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eriss research summary

2007–2008

DR Jones & AL Webb (eds)

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Supervising Scientist

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

This Supervising Scientist Report is a summary of the 2007–2008 research program of the Environmental Research Institute of the Supervising Scientist and has been reviewed internally by senior staff and the editors of this volume.

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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 Environment, Water, Heritage and the Arts (DEWHA). ***eriss*** provides specialist technical advice to the Supervising Scientist on the protection of the environment and people of the Alligator Rivers Region from the impact of uranium mining. A major part of its function is to conduct research into developing best practice methodologies for monitoring and assessing the impact of uranium mining on water and air (transport pathways) and soil, and on the bushfoods that are consumed by the local indigenous people. This research spans the operational, decommissioning, and post rehabilitation phases of uranium mining in the ARR.

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

The Supervising Scientist series of reports (SSRs) has documented the evolution of the uranium research program and the broader scientific activities of the Institute since 1995. The SSRs are subject to a rigorous technical review (including formal external peer review) and editorial process and are produced by SSD's publications section. They complement publication of the Institute's work in peer-reviewed journals.

The first SSR (number 101) was published in 1995. This latest SSR (number 200) represents a major milestone in the life of the Institute, not simply because it is the 100th SSR, but much more importantly because it documents a major evolution in the scope and nature of the physico-chemical and in situ biological monitoring components of the water quality monitoring program being conducted adjacent to Ranger mine. This evolutionary change is the culmination of many years of research into the development and validation of new approaches that aim to provide a more robust level of environmental assurance.

The balance and strategic prioritisation of work within the uranium component of ***eriss***'s project portfolio is defined by Key Knowledge Needs (KKNs) originally developed in 2004 through consultation between the Alligator Rivers Region Technical Committee (see ARRTC membership and function in Appendix 2), the Supervising Scientist, Energy Resources of Australia and other stakeholders. The KKNs comprise six thematic areas based primarily on geographic provenance (Appendix 3). The content of the research programs developed for each of these areas is assessed and reviewed annually by ARRTC in consultation with stakeholder groups.

At the time the original KKNs were formulated in 2004, Ranger mine was planned to close in 2011. However, following substantial increases in the market price of uranium and the identification of additional resources, it is now expected that processing will extend to at least 2020, with rehabilitation to 2026. As a result of this extension in mine life and the conduct of three years of research since the original KKNs and timeline priorities had been established, as well as the Jabiluka project now being in care and maintence for an indefinite period, it was judged by ARRTC that a revision of the KKNs should be conducted in 2007–08. This was done and a revised list of KKNs approved by the committee.

The original (2004–2006) KKNs are listed in Appendix 3A and the new (2008–2010) KKNs are listed in Appendix 3B so that the updates that have been made can be clearly identified. The KKN numbers that are used in this report correspond to those in Appendix 3A since the

2007–08 projects were initiated when these KKN numbers were extant. From 2008–09 onwards the KKN numbers in Appendix 3B will apply.

Not all of the KKN research areas are able to be covered by ***eriss***, since not all of the required disciplines are available within the Institute. To address these gaps, collaborative projects are initiated with researchers from other organisations. KKN projects related to the detailed hydrogeology or tailings management on the Ranger lease are conducted and reported separately by consultants engaged by Energy Resources of Australia Ltd.

This report documents research projects undertaken by ***eriss*** over the 2007–08 financial year. In contrast to the preceding wet seasons of 2005–06 and 2006–07, both of which had well above average rainfall and extreme events (a cyclone and 800 mm in three days, respectively), the 2007–08 wet season rainfall of 1658 mm was only slightly above the average of 1500 mm. This 'normal' wet season provided the opportunity to finish several important projects that had been been put back by the damage done to monitoring systems infrastructure by the extreme events.

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

1 Ranger – Current Operations

2 Ranger – Rehabilitation

3 Jabiluka

4 Nabarlek

5 General Alligators Rivers Region

6 Knowledge Management and Communication

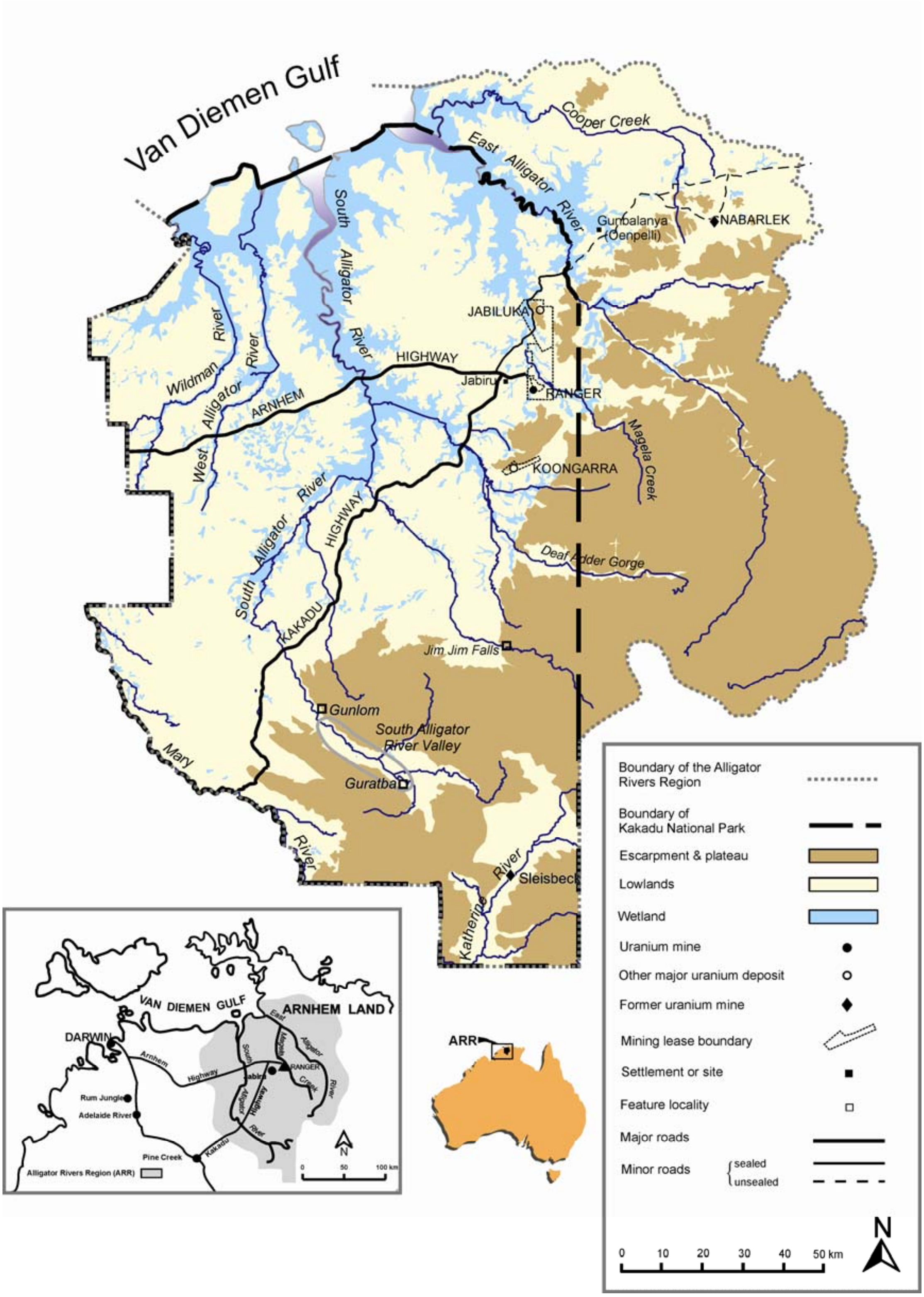
Three maps (following this Preface) provide the regional context for the locations that are referenced in the research papers. Map 1 shows Kakadu National Park and the locations of Ranger mine, Jabiluka project area, the decommissioned Nabarlek mine, and the South Alligator River Valley. A schematic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

The final section of the report contains a list of the non-uranium mining related external projects.

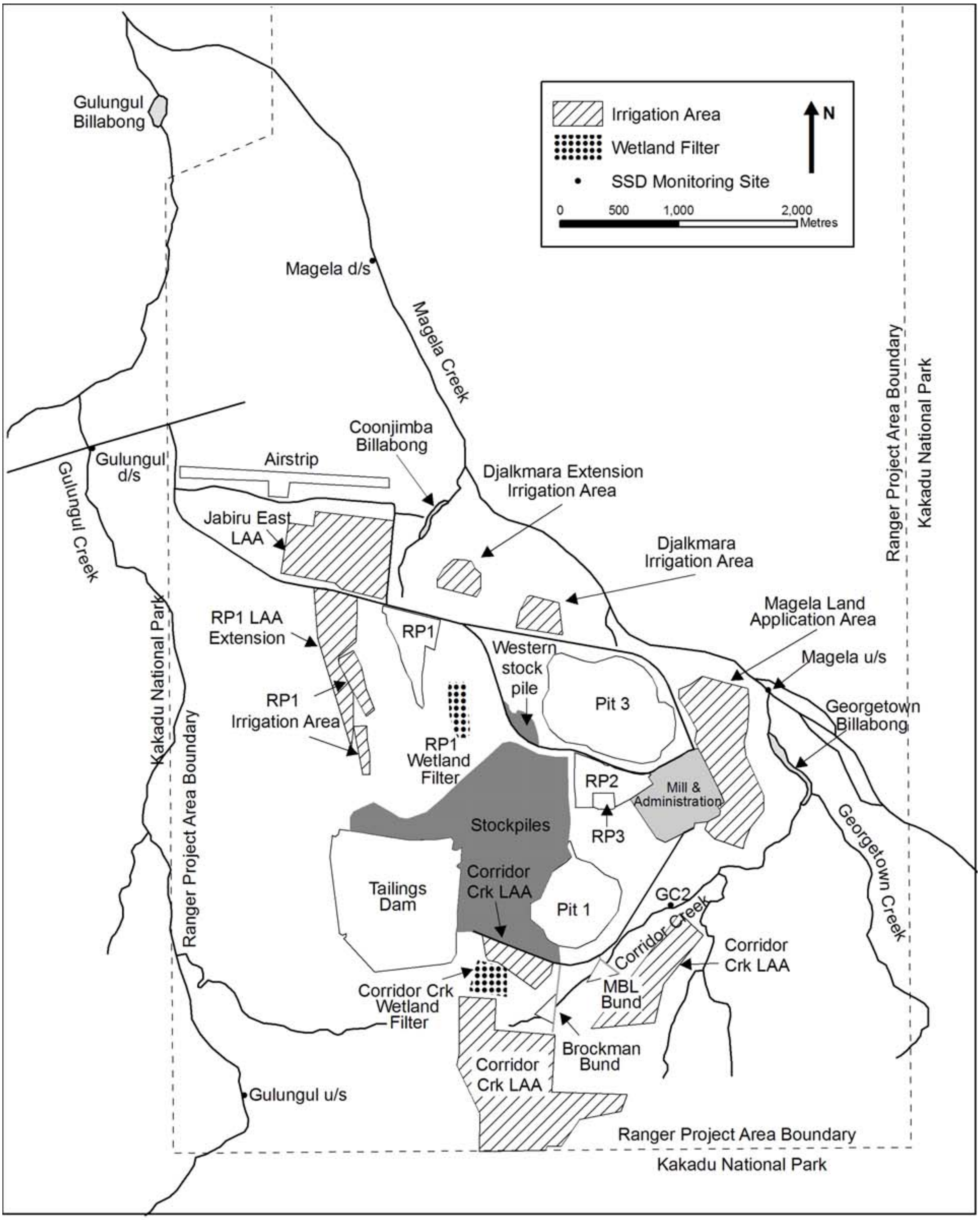
For additional information, readers are referred to the annual publications list (Appendix 1) that details all of the material published, and conference and workshop papers presented by ***eriss*** staff in 2007–08.

