment endpoints for on-site and off-site aquatic ecosystems are proposed and not finalised, as the ecological processes and functions that were identified during the workshop have not been drafted as conceptual models. The ecological processes and functions will be addressed in the near future by the Closure Criteria Working Group (CCWG).
- Develop conceptual models for the above-mentioned closure phases of the Ranger mine site.

- The aquatic ecosystems breakout group drafted one large conceptual model that included all sources, stressors, pathways, measurement points and assessment endpoints during the workshop. This large model needed to be reduced into ten sub-models in order to define differences in the pathways to measurement endpoints for each of the stressors.
- The terrestrial ecosystem (RPA) breakout group drafted four conceptual models during the workshop, with three being reviewed post workshop. The fourth model focused on 'protection of human health' and has not been revised at this stage. The human health model will be dealt with separately in the future, combining it with the human health model drafted by the people group. An example conceptual causal model is shown in Figure 3, for the risks to landform stability on the Ranger mine site.
- The terrestrial ecosystem (landscape) breakout group drafted three conceptual models during the workshop. These have been revised to two conceptual models as the sources and stressors were similar between two of the models that were output from the workshop, and review of the assessment endpoints indicated that the models were quite similar.
- The people breakout group drafted two conceptual models – one for cultural landscape and the other for human health. Currently, the group has focused on refining the cultural landscape model. The human health model will be dealt with separately in the future as mentioned above. The original cultural landscape model, drafted during the workshop, has been split into four separate conceptual models, to reflect the revised assessment endpoints.

- Communicate and document the outcomes from the problem formulation phase.
  - A report was produced by Pollino et al. (2013) which details background material, and the values and draft conceptual models produced during the workshop.
  - Produce an Internal Report that includes the draft conceptual models.
  - Revision of the conceptual models and communication is ongoing.

Further work

Once the causal models are finalised, they will be used as the basis for scoping the risk analysis phase of the risk assessment. Part of the problem formulation phase project included the development of an analysis plan (design, data needs, and methods for undertaking the risk analysis phase of the assessment), which was undertaken and reported by Pollino et al. (2013). It is recommended that the AS/NZS ISO 31000:2009 (ISO 2009) generic framework for risk management be adopted as it is considered best practice. The US EPA ecological risk assessment guidelines (US EPA 1998) can be used in conjunction with the ISO risk management standard. Pollino et al. (2013) recognise there are many approaches for undertaking risk analysis, but they focused on Bayesian networks, a recommended approach in the ISO/IEC 31010 Standard which supports the risk management standard (it is focused on risk analysis techniques). Bayesian networks are probabilistic graphical networks, which can be used to directly apply the conceptual model from the problem formulation in a modelling platform for quantifying risks and associated uncertainties.

The risk analysis phase of the rehabilitation risk assessment will commence in 2013–14. Progress to date has been impeded due to staff availability (resourcing) of the project both within SSD and ERA. This has impacted on the timeline to get the Internal Report finalised and in commencing the risk analysis phase.

Stressor pathway conceptual models for the operational, rehabilitation and post-rehabilitation phases for Ranger uranium mine (collaborative project with ERA)
(RE Bartolo, S Paulka, R van Dam, S Iles & AJ Harford)