Dr DR Jones

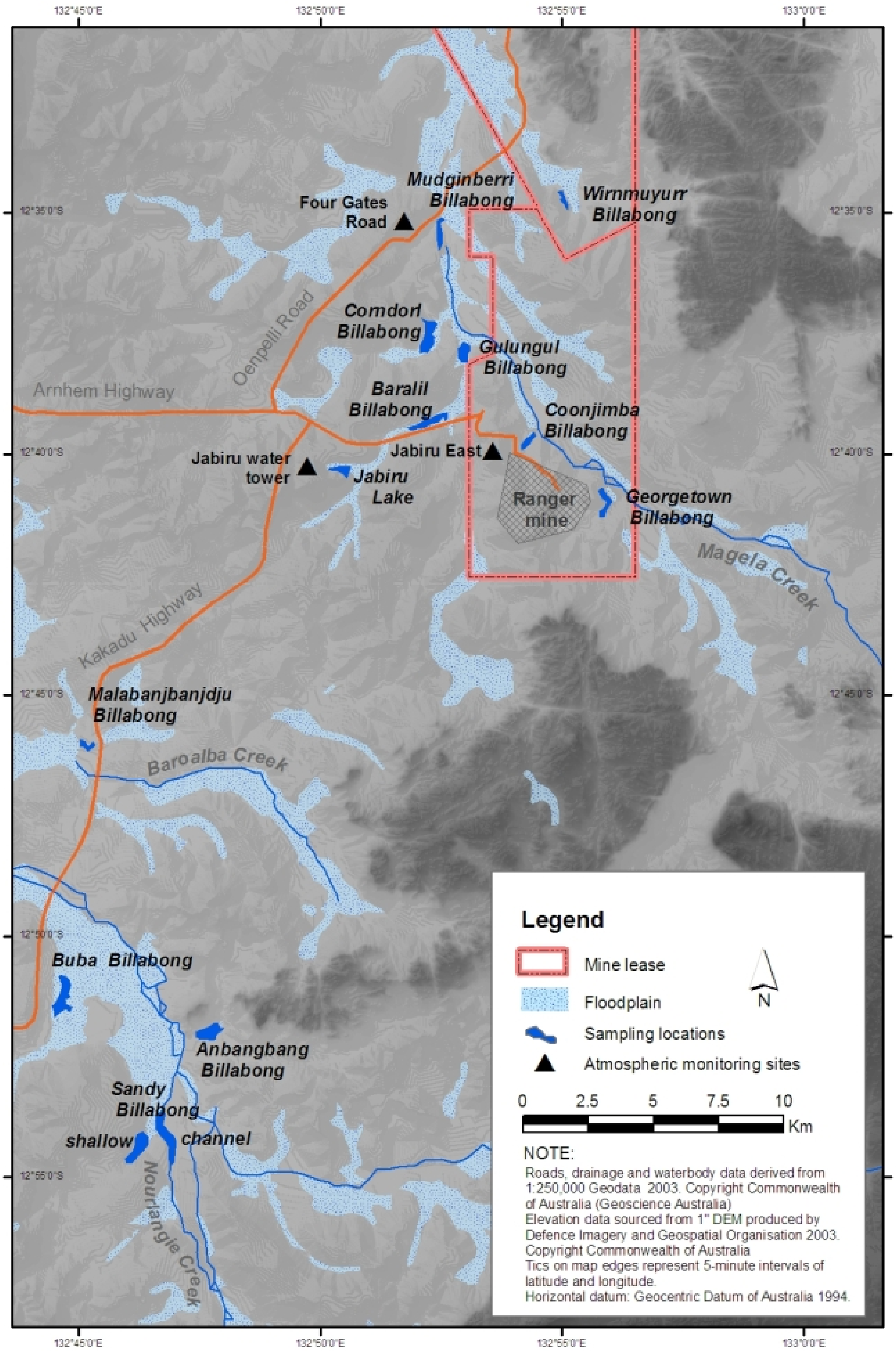
Director, Environmental Research Institute of the Supervising Scientist



Map 1 Alligator Rivers Region



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



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

Part 1: Ranger – current operations

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

K Cheng, D Parry[1], A Hogan & R van Dam

Background

Toxicity tests to assess the effects of uranium (U) on freshwater species local to the Alligator Rivers Region have been employed since the late 1980s. From some of these data a high reliability site-specific water quality Limit for Magela Creek downstream of Ranger has been dervied, using the approach recommended by the Australian and New Zealand Water Quality Guidelines (WQGs; ANZECC/ARMCANZ 2000). The Limit of 6 µg L^{-1} U was derived using chronic toxicity no-observed-effect concentration (NOEC) data, ranging from 18–810 µg L^{-1}, for 5 species (Hogan et al 2005). However, two of the NOEC values, 400 and 810 µg L^{-1}, represent estimates for two fish species, the purple-spotted gudgeon, *Mogurnda mogurnda* and the chequered rainbowfish, *Melanotaenia splendida inornata*, respectively, based on mortality after only 7 d exposure (+ 7 d post-exposure for *M. mogurnda*; Holdway 1992). Although this endpoint satisfies the current WQGs criterion for a 'chronic' endpoint (ie >96 hour test duration), its appropriateness as an indicator of longer-term, sub-lethal chronic effects has been questioned.

In 2006-07 a 28 d larval growth chronic toxicity test protocol was developed for *M. mogurnda*. This test protocol was then used to assess the chronic toxicity of U[2], making comparisons to previous U toxicity data and discussing the implications for the uranium Limit for Magela Creek (see Cheng et al 2008a). The aims for 2007–08 were to:

1 Undertake a second uranium chronic toxicity test using larval *M. mogurnda*, focusing on concentrations within the range of approximately 500 to 3000 µg L^{-1}; and

2 Analyse whole body uranium content in larval *M. mogurnda* after 28 days exposure.

Chronic toxicity and uptake of uranium

The chronic toxicity test protocol was described in detail by Cheng (2008). Newly hatched (<10-h old) *M. mogurnda* larvae were exposed to various concentrations of U (control, 400, 800, 1200, 1600, 2000 and 2200 µg L^{-1} – measured concentrations) for 28-d. For each U concentration, 10 larvae were placed in each of four replicate test containers, each containing 500 mL of test solution. Test solutions were renewed daily, and larvae were fed live brine shrimp (*Artemia salina*) nauplii two times a day (morning and early evening), at a feeding rate of approximately 10–20, 20–30 and 30–40 nauplii per larva, through days 1–7, 8–25 and 26–28, respectively. Larval survival was monitored throughout and at the end of the test, while larval length and dry weight were measured at the end of the test.

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1 Formerly School of Science and Primary Industries, Charles Darwin University, Darwin NT 0909; at time of publication: Australian Institute of Marine Science (AIMS NT), Arafura Timor Research Facility, PO Box 41775, Casuarina NT 0811.

2 Project undertaken under Charles Darwin University Animal Ethics Approval Ref No. A06008.

At the end of the test, surviving larval *M. mogurnda* were euthanased and rinsed thoroughly in Milli-Q water, dried (72-h at 60°C; for dry weight and U analysis) and digested using AR grade HNO_3 followed by H_2O_2 at 135°C (for whole body U content). Uranium in larval whole body digests was measured using ICP-MS (Agilent 7500ce) The minimum biomass required for metal analysis was approximately 0.3 g in dry weight. In order to obtain this, replicate samples were pooled and, therefore, no statistical tests for significant differences between treatments could be performed on the data.

The results of the initial and second U toxicity tests are summarised in Table 1, while the effects of U exposure on larval *M. mogurnda* in the second test only are shown in Figure 1. Exposure to 1600, 2000 and 2200 µg L^{-1} uranium resulted in 100% larval mortality within the first 48-h of exposure. Significant mortality (~80%; ANOVA, $P < 0.05$) was observed in the group exposed to 1200 µg L^{-1}. Larvae exposed to 800 and 1200 µg L^{-1} U for 28-d exhibited significant 8% and 25% reductions in length, and 15% and 60% reductions in dry weight, respectively, relative to control larvae (ANOVA, $P \leq 0.05$). Based on larval length and dry weight, the lowest-observed-effect concentration (LOEC) and no-observed-effect concentration (NOEC) were 800 and 400 µg L^{-1} U, respectively (Figure 2). As can be seen in Table 1, larval *M. mogurnda* in test 2 were more sensitive to U than in test 1.

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

Test	Endpoint	NOEC (µg L^{-1} U)	IC_{10}(µg L^{-1} U)	IC/LC_{50}(µg L^{-1} U)
1	Survival	1400	-	2150
(June 2007)	Dry Weight	880	862	>1400
	Length	880	1162	>1400
2	Survival	800	-	1060
(February 2008)	Dry Weight	400	652	1110
	Length	400	847	>1200

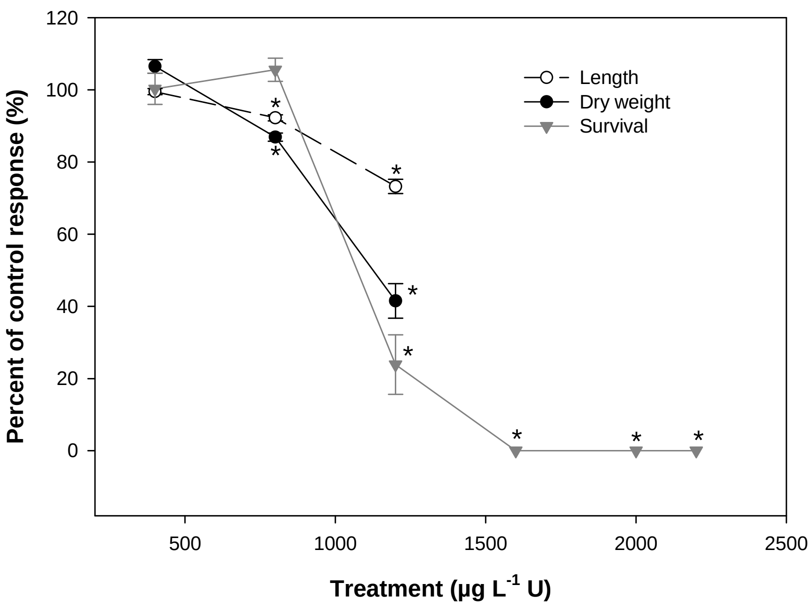

Figure 1 Uranium chronic toxicity test 2 (2007–08): Effect of U exposure over 28-d on mean (±SEM) dry weight (n = 4), length (n = 9–40) and survival (n = 4) of larval *M. mogurnda*, normalised against the control responses. Data points accompanied by an asterisk are significantly different from the control responses ($P \leq 0.05$). Mean control responses (±SEM) were as follows: dry weight – 1.70 (±0.03) mg; length – 11.2 (±0.1) mm; and survival – 95 (±3) %.

Whole body U concentrations for *M. mogurnda* in both tests are summarised as a pooled dataset in Figure 2 (data were pooled due to the similar response observed between tests). Uranium concentrations in control larvae were less than 0.05 µg g^{-1} for both toxicity tests. Results obtained from U exposed groups showed a dose dependent relationship, with whole body U concentration increasing with increasing exposure concentration. The similar whole body U concentrations for larval *M. mogurnda* at similar U exposure concentrations between the two tests suggest that the observed difference in toxicity may not have been due to a difference in U uptake (although it is acknowledged that the whole body concentrations may include U adsorbed to the outer surfaces of, as well as U taken up by, the larvae). Geochemical speciation modelling for U (see below) may shed further light on the bioavailability of U between the two tests.

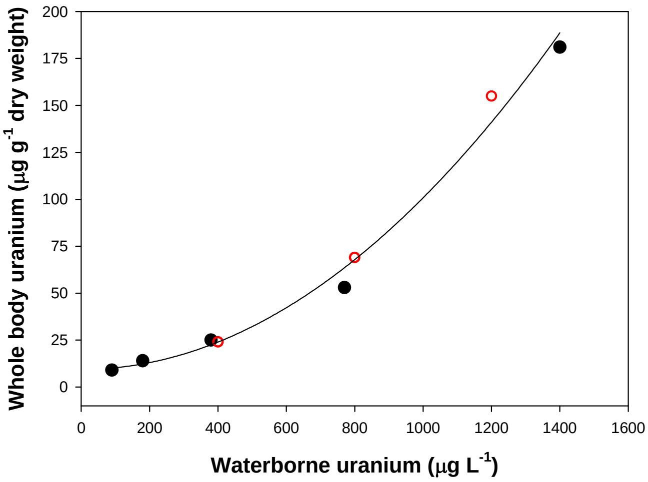

Figure 2 Whole body uranium concentrations for larval *M. mogurnda* following a 28-d exposure period. Data are pooled from tests 1 (closed circles) and 2 (open circles), and the associated relationship is described by a quadratic model ($R^2 = 0.983$, df = 7, $P < 0.001$).

Following two independent U chronic exposure tests, *M. mogurnda* did not appear to be more sensitive than that previously reported by Holdway (1992) following shorter (ie 7 and 14-d) exposure periods. This indicates that the one to two week period post-hatch provides the most sensitive time window for assessment of the toxic effects of U on *M. mogurnda*, and that longer exposure periods will not necessarily result in a more sensitive response. Consequently, the historical 7-d fish toxicity test results used for the derivation of the current U Limit of 6 µg L^{-1} appear to be reasonably representative of U concentrations that will not result in longer-term chronic effects. This provides assurance that the current uranium Limit is sufficiently conservative to ensure protection from chronic toxicity effects of U.

The results of this project have been published as an Internal Report (Cheng 2008; CDU Honours project) and were presented at the SETAC 5th World Congress in Sydney, 3–7 August 2008 (Cheng et al 2008b).

Steps for completion

At present, a manuscript detailing the results of this study is being drafted for peer-reviewed publication. The manuscript will include an assessment of U speciation and bioavailability based on geochemical speciation modelling.

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Influence of dissolved organic carbon on the toxicity of uranium

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

Background

Mining represents one of the threats to the quality and biodiversity of freshwater ecosystems in northern Australia. Uranium (U), aluminium (Al) and arsenic (As) are priority metals of ecotoxicological concern for the region's mining industry for which insufficient toxicity data exist (Markich & Camilleri 1997). Uranium is a metal of concern for the Magela Creek system adjacent to the Ranger mine, whilst Al and As are of general concern for mining in the broader northern Australian region (Markich & Camilleri 1997, Lloyd et al 2002).

Few existing studies of the toxicity of these metals have incorporated dissolved organic carbon (DOC) as a variable (see toxicity reviews for U, Al and As by Sheppard et al 2005, Gensemer & Playle 1999 and Gomez-Caminero et al 2001, respectively), despite humic substances being recognised as playing a major role in metal speciation (Tipping 2002). The few tropical freshwater toxicity studies that have investigated the influence of DOC on metal speciation and bioavailability have demonstrated it can be a key determinant of metal toxicity (Markich et al 2000; Hogan et al 2005). It is essential to build upon these studies in order to gain a comprehensive understanding of the role of DOC in metal bioavailability and toxicity in tropical freshwater ecosystems.

The objective of this study is to quantify the influence of DOC on the toxicity of U, Al and As to four freshwater species, under fixed conditions of pH, water hardness and alkalinity.

Methods

The selected tropical species, northern trout gudgeon, *Mogurnda mogurnda*, green hydra, *Hydra viridissima*, the unicellular alga, *Chlorella* sp and a unicellular flagellate, *Euglena gracilis*, were chosen to cover a range of trophic levels. Laboratory toxicity testing is being conducted using a synthetic soft water (characteristic of sandy braided streams in tropical Australia) and a Suwannee River Fulvic Acid Standard I (SRFA) produced by the International Humic Substances Society (IHSS). Tests will also be conducted using natural waters with a range of DOC concentrations. SRFA was selected as the organic carbon source for this study as it is a well characterised reference material (Cabaniss & Shuman 1988) and the only biologically relevant standard for this study. This summary focuses on the toxicity of U to the first three aforementioned species.

The test organisms were exposed to a range of U (0 to 14, 1.1 and 1.5 mg/L, for gudgeon, hydra and alga, respectively) and DOC concentrations (0, 1, 5, 10 and 20 mg/L) in combination. The synthetic water used was of very low ionic strength (low hardness and alkalinity) and was slightly acidic (pH 6). Test systems were static with no test solution

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renewals for *Chlorella* sp and 24-h renewals for *M. mogurnda* and *H. viridissima*. Test temperatures were maintained at 27±1°C for *M. mogurnda* and *H. viridissima* and 28±1°C for *Chlorella* sp. Three tests were run for each species in order to fully characterise the concentration-response relationships.

The test durations and endpoints for the tests were as follows: *M. mogurnda* 96-h sac-fry survival; *H. viridissima* 96-h population growth rate; *Chlorella* sp 72-h population growth rate. For all tests, water parameters (pH, DO, EC) were monitored daily. Water samples were taken for analyses of DOC, alkalinity, hardness and a standard suite of metals and major ions. For each species, response data from three tests were pooled, and concentration–response relationships were determined (using regression analyses). Physicochemical variables were input into the HARPHRQ geochemical computer speciation model to determine the effect of DOC on U speciation, which was related back to U toxicity.

Progress

The U toxicity testing was completed for all three species. U toxicity to all species was reduced substantially with increasing SRFA. Concentration-response relationships, and associated linear regressions of toxicity (expressed as IC/LC_{50}) against DOC concentration, are shown in Figure 1. Toxicity summary data and fitted regression equations are shown in Table 1.

Table 1 Summary of selected uranium toxicity testing results for three local freshwater species

Species	DOC[a] (mg/L)	IC_{50}[b] (95%CL)[c] (μg/L)	Reduction in toxicity with 20× DOC increase	Regression equation, r^2 & *P* value for IC_{50} v DOC plots in Fig 1
Mogurnda mogurnda[d]	0 20	1550 (1057–1961) 7200 (5907–8903)	4.6×	IC_{50}=287[DOC]+1548, r^2=98%, *P*<0.0001
Hydra viridissima (green hydra)	0 20	65 (8–85) 470 (404–512)	7×	IC_{50}=19[DOC]+85, r^2=91%, *P*<0.0001
Chlorella sp (unicellular alga)	0 20	38 (22–69) 656 (454–954)	17×	IC_{50}=30[DOC]+86, r^2=92%, *P*<0.0001

a DOC: dissolved organic carbon

b IC_{50}: concentration that results in a 50% inhibition of response relative to the control response using dose-response relationships determined by ToxCalc

c 95% CL: 95% confidence limits determined by ToxCalc

d For *M. mogurnda*, the toxicity estimates relate to concentrations that affect survival, compared to sub-lethal endpoints, such as growth and reproduction, for the other species

The extent to which DOC ameliorated U toxicity differed for each species. DOC appeared to result in a more gradual (but higher overall) reduction in U toxicity for *Chlorella* sp and *H. viridissima* than *M. mogurnda*. For *M. mogurnda*, there was an apparent increase in the threshold U concentration (the point at which survival dropped from 100%) with increasing DOC. However, the slope of the response curve from 100% to 0% survival was similar for all DOC concentrations. This may indicate a similar toxic response is occurring across DOC treatments once cation-binding sites on the DOC are saturated with UO_2^{2+} or $UO_2(OH)^+$, the most toxic forms of U in solution around pH 6.

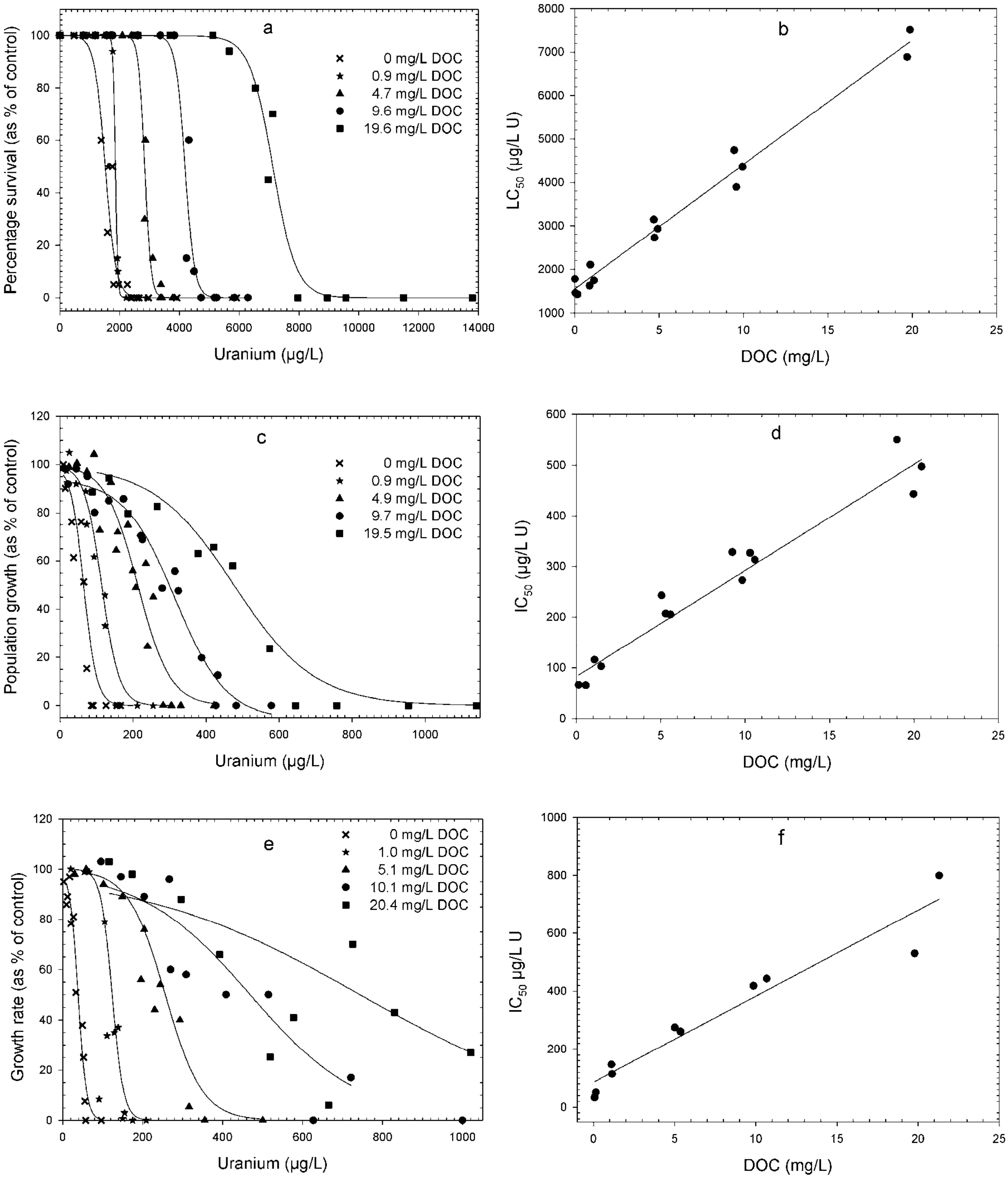

Figure 1 Effect of DOC on U toxicity to *Mogurnda mogurnda* (a & b), *Hydra viridissima* (c & d) and *Chlorella* sp (e & f). Left plots represent the concentration-response relationships, with curve fits based on a sigmoidal, 3-parameter model. Right plots represent the linear regressions of U toxicity (expressed as the IC/LC_{50}) against DOC concentration.

The decrease in toxicity of U in the presence of SRFA for all three species was shown to be due to a reduction in the free uranyl ion concentration due to its being bound by the fulvic acid (Figure 2). Differences in the proportions of U species between the three test species are most likely due to small pH and other physico-chemical differences in the test diluent waters.

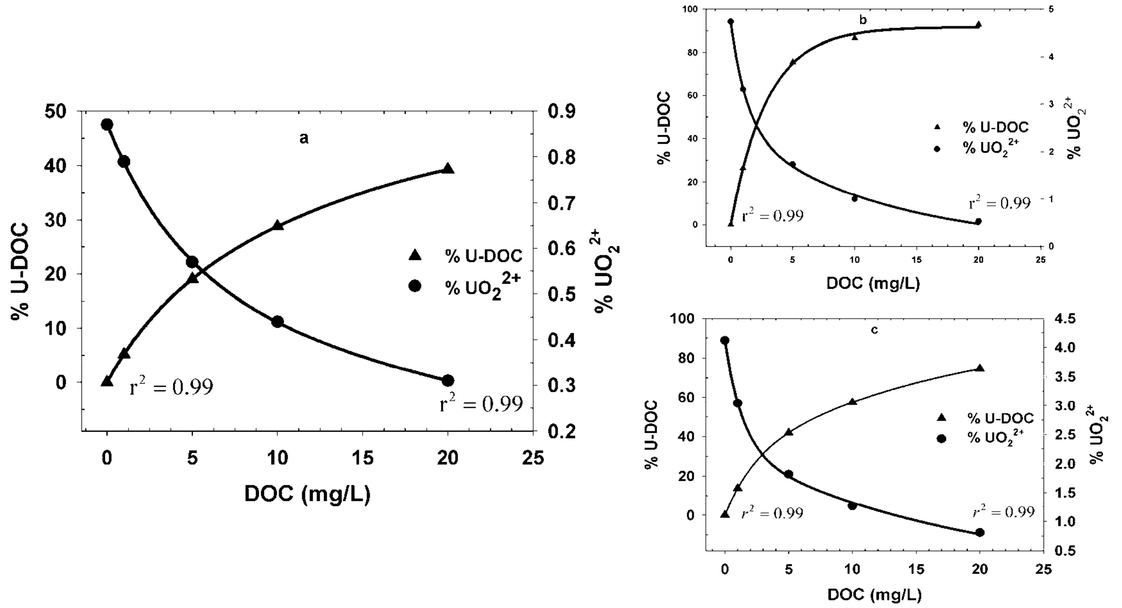

Figure 2 Percentage of U bound to DOC and UO_2^{2+} in solution at increasing DOC concentrations calculated using the HARPHRQ speciation model: (a) *Mogurnda mogurnda*, (b) *Hydra viridissima* and (c) *Chlorella* sp

Concentration-response plots were produced recently based on the concentration of the two uranyl species (UO_2^{2+} and UO_2OH^+) suggested by Markich et al (2000) to be most responsible for inducing toxicity (Figure 3). If U toxicity could be explained entirely by the concentrations of the above two U species, the separate DOC plots in each of Figure 3a–c would be expected to converge into one concentration-response relationship. Substantial convergence was observed for all three species, suggesting that U toxicity was likely to be due largely to UO_2^{2+} and UO_2OH^+. However, for *M. mogurnda* and *Chlorella* sp in particular, other U complex species may also be contributing to toxicity. In this context it must be noted that the SRFA represents a distribution of organic species, not a single organic chemical reagent. Hence it is not unreasonable to expect that there would be a distribution of toxicity across the range of U-organic species. Stepwise multiple regression will be used (as per Markich et al 2000) to identify the U species most likely contributing to the observed toxic responses.