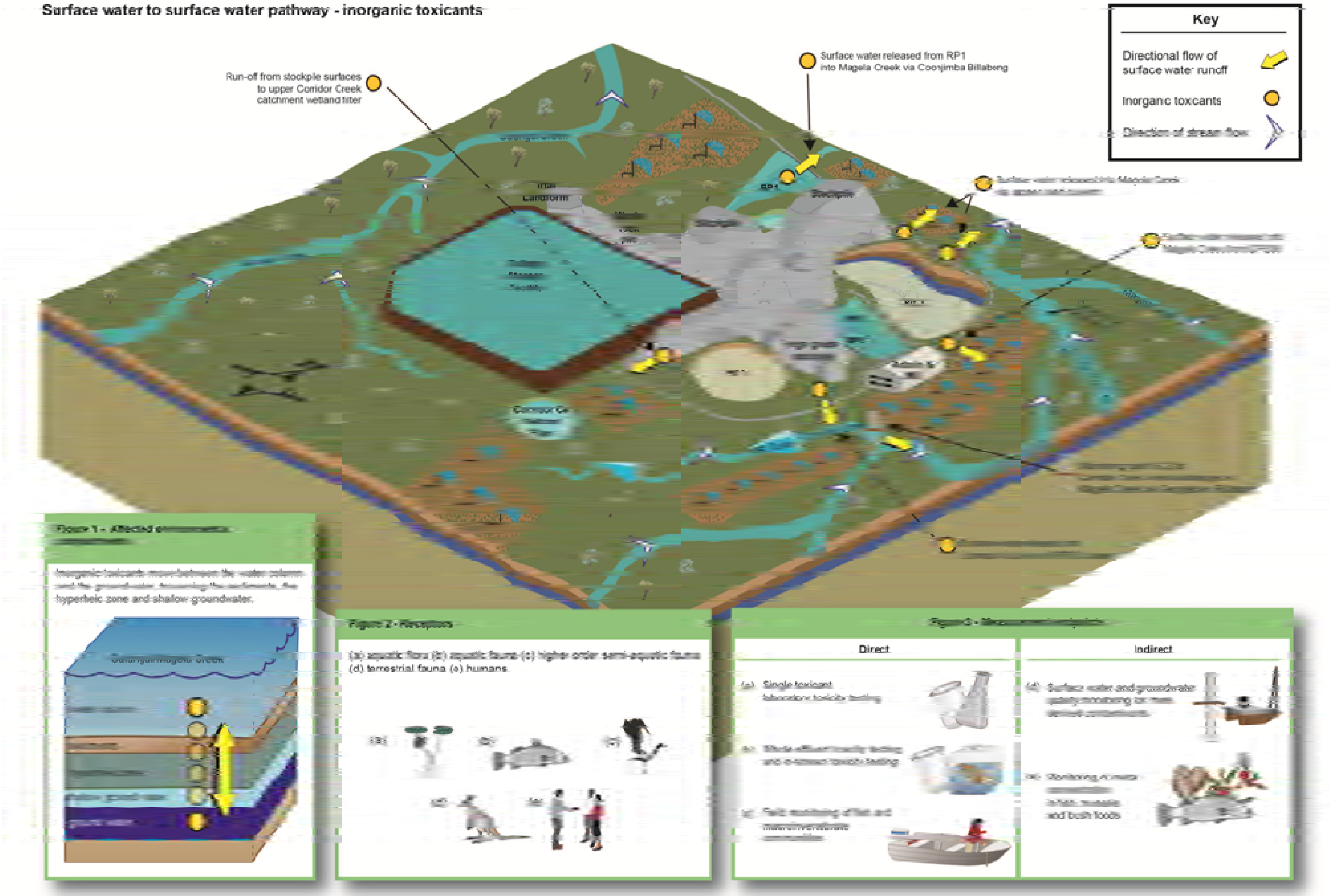

Figure 1 Conceptual model for the transport of inorganic toxicants from on-site surface water bodies to off-site surface water bodies.