The next phase of the work for U will involve comparison of the above U toxicity results obtained using SRFA in SSW to that in natural waters with natural DOC. This will be done using DOC-rich fresh water from billabongs. Attentuation of U toxicity will be especially important in natural billabongs, such as Georgetown Billabong on the Ranger lease, where the setting of water body specific surface water closure criteria for U may require consideration of the DOC concentrations, which can be higher than in Magela Creek.

The next stage of the project involving the effect of SRFA on toxicity will be undertaken for Al and As, using the same suite of test organisms as for U. The effect on As toxicity will be especially interesting given that As occurs as an oxyanion with very different chemistry, compared with the cationic forms of Al and U.

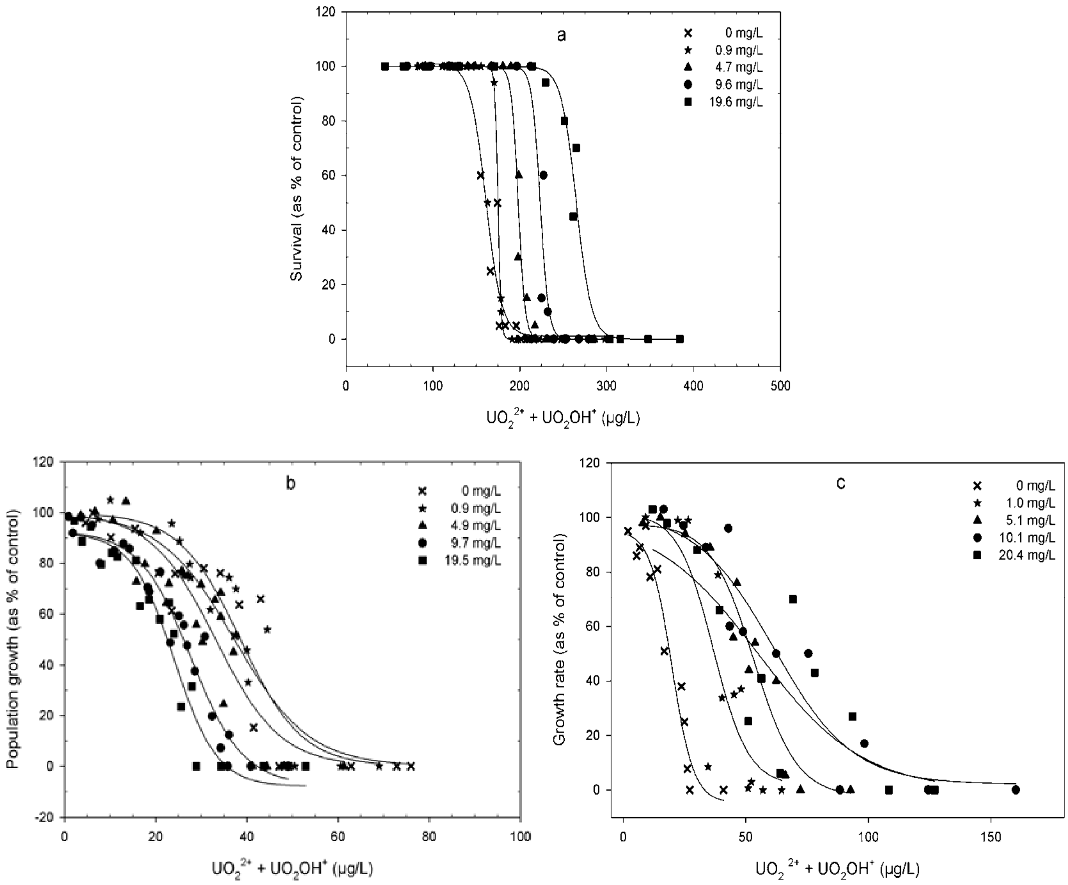

Figure 3 Concentration response plots based on the concentration of the two U species, UO_2^{2+} and UO_2OH^+: (a) *Mogurnda mogurnda*, (b) *Hydra viridissima*, (c) *Chlorella* sp

In addition, short-term (~3 minute) exposures of unicellular *Euglena gracilis* to all three metals will be used to assess cellular responses to each metal and DOC combination. The wildtype plant strain and mutant animal strain of this species will hopefully enable sites of intracellular damage to be identified and for this information to aid in assessment of potential toxic mechanisms in other unicellular or multicellular organisms. The use of *Euglena* as a test species is beneficial not only because it is a simple, rapid test system but also because small volumes of test solution are required (which lowers the demand for costly fulvic acid standard and the volume of natural water required to be collected).

The results of this study were presented at the 5th SETAC World Congress in Sydney, 3–7 August 2008 (Houston et al 2008a) and the 14th Meeting of International Humic Substances Society in Moscow, 13–19 September 2008 (Houston et al 2008b). This project is funded by an ARC Linkage grant (LP 0562507).

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Preliminary assessment of the toxicity of manganese to three tropical freshwater species

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Background

Increased attention was paid to manganese (Mn) as a contaminant of potential ecotoxicological concern at Ranger in the early 2000s following observations of increasing concentrations in a shallow groundwater bore adjacent to Magela Creek, (MC20; up to 50 000 µg L^{-1}). Addtionally, concentration 'spikes' have been observed in early wet season surface water in lower Corridor Creek (GC2; 700–800 µg L^{-1}) and Coonjimba Billabong (1300 µg L^{-1} in December 2002/January 2003) (van Dam 2004). Since then, Mn concentrations in bore MC20, which is in a local depression and acts as a collection point for surface drainage, have consistently been measured at 40 000–50 000 µg L^{-1} during the dry season (ERA 2008), with much lower values (100–1000 µg L^{-1}; based on limited data) in the wet season following flushing of the shallow groundwater system. This appears to be a localised effect, with dry season Mn concentrations in nearby shallow groundwater bores over the same time period being at least two orders of magnitude lower than in bore MC20. Four more occurrences of Mn above 800 µg L^{-1} (with a maximum of 1690 µg L^{-1} in November 2004) have been measured at GC2, while Coonjimba Billabong has experienced one additional spike above 800 µg/L, in December 2007 (ERA 2008). Two of the measured spikes exceeded the ANZECC/ARMCANZ (2000) 99% species protection trigger of 1200 µg L^{-1}, and were above concentrations reported in the literature to cause chronic toxicity to some species.

Evidence from the literature suggests the acute and chronic toxicity of Mn to freshwater biota is low (ie in the mg L^{-1} range), and this is reflected in the relatively high trigger value reported above. The current site-specific guideline for Mn in Magela Creek downstream of Ranger is 26 µg L^{-1}. This value was derived from statistical analysis of water quality data from the upstream reference site data, and applicable only when flow in Magela Creek is greater than 5 cumecs. It is approximately two orders of magnitude more conservative than the ANZECC/ARMCANZ (2000) trigger value. Since 1980, this local guideline has been exceeded less than 2% of the time. The majority of exceedances have occurred during early wet season flows or end of wet season recessional flows, often when flow is less than 5 cumecs. These periods are considered to be atypical of the season as a whole given the increased contributions from shallow groundwater at these times. Based on the very low frequency of exceedance of the local guideline for Mn, and the existing ANZECC/ARMCANZ (2000) 99% species protection trigger value of 1200 µg L^{-1}, it was perceived that Mn posed a low toxicity risk to aquatic biota in Magela Creek.

However, 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 could potentially result in higher than expected (ie from existing literature) Mn toxicity.

A previous, albeit limited, study by ***eriss*** (unpublished data, 1993) on the toxicity of Mn to *Hydra viridissima* (population growth) reported relatively low NOEC/LOEC values of 20/200 $\mu g\ L^{-1}$ and 180/630 $\mu g\ L^{-1}$ in synthetic and natural water, respectively, compared to the existing literature. The apparent sensitivity of *H. viridissima* to Mn and paucity of data for other local species provided sufficient basis to conduct a pilot site-specific ecotoxicological assessment, since it is possible that in the future higher concentrations of Mn could occur in the Creek as a result of inputs from the mine site.

The aim of this study was to determine the toxicity of manganese (Mn) to three generally sensitive tropical freshwater species, the green alga, *Chlorella* sp, the green hydra, *Hydra viridissima*, and the cladoceran, *Moinodaphnia macleayi*. Specifically, the study set out to assess whether Mn may represent a significant environmental hazard downstream of Ranger, and hence warrant a further, more detailed risk assessment.

Methods

Mn is a difficult metal to work with, because its chemistry in water is a complex function of pH and redox microenvironment. The kinetics of Mn(II) oxidation increases at higher pH. Thus, a lower pH diluent water (Ngarradj Creek Water, NCW) was chosen to reduce the probability of particulate formation as a result of the oxidation of Mn(II) in solution to form Mn(III)/Mn(IV) oxyhydroxide precipitates. NCW was collected from near the Ngarradj Creek Upstream gauging station (NCUS: 0275473; 8616847; WGS84, Zone 53) and was filtered (2.5 μm) upon arrival at the ***eriss*** laboratories. All manganese treatments were diluted in NCW. In addition, a Magela Creek Water (MCW) quality control group was included for each test (ie organisms were cultured in the standard natural MCW; pH – 6.77, EC – 16 μS/cm, DO – 97.5% saturation).

Numerous water samples (total and 0.1 μm filtered) for chemical analysis were collected and analysed both before and after exposure to track the status of the added Mn. Filtration through 0.1 μm membranes, rather than the conventional 0.45 μm filtration, was used specifically for this work to provide increased ability to identify Mn oxides in colloidal form.

One experiment was undertaken for *Chlorella* sp and *H. viridissima*, while for *M. macleayi* three chronic toxicity tests and an acute test were conducted (Table 1).

Table 1 Details of the Mn toxicity tests conducted

Test ID and Date	Species name	Endpoint	Test Duration	Feeding/ nutrition	Acute/ Chronic	Static/ daily renewals
933D 31/05/08	*Moinodaphnia macleayi*	Reproduction	3 broods 120–144 h	30 μl FFV only	Chronic	Daily renewals
934D 31/05/08	*Moinodaphnia macleayi*	Reproduction	3 broods 120–144 h	30 μl FFV + 6 x 10^6 cells of *Chlorella* sp	Chronic	Daily renewals
936B 16/06/08	*Hydra viridissima*	Population growth	72 h	30–40 artemia nauplii	Chronic	Daily renewals
938I 20/06/08	*Moinodaphnia macleayi*	Survival	48 h	No food	Acute	Static
937D 20/06/08	*Moinodaphnia macleayi*	Reproduction	3 broods 120–144 h	30 μl FFV + 6 x 10^6 cells of *Chlorella* sp	Chronic	Daily renewals
939G 24/06/08	*Chlorella* sp.	Population growth	72 h	14.5 mg/L NO_3 0.14 mg/L PO_4	Chronic	Static

With the exception of one of the *M. macleayi* tests (see below), all experiments were conducted in accordance with the standardised ***eriss*** ecotoxicological protocols described in Riethmuller et al (2003). Two of the *M. macleayi* chronic toxicity tests were conducted simulatanouelsy with one of the tests excluding the algal component of the cladocerans' food (Table 1). This was done to determine if the presence of actively photosynthesising algae would result in oxidation of the manganese and production of insoluble manganese oxyhydroxides (MnO), thereby reducing the bioavailablility and toxicity of Mn.

Results and discussion

Chemistry

Prior to filtering, the NCW had a pH of 5.29, an electrical conductivity (EC) of 13 µS/cm and a dissolved oxygen (DO) content of 85.5%. Following filtration, the water had a pH of 5.58, an EC of 12 µS/cm and a DO content of 74.9%. For the testing, the pH was slightly higher again, but remained between 6.0–7.0 for all tests. Metal analysis of filtered NCW indicated that it contained some aluminium (3.0 µg L^{-1}), zinc (2.0 µg L^{-1}), nickel (1.6 µg L^{-1}) and manganese (3.8 µg L^{-1}). All other metals analysed were at concentrations <1 µg L^{-1}.

The results of Mn analyses for the toxicity tests are reported in Table 2. The total concentration of Mn did not change during the course of the experiments, indicating that there was no loss to the test system (eg walls of the test vials). At the commencement of the tests, ~92% of the total Mn was present in the <0.1 µm fraction (ie dissolved or very fine colloidal fraction), compared to approximately 86–92% by the end of the tests. Furthermore, tests that did not receive daily water renewal and were conducted over longer time periods (ie 72-h algae test and 48-h acute flea tests) did not show markedly larger losses of Mn. To account for the change in soluble (ie bioavailable) Mn, the calculation of toxicity estimates used an average of the start and end of test filtered concentrations. Analysis of the test solutions from the initial two cladoceran tests (ie 933D and 943D) indicated that significant concentrations of oxidised Mn forms (ie insoluble forms) were not being formed in the presence of photosynthetic organisms (ie the algal food source), which was probably due to the pH of the solutions being below 7.0 units (Richardson et al 1988).

Toxicity

The concentration-response relationships for the three species are shown in Figure 1 and the toxicity estimates and control responses are summarised in Table 3.

The initial chronic toxicity experiment with *M. macleayi* demonstrated that excluding the algal food from the test significantly reduced their reproductive health (Figure 1a). Exposure of *M. macleayi* in the presence of the algal food resulted in no observed toxicity to concentrations up to 1840 µg L^{-1} Mn, while excluding the algal food resulted in a significant reduction in neonate numbers of ~40% at the highest concentration (Figure 1a). Consequently, a 6-d chronic test with algal food and a 48 h acute test without food were conducted at higher concentrations. Both these studies resulted in 100% lethality to *M. macleayi* within 48 h at concentrations ≥1845 µg L^{-1} Mn (Figure 1a & b). A Mn concentration of 870 µg L^{-1} Mn resulted in a statistically significant reduction in the number of neonates (ie 13%) in the chronic test, while in the acute test no significant effects were observed at 770 µg L^{-1} Mn (Table 3). The initial (ie 933D and 934D) and subsequent cladoceran tests (ie 937D and 938I) were somewhat contrasting in terms of the measured responses at around 1800–1900 µg L^{-1}. Nevertheless, the results of the second set of tests indicate a dramatic threshold response for survival at between

1000–2000 $\mu g\ L^{-1}$ Mn. This concentration is quite low compared to other studies reported in the literature with the exception of one previous freshwater study (see below).

Table 2 Measured and predicted[1] Mn concentrations in the tests

Test number/Code	Nominal Mn ($\mu g/L$)	Start of Test ($\mu g\ L^{-1}$)		End of Test ($\mu g\ L^{-1}$)	
		Total Mn	**0.1 μm Filtered Mn**	**Total Mn**	**0.1 μm Filtered Mn**
Initial chronic cladoceran tests					
934D/933D PB[2]	0	0.3	NA[3]	NA	NA
934D/933D A	0 (NCW)	5.3	3.8	***5.3***	***4.6***
934D/933D B	20	7.2	6.5	NA	4.7[4]/4.3[5]
934D/933D C	63	70	60	***70***	***60***
934D/933D D	200	210	190	NA	150/150
934D/933D E	630	660	600	***660***	***570***
934D/933D F	2000	2040	1940	NA	1740/1800
Chronic Hydra test					
936B PB	0	<0.01	NA	NA	NA
936B B	0 (NCW)	6.3	5.0	***6.3***	***5.5***
936B C	200	200	140	170	70
936B D	666	670	540	***670***	***580***
936B E	2000	2070	1800	2170	1650
936B F	6660	6600	6470	***6590***	***5700***
936B G	20,000	22,100	19200	21700	19100
Repeat chronic cladoceran test					
937D PB	0	0.06	NA	NA	NA
937D B	0 (NCW)	5.1	4.9	***5.1***	***4.4***
937D C	1000	1030	950	1010	800
937D D	2000	2080	1910	***2080***	***1800***
937D E	4000	4160	3800	4080	3700
937D F	8000	8380	7900	***8380***	***7250***
937D G	16000	16500	15500	16300	1510
Acute cladoceran test					
938I PB	0	0.06	NA	NA	NA
938I B	0 (NCW)	5.1	4.9	***5.1***	***4.4***
938I C	1000	1030	950	1010	590
938I D	2000	2080	1910	***2080***	***1800***
938I E	4000	4160	3800	4050	3570
938I F	8000	8380	7900	***8380***	***7250***
938I G	16000	16500	15500	16400	14700
Chlorella test					
939G PB	0	0.02	NA	NA	NA
939G B	0 (NCW)	5.2	4.7	***5.2***	***4.5***
939G C	200	220	200	***220***	***190***
939G D	1000	720	660	650	460
939G E	2000	2090	1910	***2090***	***1810***
939G F	8000	7180	6320	7030	5600
939G G	20000	21400	19800	***21400***	***18520***
939G H	66000	68300	***62500***	69000	59300

1 Predicted concentrations (shown in bold italics) were determined based on regression equations derived from the measured Mn concentrations, ie End of test Total Mn = 1 x Start of test Mn Total (r^2 = 0.99); End of test Filtered Mn = 0.87 x End of test Total Mn (r^2 = 0.99).

2 PB=Procedural Blank

3 NA = not analysed

4 Measured Mn at the end of test 933D and [5] Measured Mn at the end of test 934D

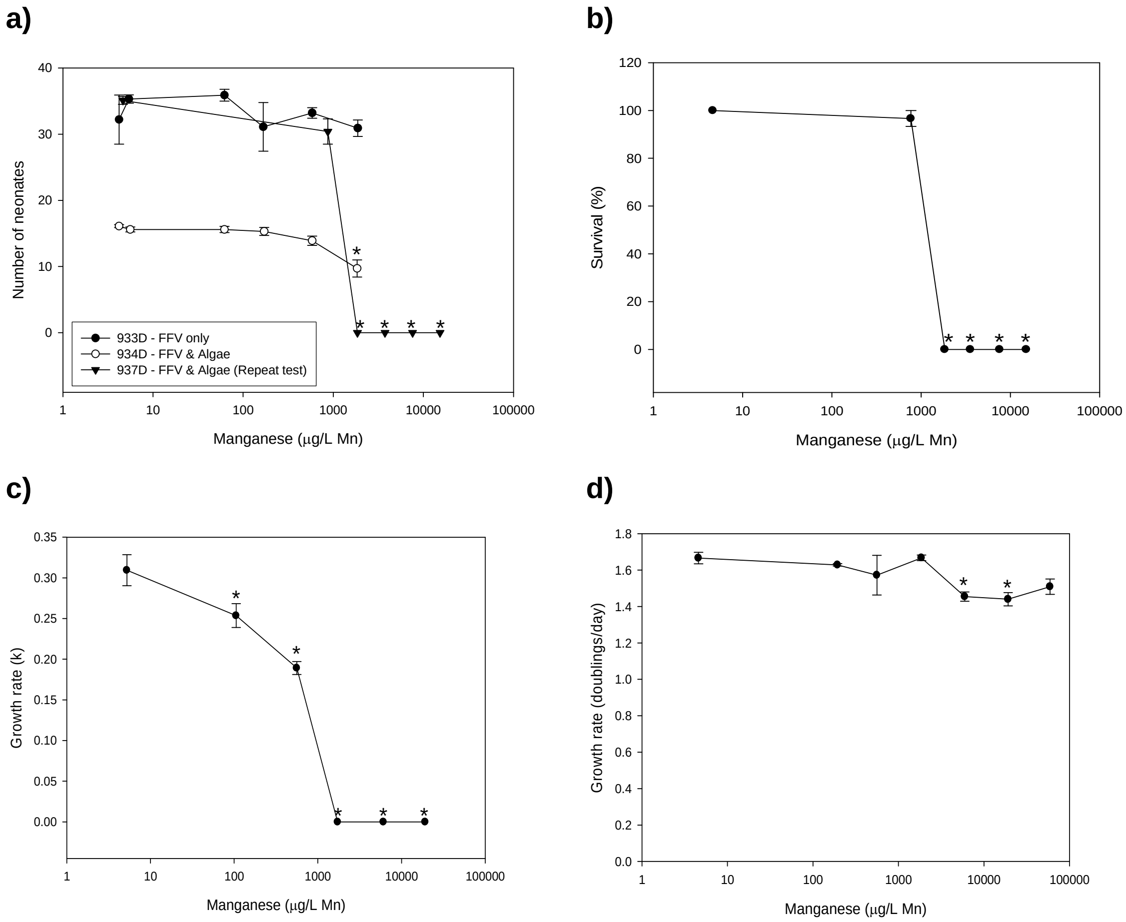

Figure 1 Effect of manganese on a) the reproduction of *M. macleayi* over 6 days; b) the survival of *M. macleayi* over 48 h; c) the population growth rate of *H. viridissima* over 96 h; and d) the growth rate of *Chlorella* sp over 72 h. * denote significantly different from the NCW control (p<0.05).

Of the three species tested, *H. viridissima* was the most sensitive to Mn exposure. The lowest concentration of Mn tested resulted in a significant reduction of population growth rate (ie a NOEC of <106 µg L^{-1} and a LOEC of 106 µg L^{-1}; Figure 1, Table 3). An IC_{10} of 60 µg L^{-1} and an IC_{50} of 770 µg L^{-1} were determined from the concentration-response relationship. Reanalysis of data from the previous *H. viridissima* study conducted at ***eriss*** (described above), based on population growth rate over 96 h as the test endpoint, yielded a NOEC and LOEC of 180 and 600 µg L^{-1}, respectively, an IC_{10} of 270 µg L^{-1} and an IC_{50} of 990 µg L^{-1}.

The IC_{50} values, which are the more reliable toxicity estimate for comparison purposes, demonstrate that the two studies found similar responses of *H. viridissima* to Mn. Only one other study has reported higher toxicity of Mn to a freshwater organism. Fargašová (1997) reported 43% mortality of the midge larva, *Chironomus plumosus*, at 55 µg L^{-1} Mn. However, this was the only concentration tested and many details of the test method (eg. physico-chemistry of diluent water, chemical analysis of the test chemical) were not described, making it difficult to establish the quality of the data. Hence this result was not used by ANZECC/ARMCANZ (2000) in the derivation of the default water quality trigger value for Mn.

Table 3 Summary of the Mn toxicity estimates to three local freshwater species

Test ID and Date	Species name	Endpoint	Control performance			Toxicity (μg L-1)			
			Creek water	mean	%CV[2]	IC10[3]	IC50[4]	NOEC[5]	LOEC[6]
933D 31/05/08	*M. macleayi*	# neonates	Magela	35.4	6.2	1750	>1870	1870	>1870
			Ngarradj	32.2	36.4				
934D 31/05/08	*M. macleayi*	# neonates	Magela	13.6	8.6	410	>1840	580	1840
			Ngarradj	16.1	4.6				
936B 16/06/08	*H. viridissima*	Population growth rate	Magela	0.3	5.8	60 (30–330)	770 (590-940)	<106	106
			Ngarradj	0.3	10.6				
937D 20/06/08	*M. macleayi*	# neonates	Magela	27	43	650 (360–920)	1290 (1200–1340)	<870	870
			Ngarradj	35.1	5.3				
938I 20/06/08	*M. macleayi*	Survival	Magela	100	0	880 (730–880)	1310 (1230–1310)	770	1850
			Ngarradj	100	0				
939G 24/06/08	*Chlorella* sp	Growth rate	Magela	1.8	3.3	5100	<59300	1860	5960
			Ngarradj	1.7	3.3				

[1] Control growth rate in doublings day^{-1}

[2] %CV: percent co-efficient of variation

[3] IC_{10}: the concentration that results in a 10% reduction in growth rate relative to the controls

[4] IC_{50}: the concentration that results in a 50% reduction in growth rate relative to the controls

[5] NOEC: highest concentration tested where growth was not significantly different from the control growth rate

[6] LOEC: lowest concentration tested where growth was significantly different from the control growth rate

Manganese had very little effect on the growth rate of *Chlorella* sp (Figure 1, Table 3) over the concentration range that was tested. An EC_{10} of 5100 μg L^{-1} was calculated, while the EC_{50} could not be determined but was > 59 300 μg L^{-1}. However, due to very low intra-treatment variability in the control and treatment groups in a statistically significant inhibition of growth rate was detected in the intermediate treatments of 1860 μg L^{-1} and 5960 μg L^{-1} Mn. The results demonstrate that *Chlorella* sp is very tolerant to Mn exposure.

The results from this pilot investigation indicate that, compared to values reported in the scientific literature, Mn toxicity was higher to two of the three local species tested. The higher toxicity may be due to the lower pH and ionic strength of the NCW diluent. Although, the green alga, *Chlorella* sp was extremely tolerant to Mn, concentrations of 1000–2000 μg L^{-1} appeared to be acutely toxic to the cladoceran, *M. macleayi*. Further characterisation of this latter species' strong threshold response would strengthen confidence in the toxicity estimates obtained by this study. The hydra, *H. viridissima*, was clearly the most sensitive species with a significant reduction in population growth rate at the lowest concentration tested (106 μg L^{-1}), with a resultant IC_{10} value of 60 μg L^{-1}. This is one of the most sensitive responses reported in the literature and warrants further investigation.