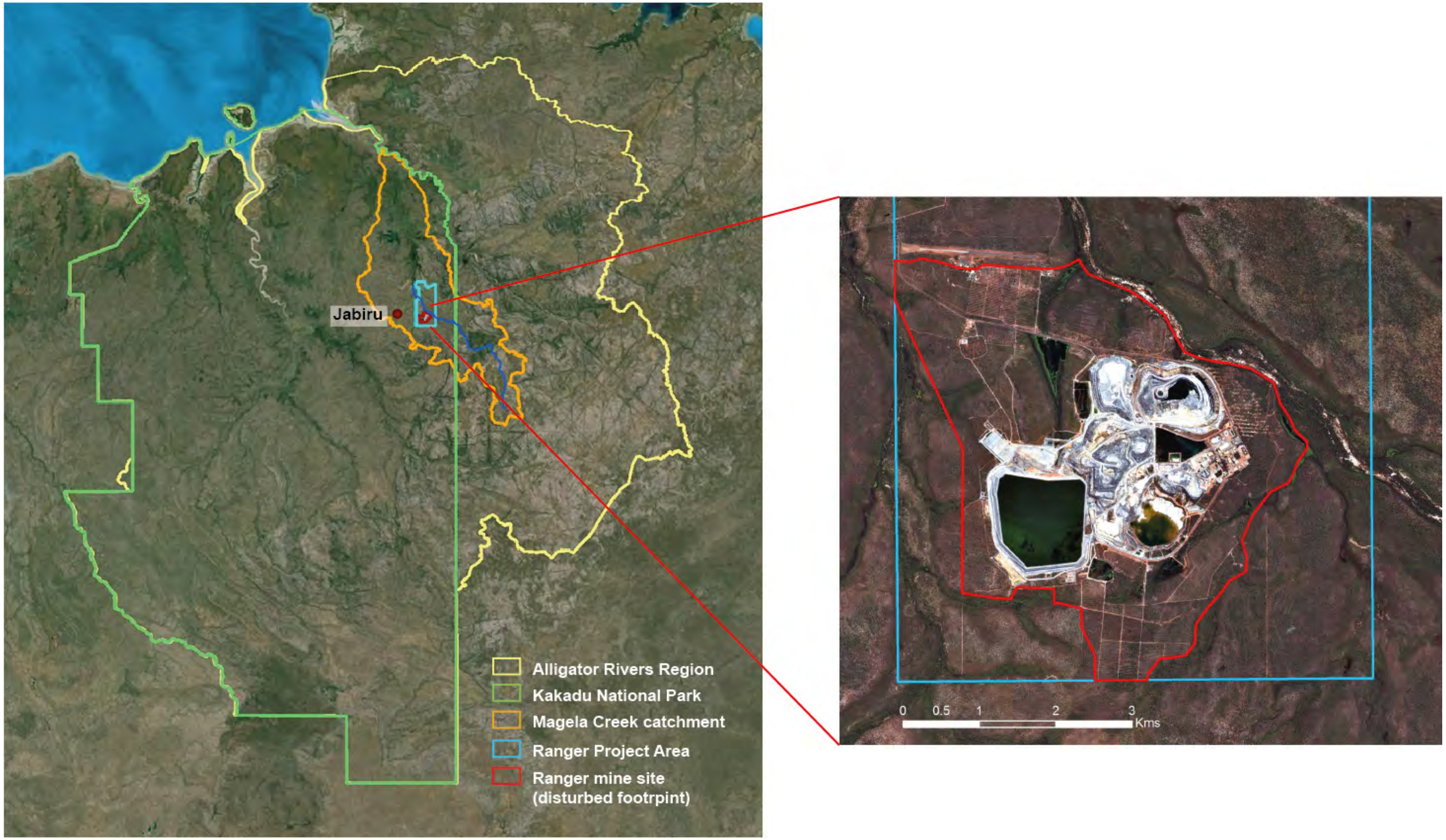

Figure 2 Spatial scales defined for the ecological risk assessment

Table 1 Draft assessment endpoints for the Ranger rehabilitation risk assessment

High level value	Assessment endpoints
Protection of off-site aquatic ecosystems	Off-site water quality meets agreed closure criteria specified for water quality
	Biodiversity (structure and function) of off-site aquatic ecosystems are comparable to the agreed reference condition
	Habitat diversity of off-site aquatic ecosystems is comparable to the agreed reference condition
Restoration/rehabilitation of on-site aquatic ecosystems	On-site water quality is on a trajectory towards meeting agreed closure criteria specified for water quality on-site
	Biodiversity (structure and function) of on-site aquatic ecosystems are on a trajectory towards meeting agreed closure criteria
	Habitat diversity of on-site aquatic ecosystems is on a trajectory towards meeting agreed closure criteria
Protection of off-site terrestrial ecosystems (Landscape)	Habitat diversity and ecosystem functions within the landscape of the Magela sub-catchment and broader KNP are comparable to an agreed reference condition
	Aesthetic values meet the expectations of the stakeholders in the ARR
Restoration/rehabilitation of on-site terrestrial ecosystems (Ranger Project Area)	Erosion characteristics of the rehabilitated landform meet agreed closure criteria
	Vegetation on the disturbed sites of the RPA is on a trajectory towards meeting agreed closure criteria
	Wildlife on the rehabilitated site is on a trajectory towards meeting agreed closure criteria
Re-creation of a cultural landscape in which traditional owners can resume traditional practices	Landform is able to be accessed, and is traversable, by people
	Presence of cultural important species at right time and abundance
	Landform, vegetation and water bodies on-site meet agreed cultural closure criteria
	Return of traditional practices (e.g. burning, harvesting)
Protection of human health on the rehabilitated site	Radiation doses to people from the rehabilitated site are less than the dose limits
	Rehabilitation works do not negatively impact on worker safety
Protection of human health off the rehabilitated site	Water resources used for drinking continue to meet drinking water limits for mine-derived contaminants
	Water resources used for recreation continue to meet recreational water quality limits for mine-derived contaminants
	Radiation doses to people from the rehabilitated site are less than the dose limits

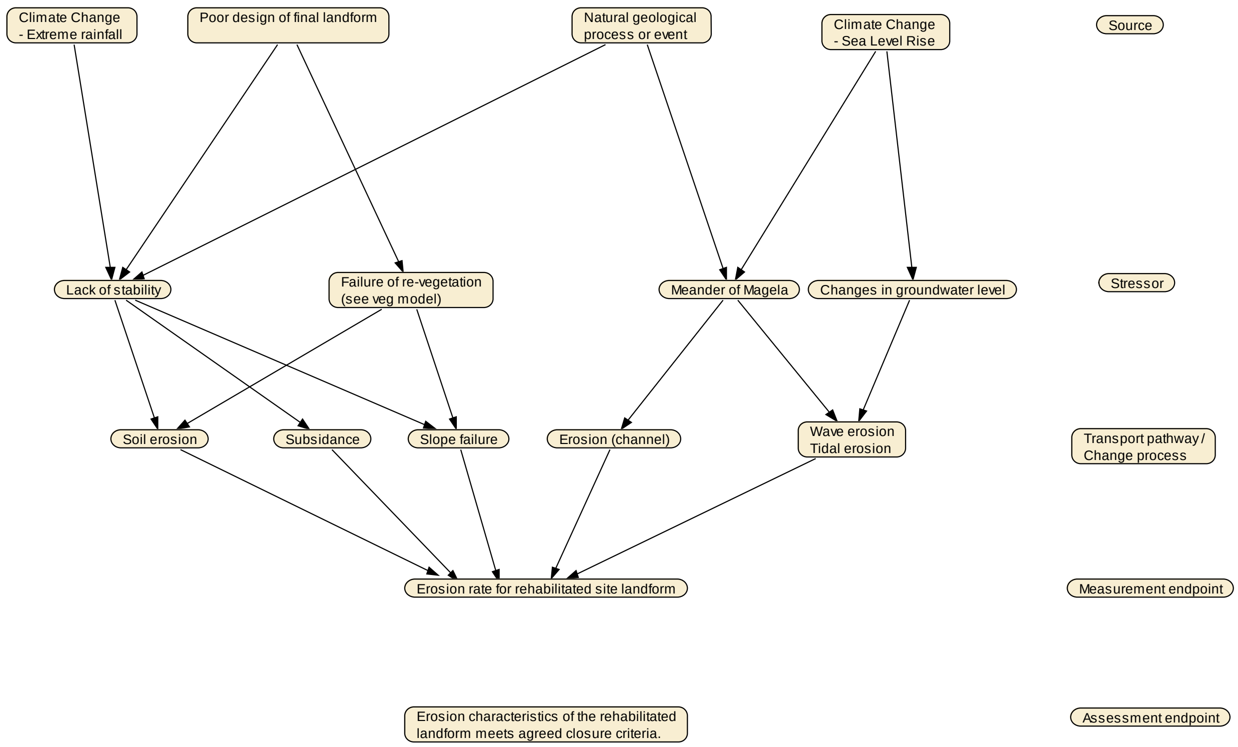

Figure 3 Draft conceptual causal model for risks to landform stability on the Ranger mine site

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Part 3: Jabiluka

Surface water quality monitoring

In accordance with the Jabiluka Authorisation, ERA is required to monitor a range of surface and ground waters on the lease and to demonstrate that the environment remains protected. Specific water quality objectives (criteria thresholds were described in Supervising Scientist Annual Report 2003–04) must be met. Each month during the wet season, ERA reports the water quality in Swift Creek (Ngarradj) to the major stakeholders (SSD, DME and NLC). A detailed interpretation of water quality across the site is provided at the end of each wet season in the ERA Jabiluka annual wet season report (WSR).

In addition to the ERA programme, the Supervising Scientist conducts monitoring in Swift Creek (Ngarradj). Jabiluka has been in a long-term care and maintenance phase since late 2003 and poses a low risk to the environment. As a consequence of this low risk and the good data set acquired indicating the environment has been protected, the monitoring programme has been systematically scaled down.

The SSD biological monitoring programme for Jabiluka ceased in 2004, commensurate with the low risk posed while the site is in long-term care and maintenance mode. Results from six years (1999–2004) of fish community structure studies were reported in Supervising Scientist Annual Report 2003–04 along with results for macroinvertebrate community structures.

Since 2009–10, SSD has collected continuous monitoring data (electrical conductivity and water level) from the downstream statutory compliance site only. ERA collects monthly grab samples from both the upstream and downstream site. Previous grab sample monitoring data can be found on the SSD website at www.environment.gov.au/ssd/monitoring/ngarradj-chem.html and have been reported in previous annual reports.

Chemical and physical monitoring of Swift Creek (Ngarradj)

A Sinclair

Flow was first recorded at the Swift Creek (Ngarradj) monitoring station on 17 January 2013. Continued low rainfall conditions resulted in low flow levels in the creek through most of February. March experienced an increase in the frequency of rainfall events, which increased the flow levels in Swift Creek (Ngarradj) with a corresponding decrease in EC (Figure 1).

A significant rainfall event occurred on 30 and 31 March 2013 with Jabiru Airport recording a total rainfall of 240 mm over the two days. Flow within Ngarradj quickly rose with a corresponding decline in EC. EC subsequently increased through April as flow levels decreased with typical recessional flow conditions becoming established in May and June.

Continuous monitoring continued until 12 June 2013, when the multi-probes were no longer submersed and could not be lowered any further. Cease to flow in Ngarradj was agreed by stakeholders on 19 June 2013.

During the season there were intermittent communications issues between the monitoring station and the server. Data was able to be manually downloaded during site visits with the SSD web page updated as data became available. A modified satellite antenna system has been installed and has rectified the communications issues.

Overall, the water quality measured in Swift Creek (Ngarradj) for the 2012–13 wet season is comparable with previous wet seasons (Figure 2), and does not appear to show a detectable effect from the discharge of water from the Interim Water Management Pond (IWMP).

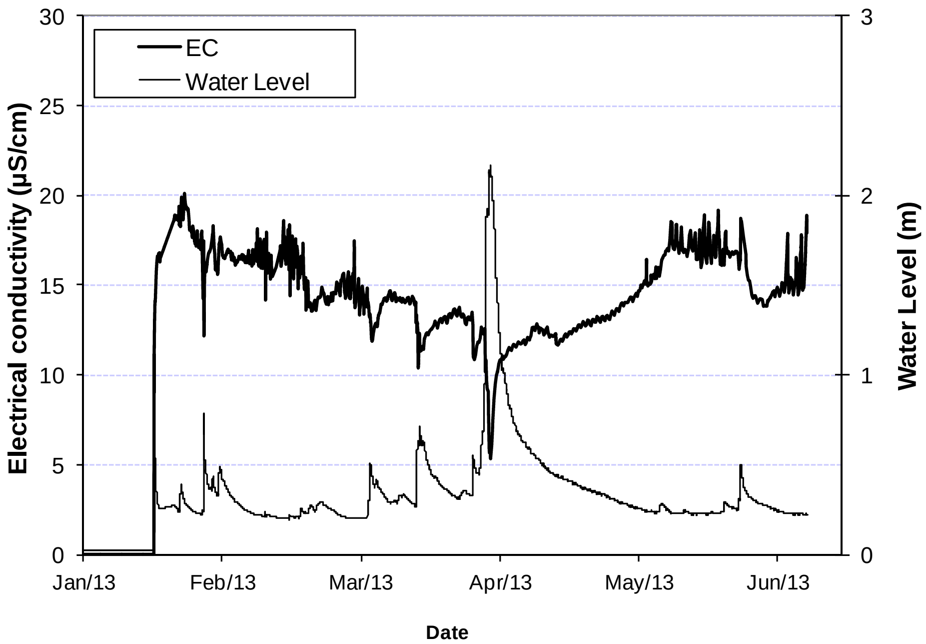

Figure 1 Continuous electrical conductivity in Ngarradj between January and June 2013