Comparison of *H. viridissima* toxicity data with environmental concentrations

Figure 2 presents a comparison of the *H. viridissima* IC_{10} value with concentrations of Mn recorded at the Magela creek downstream (ie MG009) and upstream (ie MCUS) monitoring points. Although recorded Mn concentrations at MG009 have not exceeded the IC_{10} concentration, they have approached it on several occasions, whereas Mn upstream of Ranger remains mostly below 25 µg L^{-1}. The highest concentrations of Mn occur at MG009 during the late wet season/early dry season recessional flow period, and appear to be at least partially due to the surface expression of mine-impacted groundwaters. This period of the annual seasonal cycle coincides with the time of year when the maximum number of aquatic biota have recolonised the creek channels and, therefore, can be potentially exposed to contaminants. Although pH in Magela Creek is typically about one unit higher than Ngarradj Creek, which may reduce the toxicity of Mn, the results still suggest that a closer look at Mn toxicity in Magela Creek water is warranted.

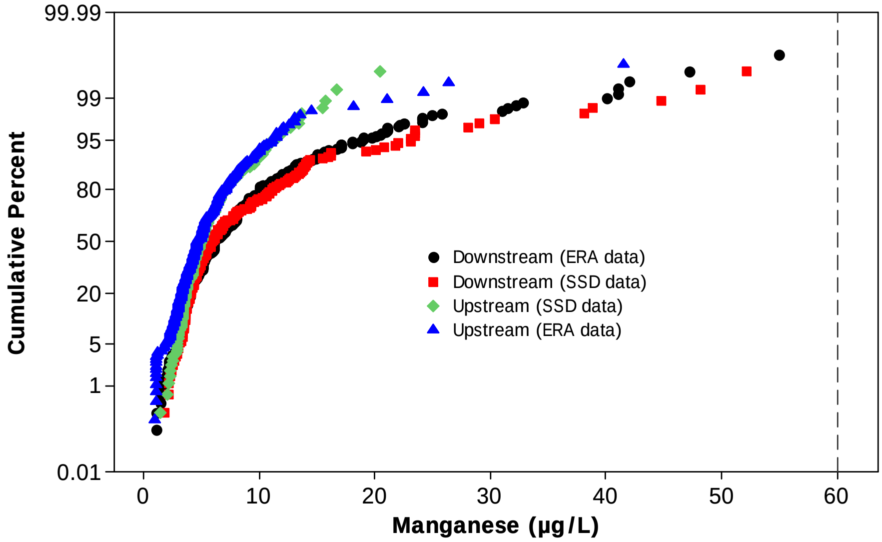

Figure 2 Cumulative frequency distributions for filtered (<0.45 µm) manganese concentrations at the Magela Creek downstream monitoring point (MG009) based on ERA data (June 1980 to June 2008) and SSD data (November 2000 to June 2008) and at the upstream monitoring point (MCUS) based on ERA data (December 1993 to June 2008) and SSD data (November 2000 to June 2008). The reference line represents the manganese IC_{10} value for *Hydra viridissima*.

Recommendations for further work

Based on the results of this study, the following additional studies are recommended:

i Assess the toxicity of Mn in Magela Creek water (pH ~6–6.5) to *H. viridissima* and *M. macleayi* to determine if it is similar to that observed in Ngarradj water;

ii Assess the toxicity of Mn in Magela Creek water to four additional species from the standard suite of test species and derive a site-specific water quality trigger value/Limit.

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Screening level ecotoxicological assessment of treated pond water from Ranger uranium mine to five local freshwater species

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Background

Ecotoxicological testing of Ranger mine treated pond water from a trial portable treatment unit (that treated water using microfiltration and reverse osmosis alone) was requested by Energy Resources of Australia Ltd (ERA) in July 2007. This followed an ecotoxicological assessment in late 2005 on samples of treated pond water from a permanent treatment plant. In the 2005 study, treated pond water permeate was found to be non toxic to four of the five species tested. The affected species, the water flea, *Moinodaphnia macleayi*, was found to respond to the 67 and 100% concentrations of permeate, resulting in ~ 25 and 50% reductions in reproduction, respectively (van Dam et al 2007). There was some uncertainty regarding the cause of this toxicity, as uranium (U), at 4 µg L^{-1}, was the only contaminant measured above detection limits and historical data indicated that *M. macleayi* is only sensitive to concentrations above 18 µg/L U (***eriss*** unpublished data; Semaan et al 2001). As such, two hypotheses were proposed to explain the effect on *M. macleayi*:

3 Enhanced U toxicity due to a lack of dissolved organic carbon in the treated water; and

4 Nutrient/essential ion deficiency due to the minerally deficient nature of the treated water.

An experiment was undertaken during 2006-2007 to address these hypotheses, where the toxicity of U to *M. macleayi* was tested concurrently in (i) 'synthetic' Magela Creek water (SMCW), which simulates the inorganic composition of the creek water but contains no DOC; and (ii) natural Magela Creek water (van Dam et al 2008). The results indicated that not only was *M. macleayi* reproduction impaired in SMCW but that sensitivity to U was at least three fold higher than the response of the water fleas in NMCW. As such, further testing was required to further elucidate whether either of the above factors, or a combination of both (ie. enhanced sensitivity of *M. macleayi* as a result of the stress also imposed by nutrient/essential ion deficiency) was the primary contributor to the observed toxicity. As such, this research summary not only describes the screening level testing undertaken on treated pond waters from the portable treatment unit, but also presents the results of further exploratory testing to determine the cause of observed toxicity to *M. macleayi*.

Methods

A similar approach to that undertaken in late 2005 was used in this study. In late July 2007, five local freshwater species, a unicellular alga (*Chlorella* sp), macrophyte (duckweed; *Lemna aequinoctialis*), cnidarian (*Hydra viridissima*), crustacean (water flea, *Moinodaphnia macleayi*) and a fish species (*Mogurnda mogurnda*) were exposed to treated pond water permeate and a natural Magela Creek water (NMCW) control. Physical and chemical parameters of the water indicated that the expected toxicity would be very low (U = 7.7 µg L^{-1}, pH = 6.43, EC = 19 µS cm^{-1}). Therefore, it was considered unnecessary to expose the organisms to a typical series of 5–6 dilutions. Instead, a much reduced testing regime was employed where all species except *M. macleayi* were exposed only to 100% treated pond water, with *M. macleayi* exposed to 25, 50 and 100% treated pond water permeate (in addition to the NMCW control). Testing methods followed those described by Riethmuller et al (2003).

A further experiment was undertaken in order to investigate the cause of the observed toxicity to *M. macleayi*, and more specifically, to understand the role of dissolved organic carbon on the response of the water fleas in NMCW. To do this, the acute toxicity (48-h survival) of U to *M. macleayi* was assessed in three water types: (i) SMCW alone; (ii) SMCW + 3.8 mg L^{-1} DOC (as IHSS Suwannee River Fulvic Acid Standard 1); and (iii) natural Magela Creek water alone (which had a measured DOC concentration of 3.7 mg L^{-1}). Acute toxicity was assessed because the test animals are not fed during the 48-h survival test, and hence there is no additional organic component to potentially confound the results. The main aim of the experiment was to see whether the addition of DOC to SMCW would change the sensitivity of *M. macleayi* to U compared to in SMCW alone, and how this would compare to NMCW with a similar natural DOC concentration.

Results and discussion

The effects of treated pond water permeate on the five species tested are reported in Table 1. A full report on the study is provided by Harford et al (2008). The permeate had no significant effect (P>0.05) on *Chlorella* sp, *L. aequinoctialis* and *M. mogurnda*. However, exposure to 100% permeate resulted in statistically significant reductions in the growth of *H. viridissima* (12% reduction relative to control growth; P<0.05) and the reproductive success of *M. macleayi* (53% reduction relative to control reproduction; P<0.05). *M. macleayi* was unaffected by exposure to 25 or 50% permeate (P>0.05). The IC_{10} and IC_{50} (95% confidence limits; CLs) of pond water permeate to *M. macleayi* were 54 (19–63)% and 97 (CLs not calculable)%, respectively.

Table 1 Effect of pond water permeate on five local freshwater species

Test organism	Test duration & endpoint (metric)	% permeate	Mean ± SEM
Chlorella sp (unicellular alga)	72 h cell division rate (doublings/day)	0	1.41 ± 0.02
		100	1.19 ± 0.11
Lemna aequinoctialis (duckweed)	96 h plant growth (growth rate)	0	0.492 ± 0.005
		100	0.482 ± 0.013
Moinodaphnia macleayi (cladoceran)	3 brood (5-6 d) reproduction (number of neonates)	0	35.7 ± 3.1
		25	35.4 ± 1.8
		50	33.5 ± 3.0
		100	16.7 ± 2.9*
Hydra viridissima (green hydra)	96 h population growth (growth rate)	0	0.402 ± 0.005
		100	0.355 ± 0.002*
Mogurnda mogurnda (fish)	96 h survival (percentage surviving)	0	100 ± 0
		100	100 ± 0

* Significantly different from the control (P < 0.05)

The results of the current study were generally consistent with those from the 2005 study where pond water permeate exhibited no or low ‘toxicity’ to the five species assessed, as follows:

- 100% pond water permeate had no adverse effects on the green alga, *Chlorella* sp, the duckweed, *Lemna aequinoctialis*, and the purple-spotted gudgeon, *Mogurnda mogurnda*;
- A 12% reduction in growth rate of the green hydra, *Hydra viridissima*, was observed at 100% pond water permeate; and
- A 53% reduction in reproduction of the cladoceran, *Moinodaphnia macleayi*, occurred at 100% pond water permeate.

The minor response of *H. viridissima* was most most likely attributable to the nutrient/mineral deficient nature of the pond water permeate. The U concentration was at least five-fold lower than that known to exhibit low toxic effects to *H. viridissima* in organic-deficient synthetic Magela Creek water (ie >40 µg L^{-1}), which is considered a reasonable analogue of pond water permeate. Hence, U toxicity is highly unlikely to have contributed to the response of *H. viridissima*.

The larger response of *M. macleayi* also was most likely attributable to the nutrient/mineral deficient nature of the pond water permeate. This was supported by existing data that showed an approximate 50% reduction in reproduction of *M. macleayi* in synthetic Magela Creek water compared to natural Magela Creek water. Uranium is unlikely to have been a causative factor for the following reasons. Firstly, the pond water permeate U concentration was lower than that previously reported to exhibit low toxic effects to *M. macleayi* in synthetic Magela Creek water (ie ~11 µg/L, for reproductive test endpoint with feeding). Secondly, because of an unexpected and unacceptable increase (from ~6.4 to ~8.2) in the pH of the test water during the test, the the bioavailability of U would have been greatly reduced owing to changes in solution speciation.

More broadly, however, the range of concentrations reported for pond water permeate from Ranger’s existing on-site water treatment plant overlap with those known to be toxic to *M. macleayi* in SMCW. Consequently, it is important that the cause(s) of adverse effects exhibited by *M. macleayi* exposed to pond water permeate are further elucidated.

The results of the 48 h acute flea experiment are shown in Figure 1. Control survival was >90% in all three water types. The addition of 3.8 mg L^{-1} DOC to SMCW resulted in an 8–9 fold reduction in the toxicity of U to *M. macleayi* (based on EC_{50} values). Uranium was approximately 4 times less toxic in NMCW (3.7 mg L^{-1} DOC) compared to SMCW + DOC. Notwithstanding the potential influence of a nutrient/mineral deficiency, the difference in toxicity of U between NMCW and SMCW + DOC could be attributed to a difference in the DOC constitutents between the two water types. There is some concern that extraction procedures may result in chemical and structural alterations of fulvic acids which may alter these standards from being representative of humic substances in their natural state (Aiken & Malcolm 1981). Moreover, the difference in the ability of NMCW DOC to complex U may be due to it containing a mixture of humic substances (both humic and fulvic acids). Fulvic acids are known to have greater complexing ability than humic acids (Tipping 2002). In addition, natural waters also contain other constituents that may combine with each other in simultaneous and competitive reactions. The amelioration of metal toxicity by DOC is well documented for many metals (Tipping 2002), and the effect of DOC on U toxicity is under continued investigation in our laboratory (see summary above).

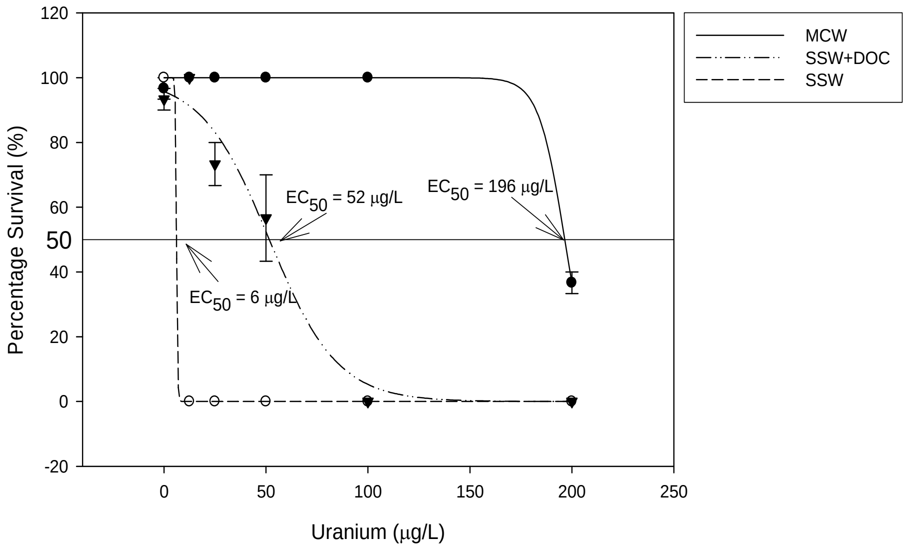

Figure 1 Acute toxicity of uranium to *Moinodaphnia macleayi* (48-h survival) in three water types: (i) synthetic Magela Creek water (SMCW) only; (ii) SMCW + 3.8 mg/L DOC (as IHSS Suwannee River Fulvic Acid Standard 1); and (iii) natural Magela Creek water (NMCW; containing 3.7 mg/L natural DOC)

Steps to completion

The issue of poor pH control in low ionic strength test solutions such as the treated pond waters needs to be addressed. Laboratory tests have shown that the food provided to the water fleas is primarily responsible for the increase in pH that occurs with the use of poorly buffered water (***eriss*** unpublished data). Laboratory staff are currently investigating the viability of reducing the food ration for this test protocol with the aim of reducing pH drift during testing.

As the potential for U to be toxic in treated pond water is not yet fully understood, additional acute and chronic tests using *M. macleayi* could be undertaken where a high quality treated pond water sample (ie one with starting U concentration < 1 µg L^{-1}) is tested after it has been spiked with additional U at various increasing concentrations. Such a study would be relevant and informative to the overall issue and associated management of treated pond waters, but is not of high priority at present.

Although the testing of treated pond water has been completed, the treatment plant is yet to be commissioned for mine process water, at which time the process water permeate containing ammonium ion will require ecotoxicological assessment. This work has been delayed due to delays in commissioning the plant, with the latest information indicating that testing is unlikely to be required until 2009.

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

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

Background

Over the past four years, and in response to recommendations by van Dam (2004) and also by Dr Jenny Stauber at ARRTC's 14th meeting, the ***eriss*** ecotoxicology laboratory has been implementing a program of reference toxicant testing, using uranium, for its routine testing species. Since 2004–05, reference toxicant control charts have been developed for four of the five routine testing species. The aims for 2007–08 were to:

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

2 continue to investigate problems with the *Lemna aequinoctialis* reference toxicity test.

Progress

In total, 20 reference toxicants tests (*Chlorella* – 5; *Hydra* – 4; *Moinodaphnia* – 4; and *Mogurnda* – 7) were completed during 2007–08, 18 of which provided valid results. The results of these tests are summarised in Table 1, with the control charts presented in Figure 1.

Table 1 Summary of uranium reference toxicity test results for 2007–08

Species & endpoint	Test Code	IC_{50} (µg/L)	Valid test?	Comments
Chlorella sp	893G	61 (29, 77)	Yes	
(72-h cell division rate)	898G	NC[b]	No	Culture 10 days old at start of test[b]
	899G	37 (32, 42)	Yes	
	942G	34 (22, 70)	Yes	
	941G	35 (25, 45)	Yes	
Moinodaphnia macleayi	855I	50 (41, 62)	Yes	
(48-h immobilisation)	870I	14(12, 16)	Yes	
	912I	9 (83, 10)	Yes	
	929I	16 (14, 19)	Yes	
Hydra viridissima	834B	84 (73, 96)	Yes	
(96-h population growth)	879B	66 (89, 138)	Yes	
	886B	65 (56, 72)	Yes	
	942B	NC[a]	No	No effect at highest conc[b]
Mogurnda mogurnda	859E	1383 (NC)	Yes	
(96-h sac fry survival)	869E	1166 (1109, 1225)	Yes	
	872E	1777 (1609, 1962)	Yes	
	875E	1428 (1057, 1576)	Yes	
	880E	1458 (1338, 1590)	Yes	
	896E	1226 (1057, 1422)	Yes	
	928E	1262 (1131, 1366)	Yes	

[a] NC: Not calculable. See 'comments' column for reason

[b] See text for discussion

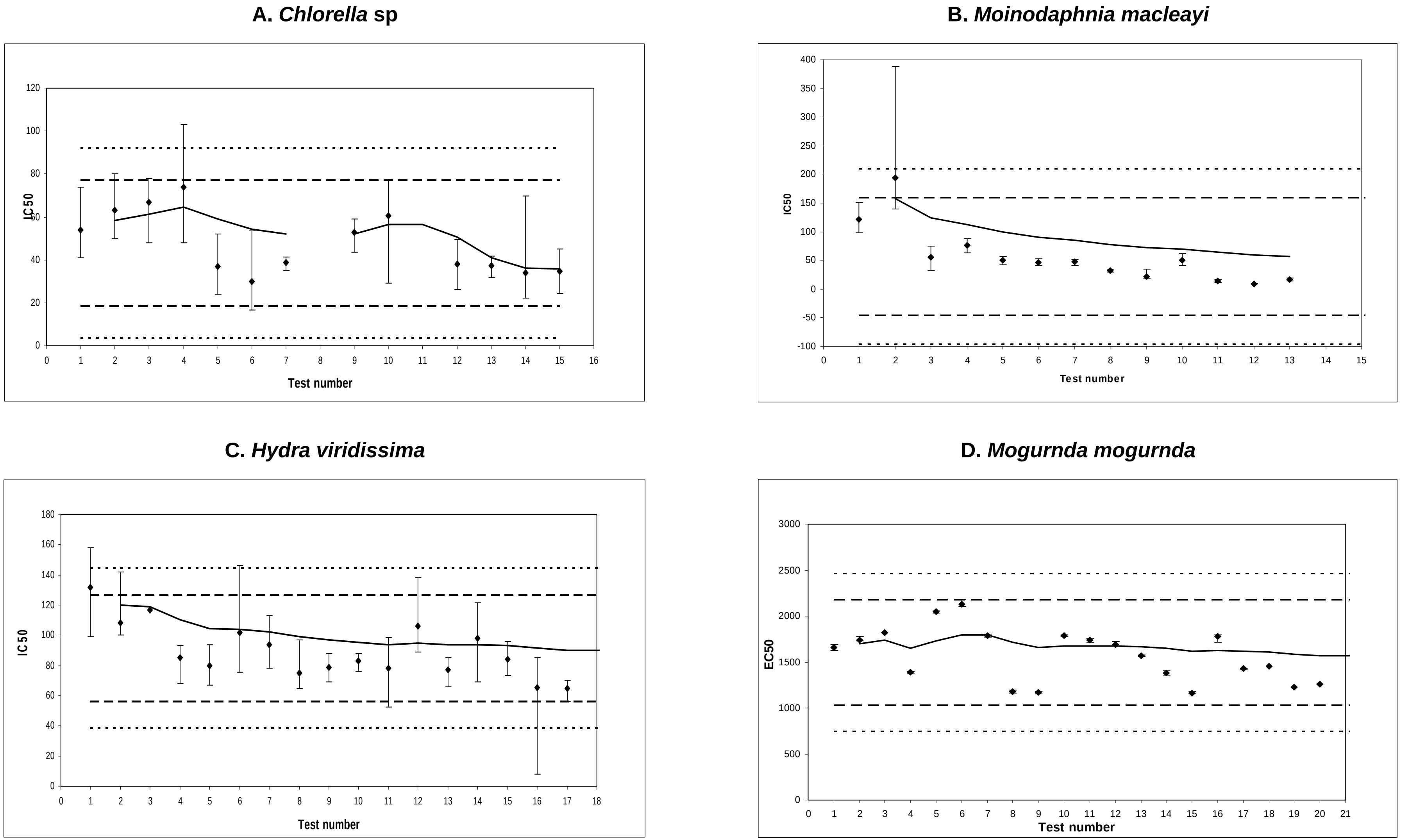

Figure 1 Reference toxicant control charts for A. *Chlorella* sp, B. *M. macleayi*, C. *H. viridissima* and D. *M. mogurnda*, as at September 2008. Data points represent IC_{50} or EC_{50} toxicity estimates. Reference lines represent the following: dotted lines – upper and lower 99% confidence limits; broken lines – upper and lower warning limits; unbroken line – running mean.

A brief summary of the major issues for this program is provided below.

Chlorella sp

Over the course of the year algal toxicity testing results were valid with the exception of one test, where a 10-d old culture was used inadvertently instead of a 4 to 5-d old culture.

H. viridissima

Three of four reference toxicity tests for *H. viridissima* were valid. Test 942B showed no effect at the highest concentration, therefore an IC_{50} could not be established. Hydra in this test appeared less sensitive than the previous three tests. Chemistry results for all tests were within acceptable ranges.

M. macleayi

M. macleayi reference toxicity testing was initiated for the first time in 2006–07. Few problems were observed with this test, with all tests being valid. The three tests conducted in 2007-08 were also valid. However, there has been a trend where the IC_{50} appears to be decreasing over time (See Figure 1B), indicating that the *M. macleayi* stock may be becoming more sensitive. This will be closely monitored during 2008-09 to determine if this is the case. If necessary, an investigation will be initiated to assess whether the current laboratory stock of *M. macleayi* differs in its sensitivity to U from ‘wild’ *M. macleayi*. If so, a new stock of *M. macleayi* will most likely be sourced.

M. mogurnda

All seven reference toxicity tests for *M. mogurnda* were valid. There are no problems associated with this protocol.

Reference toxicity test development for L. aequinoctialis

The development of the *L. aequinoctialis* reference toxicity test continues to be problematic. The challenge has been optimising the test medium so as to enable adequate control growth and also an effect to be observed at uranium concentrations that are not excessively high. Unfortunately, nutrients (eg PO_4) and trace elements added to the test medium to facilitate plant growth also interact with uranium, greatly reducing its bioavailability and toxicity. In 2006–07, numerous experiments were done, assessing different diluent water types and nutrient additions (eg SSW + NO_3 and PO_4; MilliQ water + various concentrations of CAAC medium). Despite some promising indications, consistent results could not be obtained.

A potential solution to this problem, which needs further discussion, is to lower the minimum acceptability criterion for control plant growth (eg. from a four-fold increase in frond numbers to a three-fold increase in frond numbers). A trial reference toxicity test (831 L) was conducted in 2007–08, using 0.5% CAAC medium as the diluent/control water. Due to insufficient control growth (ie increase in frond number of 41, compared to the minimum acceptability criterion of 48) the test was deemed invalid. However, if the minimum acceptability criterion for control plant growth was lowered as suggested above, the test would have been valid. The resultant IC_{50} (95% CLs) from the test (based on nominal U concentrations) was 1591 (894–2812) µg/L. However, reflecting the previously observed inconsistency in plant growth, a growth trial using 0.5% CAAC medium conducted in

September 2008 resulted in an increase in frond number of 33, which does not meet either the existing or proposed lower minimum acceptability criterion for growth.

Planned testing in 2008–09

The reference toxicity testing programs for *Chlorella* sp, *M. macleayi*, *H. viridissima* and *M. mogurnda* will continue in 2008–09, with the aim of completing at least four tests per species. Further testing will take place to try to resolve the issues of the *L. aequinoctialis* reference toxicity test protocol. This will involve optimisation of the test medium so as to enable adequate control growth and also a concentration-response relationship to be observed at uranium concentrations that are not execessively high. In addition to the reference toxicity test protocol, there is a need to develop an additional endpoint, based on plant surface area or dry weight. The development of this endpoint will be done in conjunction with the reference toxicity test development work.

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Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Esparon & A Bollhöfer

Introduction

The recommended dose limit to the public from a practice of 1 milli Sievert (mSv) per year applies to the sum of all pathways and relevant practices that people could potentially be exposed to. However, the ICRP (1997) states in paragraph 6.2.1 that:

> To allow for exposures to multiple sources, the maximum value of the constraint used in the optimisation of protection for a single source should be less than 1 mSv in a year. A value of no more than about 0.3 mSv in a year would be appropriate.

Consequently, a dose constraint of 0.3 mSv should be applied when assessing radiological monitoring data for the Ranger mine. As the inhalation pathway has previously been identified as the main contributor to public dose from the mine site for an adult living in Jabiru and working in Jabiru East during the operational phase (Martin 2000), both ERA and SSD monitor the two airborne exposure pathways in the region. The two potential pathways are radioactivity trapped in or on dust (or long lived alpha activity, LLAA) and radon decay products (RDP).