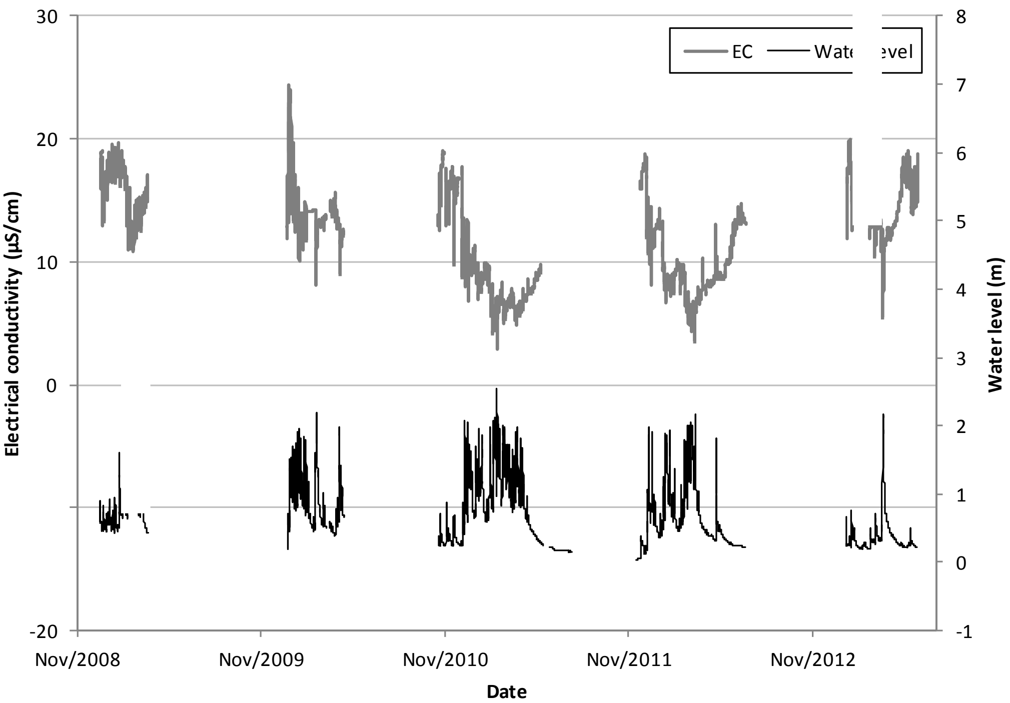

Figure 2 Electrical conductivity measurements at the downstream monitoring station in Swift Screek (Ngarradj) for each wet season between November 2008 and June 2013

Radiological exposure monitoring

Radiological exposure of employees

The Jabiluka Authorisation was revised in July 2003 and the statutory requirement of quarterly reporting of radiological monitoring data for Jabiluka was removed. The current Authorisation requires reporting of radiation monitoring data only if any ground disturbing activities involving radioactive mineralisation occur on site. No ground disturbing activities took place during this reporting period.

Radiological exposure of the public

Although there were no activities reported at the Jabiluka Mineral Lease, the population group that may, in theory, receive a radiation dose due to future activities at Jabiluka is a small community approximately 10 km south of Jabiluka at Mudginberri.

SSD has a permanent atmospheric monitoring station at Four Gates Road radon station, which is located a few kilometres west of Mudginberri. Radon progeny and dust-bound LLAA radionuclide concentrations are measured at the station.

Figures 2 and 5 in the *Atmospheric radioactivity monitoring in the vicinity of the Ranger mine* paper in the KKN 1.3.1 section of this report show radon progeny PAEC and dust-bound LLAA radionuclide concentrations measured in air at Four Gates Road radon station during 2012. Tables 1 and 2 in the same section provide public dose estimates for these exposure pathways for a person living at Mudginberri in 2012.

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 plant have meant that the involvement of SSD has been reduced following completion of the suite of projects that had been initated to define for stakeholders the rehabilitation status of the site.

Part 5: General Alligator Rivers Region

Radiological monitoring and assessment at the El Sherana airstrip containment

A Bollhöfer, C Doering, P Medley & L Da Costa

Introduction

The El Sherana airstrip containment is a near-surface disposal facility located in the South Alligator River Valley in the south of Kakadu National Park. It contains approximately 22 000 m^3 of contaminated waste from the remediation of legacy uranium mining sites in the area. Background on the history and remediation of legacy uranium mining sites in the South Alligator River Valley is provided in Waggitt (2004) and the Supervising Scientist 2008–09 Annual Report (Supervising Scientist 2009).

The containment was constructed, filled and covered in the 2009 dry season (O'Kane 2012). It is currently in the institutional control period. This is the period following closure of the facility during which public access to, or alternative use of, the site must be restricted (NHMRC 1993). A boundary fence and warning signs are in place at the containment to restrict access and deter against alternative uses of the site, including camping. Parks Australia has primary responsibility for the containment and is licensed and regulated by ARPANSA for its management of the site.

eriss conducts periodic radiological monitoring at the containment (Doering et al. 2011, Bollhöfer et al. 2013). The purpose is to assess its performance through time, including whether site radiological conditions are stable. A summary of the monitoring results is presented here together with an assessment of expected maximum annual doses for occupational and public exposure scenarios (Bollhöfer et al. 2013).

Methods

eriss has made dry season measurements of external gamma dose rates and radon (^{222}Rn) activity flux densities at the containment during the following years:

- 2007 (gamma dose rates) and 2009 (radon exhalation): baseline conditions
- 2010: one year after closure (Doering et al. 2011)
- 2012: three years after closure (Bollhöfer et al. 2013)
- 2013: four years after closure.

External gamma dose rates were measured across the containment within the fenced area, using environmental dose rate meters. The above background component of the measured gamma dose rate after containment construction was determined by subtracting the mean 2007 baseline gamma dose rate from the gamma dose rates measured in 2010, 2012 and 2013.

Radon activity flux densities were measured using brass canisters filled with activated charcoal, which were deployed on the containment for around three days to entrap radon exhaled from the soil surface. The canisters were then collected and sealed for radioactivity analysis. Radon trapped on the charcoal decays and the activity of radon decay products in the canisters is measured using a sodium iodide detector. The radioactivity measurement coupled with the length of the deployment and measurement periods enabled radon activity flux densities from the soil surface to be determined. The above baseline radon activity flux densities were determined by subtracting the baseline from the radon activity flux densities measured in 2010, 2012 and 2013.

Soil samples from the top 5 cm were also collected in 2012 from directly underneath the canisters deployed on the containment above the buried waste. This was done to determine the soil ^{226}Ra activity concentration, using the ***eriss*** high purity germanium detectors, and establish a relationship with radon activity flux density (Lawrence et al. 2009).

Results

Figure 1 shows the location and magnitude of the 2007 baseline, 2012 and 2013 gamma dose rate measurements, and the 2010, 2012 and 2013 radon activity flux density measurements at the containment. The results are overlaid on an aerial photograph of the area from March 2007. The outer white rectangle indicates the approximate location of the boundary fence around the containment. The inner rectangle shows the approximate location of the containment and buried waste.

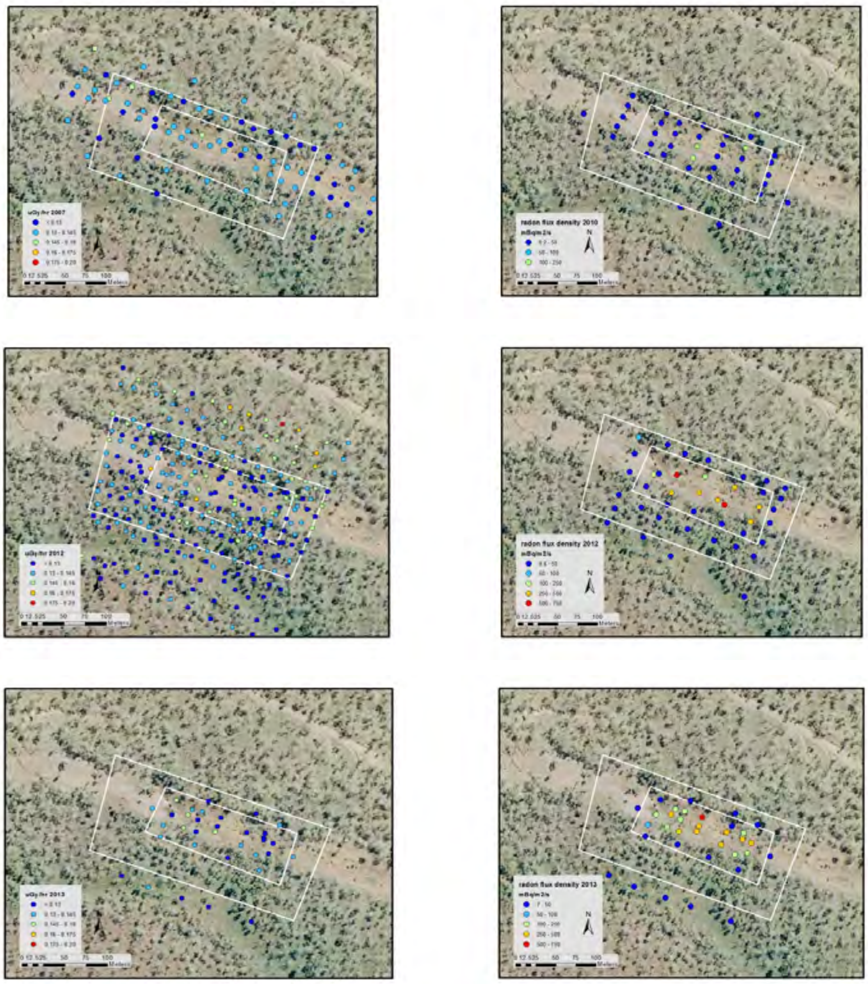

Figure 1 Baseline, 2012 and 2013 external gamma dose rates [µGy·hr-1], and radon activity flux densities [mBq·m-2·s-1] measured at the containment in 2010, 2012 and 2013

The baseline gamma dose rate in 2007 was no different to the arithmetic means measured within the fence in 2012 and 2013 (0.13 ± 0.01 μGy·hr^{-1}). Thus the above baseline gamma dose rate three and four years after containment construction was zero. In contrast the geometric mean radon flux density on top of the buried waste (inner rectangle in Figure 1) increased from 20 mBq·m^{-2}·s^{-1} in 2010 to about 120 mBq·m^{-2}·s^{-1} in 2012, but no further increase was observed in 2013. It is important to note that in June 2013 vegetation in the middle of the containment was cleared and approximately 1 metre of clean soil placed on top of the existing capping to re-contour the surface and prevent erosion of the capping material at the containment. This additional layer does not appear to have lowered overall radon activity fluxes from the containment measured in October 2013.

^{226}Ra activity concentrations in the top 5 cm of soils at the El Sherana containment were low (mean: 23 ± 6 Bq·kg^{-1}) and the ratio of radon activity flux to ^{226}Ra activity concentration at the containment exceeded 10 mBq·m^{-2}·s^{-1} per Bq·kg^{-1} in places. This is about one order of magnitude higher than expected for natural soils in the Alligator Rivers Region (Lawrence et al. 2009). The radon diffusion length in natural soils is typically about 1.5 m (Porstendörfer 1994) and the increase in radon activity flux is thus likely originating from deeper layers in the soil, due to changes in the physical properties of the containment cover, caused by roots penetrating into the surface soils or drying and cracking of the compacted clay layer immediately on top of the buried waste.

Dose assessment

Using the 2012 monitoring results, an assessment of expected maximum annual doses has been made for the normal (passive) operation of the containment for occupational and public exposure scenarios. No breach of the surface cover of the containment, either intentionally or by natural processes, has been assumed in the assessment of each scenario. Details of the assessment methodology is given in Bollhöfer et al. (2013).

Exposure scenarios

Occupational

It was assumed that a Park Ranger spends 80 hours per year working at the containment. Work conducted at the site was assumed to include general maintenance, weed and fire management and downloading in-situ monitoring equipment. Digging into the buried waste or repair of the capping layer or surface cover was not considered in the assessment, as this was not considered to be part of the normal (passive) operation of the containment.

Public (tourist camping)

It was assumed that a tourist camped for four nights (40 hours in total) next to the boundary fence of the containment. The camping location was assumed to be immediately downwind of the containment to give the highest possible exposure scenario. No inadvertent or intentional intrusion of the fence was assumed.

Public (Aboriginal person)

An exposure scenario for Aboriginal people was not assessed, as anecdotal evidence suggested that they do not spend time in the immediate vicinity of the containment, but rather prefer other sites in the South Alligator Valley for hunting, fishing and camping. The likelihood of exposure of an Aboriginal person was considered negligible.

Exposure pathways

The exposure pathways included in the occupational scenario were radon progeny inhalation and external gamma radiation. The only pathway included in the public (tourist camping) scenario was radon progeny inhalation, as external gamma radiation from the containment will be negligible outside the boundary containment fence.

The dust inhalation pathway was not considered in either scenario as the contaminated waste was assumed to be permanently buried under a clay capping layer and clean soil cover, with no breach of the capping layer. Any radionuclides in dust from the soil cover were considered part of the natural background.

The ingestion pathway was not included as bushfoods and locally sourced water were not considered to be consumed in either the occupational or public exposure scenario. Additionally, it is unlikely that any terrestrial plant-based bushfood would be collected onsite by the public, nor is it likely that terrestrial animals (such as wallaby, pig or buffalo) that are sometimes consumed by Aboriginal people would be able to access the site and take up radionuclides from the buried waste due to the presence of the boundary fence during the institutional control period. Groundwater levels are deeper than the buried waste and flow of groundwater from the containment to the South Alligator River is slow, with hydraulic conductivities less than 10^{-7} $m{\cdot}s^{-1}$ (Puhalovich et al. 2006). Consequently, the ingestion pathway for aquatic bushfoods and water is negligible, at least during the institutional control period.

Assessment method

The maximum dose from external gamma radiation was calculated from the 99th percentile of the above baseline dose rate (0.03 $\mu Gy{\cdot}hr^{-1}$) and the time spent onsite. The maximum dose from radon progeny inhalation was calculated using the RESRAD-Offsite computer model (Yu et al. 2009), with the 99th percentile of the above baseline radon activity flux density (900 mBq m^{-2} s^{-1}; Bollhöfer et al. (2013)) to determine the radon and radon progeny concentration in air on and downwind of the containment for highly stable atmospheric conditions. Figure 2 shows the containment origin ^{222}Rn concentration in air with increasing distance from the fenceline.

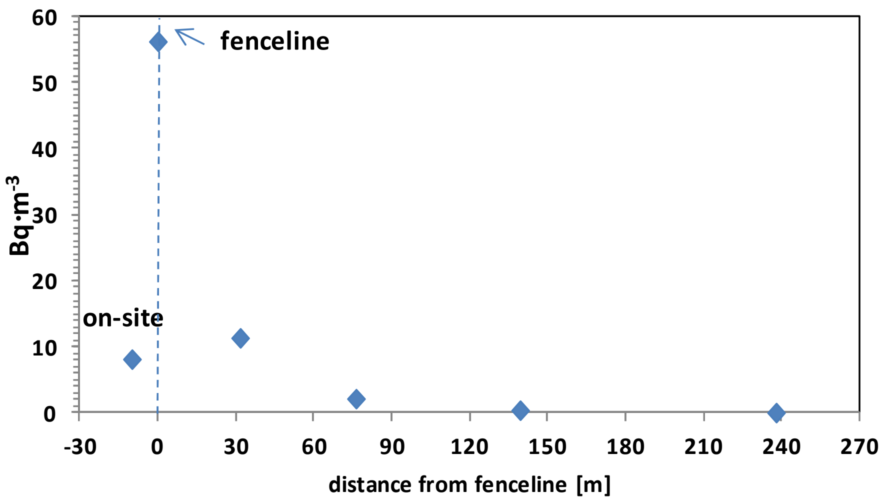

Figure 2 Containment origin ^{222}Rn concentration in air with distance from the fenceline for stable atmospheric conditions.

Assessment results

Table 1 gives the expected maximum doses to a park ranger and a member of the public from the containment for the assumed exposure scenarios.

Table 1 Expected maximum above background annual effective doses (µsv) to a worker and the public from the containment

	Gamma	Inhalation		Ingestion	Total
		Radon progeny	Dust		
Park Ranger	3	1	0	0	4
Public (tourist camping)	0	6	0	0	6

The results indicate that in both cases the expected maximum dose from all pathways is less than 10 µSv per year for the current radiological characteristics of the containment. The tourist camping received the higher dose due to radon progeny as it was assumed the park ranger was in the centre of the containment and thereby only exposed to radon expressed from the upwind section, whereas the camper was assumed downwind of the entire containment footprint.

Conclusions and future work

Our assessment of the maximum above background annual effective doses showed that doses are less than 10 µSv per year for the current radiological characteristics of the containment. This is less than 1 per cent of the public dose limit and of no concern to people working on site or camping in its vicinity. There is some evidence to suggest that radiological site characteristics have changed between 2010 and 2012, allowing more radon to diffuse through and exhale from beneath the containment cover. Further increases in radon exhalation from the site would imply higher maximum expected doses, but radon activity flux densities measured in 2013 are comparable with 2012 results. SSD will continue to monitor the radon activity flux density from the site, to assess whether it increases further or stabilises to a value similar to that measured in 2012 and 2013.

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Appendix 1 SSD publications and presentations for 2012–2013

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Appendix 2
List of inactive or delayed projects for which summaries are not provided in this report

KKN / Project	Comments
KKN 1.2.2 land Irrigation	
Characterisation of contamination at land application areas at Ranger (collaborative project with ERA)	Project completed and reports being reviewed and published
KKN 1.2.4 Ecotoxicology	
The direct effects of suspended sediment on tropical freshwater biota	Project suspended due to higher priorities and resource constraints.
KKN 2.6.2 Off-site monitoring during and following rehabilitation	
Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on Gulungul Creek catchment, Ranger mine site and Nabarlek area.	Project suspended due to higher priorities and resource constraints.
KKN 5.1.1 Develop a landscape-scale ecological risk assessment framework for the Magela catchment	
Use of LiDAR and other Digital Elevation Data to develop a landscape model of the Magela catchment	Project suspended due to resource constraints.

Appendix 3
Alligator Rivers Region Technical Committee

Key Knowledge Needs 2012–2013

Background

At the 14th meeting of ARRTC, it was agreed that ***eriss*** would present to ARRTC, at its first meeting each calendar year, its proposed research plan for the following financial year and would present a summary of the outcomes of the previous financial year's research at the second meeting. This paper summarises the outcomes from the ***eriss*** research programme for 2012–13 addressing the ARRTC Key Knowledge Needs (KKN).

It should be noted that not all of the projects in ***eriss***'s KKN project suite have been reported on since, for reasons of staff resourcing or other demands in the context of absolute project priority, no substantive work may have been done on a particular project during the past year. In order to ensure that ARRTC is kept fully informed about the status of the entire ***eriss*** project suite a list of those projects that have not been reported on, and a summary of the reasons for the reduced activity, is provided in Appendix 1 of this report.

The outcomes from the full ***eriss*** 2012–13 research program, including the KKN components, will be formally published in the Supervising Scientist Report series of publications.

Much of the monitoring and research work conducted by ***eriss*** is focussed on the wet season and its immediate aftermath since it is during this period that the environment is potentially at most risk from past and current uranium mining activities. The 2012–13 wet season was slightly below average with a total of rainfall at Jabiru Airport of 1238 mm (annual average 1536 mm).

A map of the Ranger mine site and its immediate surrounds, and locations of other key sites used for ***eriss***' research and monitoring programmes are provided for reference in Maps 1 to 3 at the start of this report.