Of these two airborne pathways, RDP accounts for most of the dose received by the public. Since the main areas of habitation in the vicinity of Ranger and Jabiluka are Jabiru, Mudginberri and Jabiru East, the SSD monitoring program focuses on those three population centres as shown by the black triangles in Map 3. RDP and LLAA concentrations in the air are measured monthly and the results are periodically compared with those from ERA's atmospheric radiological monitoring program.

Results

Radon pathway

Figure 2 shows the quarterly RDP data from Jabiru, Jabiru East and Mudginberri measured by ***eriss*** from mid 2003 to December 2008. Two new Environmental Radon Decay Product Monitors (ERDM) have been purchased from Radiation Detection Systems, Adelaide, and have replaced the old RDP monitor (alphaprismII) from July 2008. The ERDMs have the ability to log data continuously. They will be fitted with solar panels and tested in Darwin, before being deployed continuously at Jabiru Water Tower and Mudginberri, respectively.

Median RDP concentrations [$\mu J/m^3$] for 2003-2008 at Jabiru, Jabiru East and Mudginberri are 0.043, 0.069 and 0.038, respectively. The values from Jabiru East are generally higher and show more variation due to the closer proximity of Jabiru East to the mine pit and ore stockpiles, the largest localised sources of radon in the area.

In Jabiru, most of the mine origin radon has dispersed, and variations in concentrations are mainly caused by diurnal variations, and the annual cycle of wet and dry seasons Airborne radon concentrations are generally lower during the wet season, as radon exhalation from the soil decreases with increasing soil moisture content. The influence of other factors such as soil ^{226}Ra activity concentration, soil morphology, and vegetation cover have been investigated and the results from this study have now been published (Lawrence et al 2009).

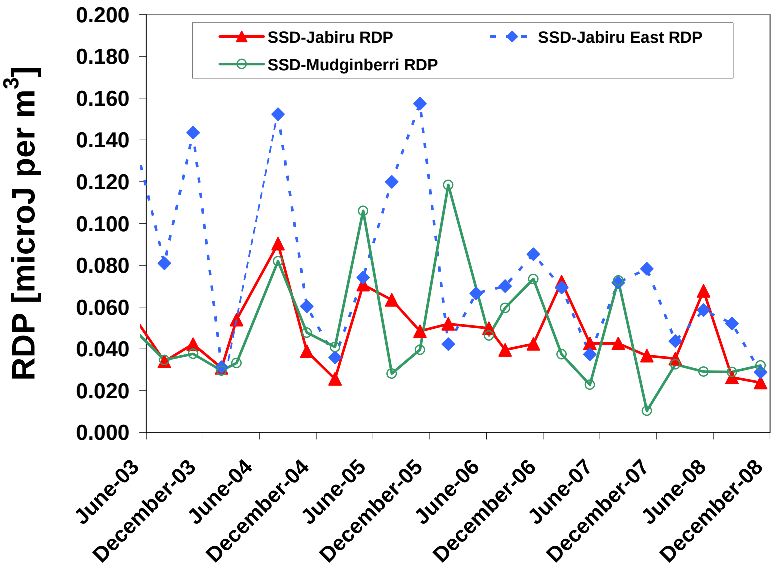

Figure 2 Radon decay product concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

Since the exposure due to naturally occurring RDP in the region is about 1 mSv per year, one of the challenges of determining the mine-related dose due to the inhalation of RDP has been distinguishing between the mine-derived and natural background signal. ERA estimates the mine origin RDP using a wind correlation model and calculates the exposure via the radon pathway.

Table 1 shows the annual averages for the radon decay product concentrations measured by ***eriss***, and reported by ERA, at Jabiru and Jabiru East, and the calculated total annual doses from RDP inhalation. This is assuming an occupancy of 8760 hrs (1 year) and a dose conversion factor for the public of 0.0011 milli Sievert (mSv) per µJ/hr/m^3. In 2007, ERA reported that there was no significant difference between RDP concentration in wind blowing from the mine and the environmental sector, respectively, at Jabiru. In other years, the reported mine related dose from the inhalation of radon progeny is generally low and generally amounts to less than 10 per cent of the public dose constraint of 0.3 mSv per year from a single source

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

		2006	**2007**	**2008**
RDP concentration [µJ/m³]	Jabiru East	0.066 (0.071)	0.064 (0.059)	0.046 (N/A)
	Jabiru	0.046 (0.039)	0.049 (0.038)	0.038 (N/A)
	Mudginberri	0.075	0.036	0.031
Total annual dose [mSv] Jabiru		0.44 (0.38)	0.47 (0.37)	0.37 (N/A)
Mine derived dose [mSv] at Jabiru[a]		0.003	0	N/A

[a] predicted from wind field model

Dust pathway

Atmospheric dust activity concentration, or long lived alpha activity (LLAA) concentration, is routinely monitored by both, ***eriss*** and ERA at the monitoring sites displayed in Figure 1. Figure 3 shows the long lived alpha activity at Jabiru, Jabiru East and Mudginberri measured by ***eriss*** from mid 2003 to December 2008.

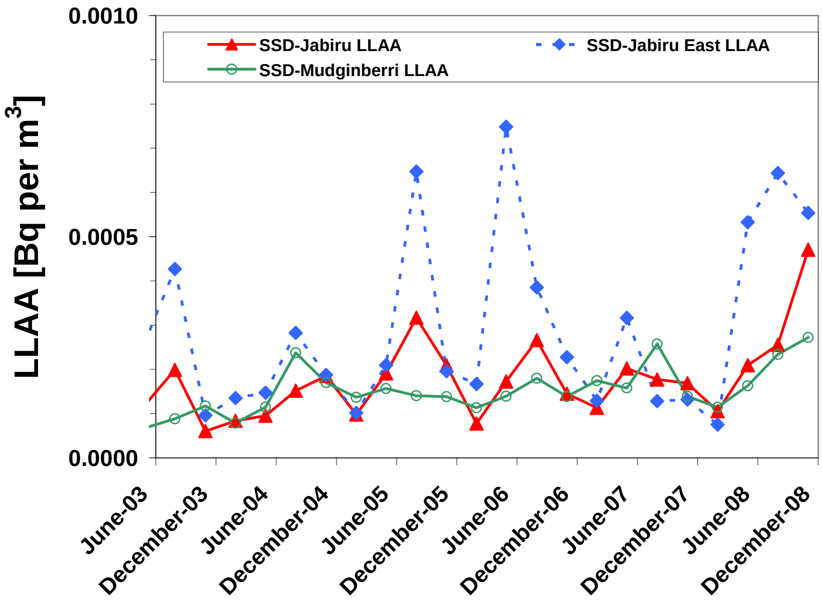

Figure 3 Long lived alpha activity concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

Similar to the atmospheric radon concentration, the dust concentration is lower during the wet season due to the higher soil moisture content that suppresses dust generation. Generally, LLAA concentration is higher at Jabiru East due to its proximity to the mine. The average values measured from mid 2003 to December 2008 at Jabiru, Jabiru East and Mudginberri are 0.00017, 0.00028 and 0.00015 Bq/m^3, respectively.

The total annual dose from inhalation of dust was calculated using a dose conversion factor for the inhalation of dust of 0.0057 mSv per alpha decay per second (Zapantis 2001), and a breathing rate of 7300 m^3 per year for adults (UNSCEAR 2000). This gives a total dose range of 6-11 μSv at Mudginberri, Jabiru and Jabiru East for 2008. Only a fraction of that dose would be mine-related (Bollhöfer et al 2006).

Steps for completion

The routine monitoring of dust and radon progeny will continue at, Jabiru, Mudginberri Four Gates Road Radon Station and Jabiru East. Continuous RDP monitors have been acquired and tested, and will be permanently deployed at the Jabiru and Four Gates Road radon stations early in 2009.

Summary

Monitoring of radon and dust exposure pathways over the past 5 years has shown that the only significant contribution to radiological exposure of the public at Jabiru via inhalation is the inhalation of radon decay products. Although the contribution from the mine site has been shown consistently to be much less than the public dose constraint of 0.3 mSv per year and is of no concern according to current best practice standards, atmospheric monitoring will be continued to provide re-assurance to the public that the inhalation of mine derived radionuclides remains low.

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Monitoring of radionuclides in groundwater at Ranger

B Ryan

Introduction

Groundwater samples collected by the Northern Territory Department of Regional Development, Primary Industry, Fisheries and Resources (DRDPIFR) from Ranger mine continue to be analysed by ***eriss*** for radionuclides and a suite of dissolved metals. This groundwater monitoring program aims to investigate temporal and spatial variability of uranium and radium in groundwater at the mine and contribute to providing some insights into aspects of groundwater contaminant transfer, origin and the associated processes. The information can also be used to help validate contaminate transport models that are currently being developed by Ranger and EWL Sciences (EWLS).

As part of an effort to improve the Ranger groundwater knowledge base and to progress the development of closure criteria for Ranger, a joint organisational approach is being put in place to help facilitate a more coordinated research approach. The current groundwater sampling and analysis work being conducted by ***eriss***, EWLS and DRDPIFR will be reviewed and a collaborative study instigated.

Results

The groundwater radionuclide and dissolved metal data that have been collected by ***eriss*** on and around the mine site over the last 25 years have been reviewed and converted from old database formats, reports and papers and entered into Excel spreadsheets. The results have gone through a quality assessment process and rated according to the quality of the data. The results for each bore have then been put into a time series with some bores having a continuous series of data of more than 15 years. There are other bores that have incomplete time series data due to discontinued use because of ongoing mine operations that include waste dumping and covering of the bores. The locations of groundwater bores are shown in Figures 1 and 2. Entry of the location and basic water quality data from the current DRDPIFR and EWLS/Ranger sampling programs into a GIS has started. A groundwater database is also being developed by ***eriss*** with the view to migrating all ***eriss*** groundwater data into it. This database would contain all historical physical and spatial information on the groundwater bores ***eriss*** has collected and allow ready access to the data.

Steps for completion

Several meetings have been held with DRDPIFR to discuss each organisation's sampling program, historic data, process for data review and knowledge gap identification. With groundwater database developments continuing for both organisations, compatibility issues are being addressed to have a seamless exchange of data between the two groups.

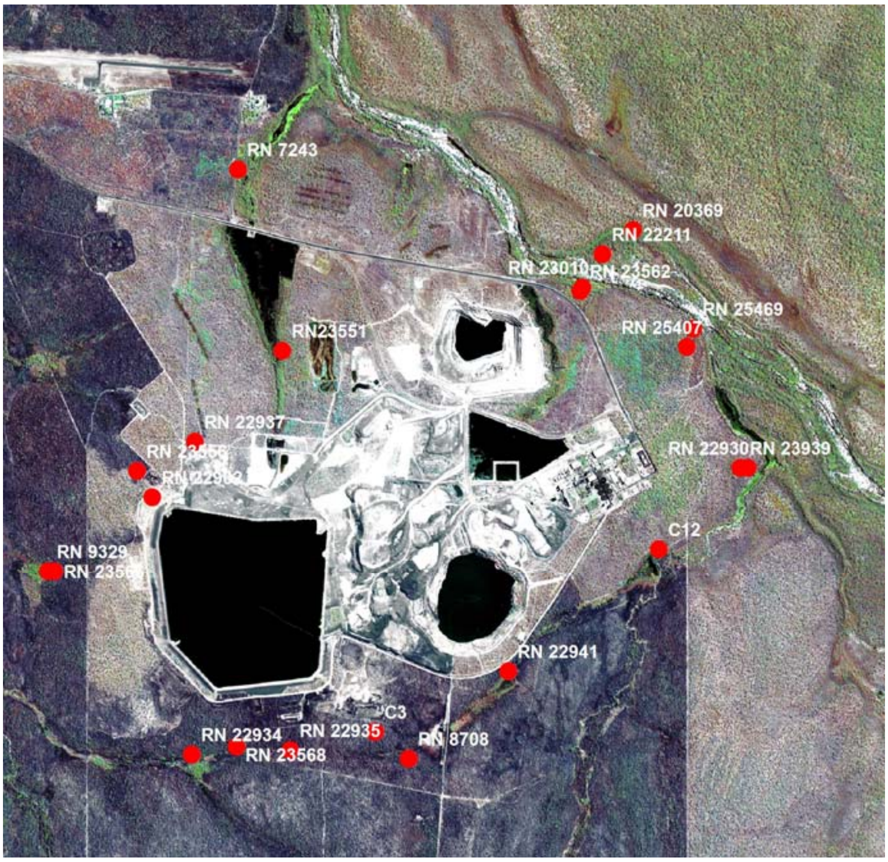

Figure 1 DRDPIFR monitoring bore sampling sites

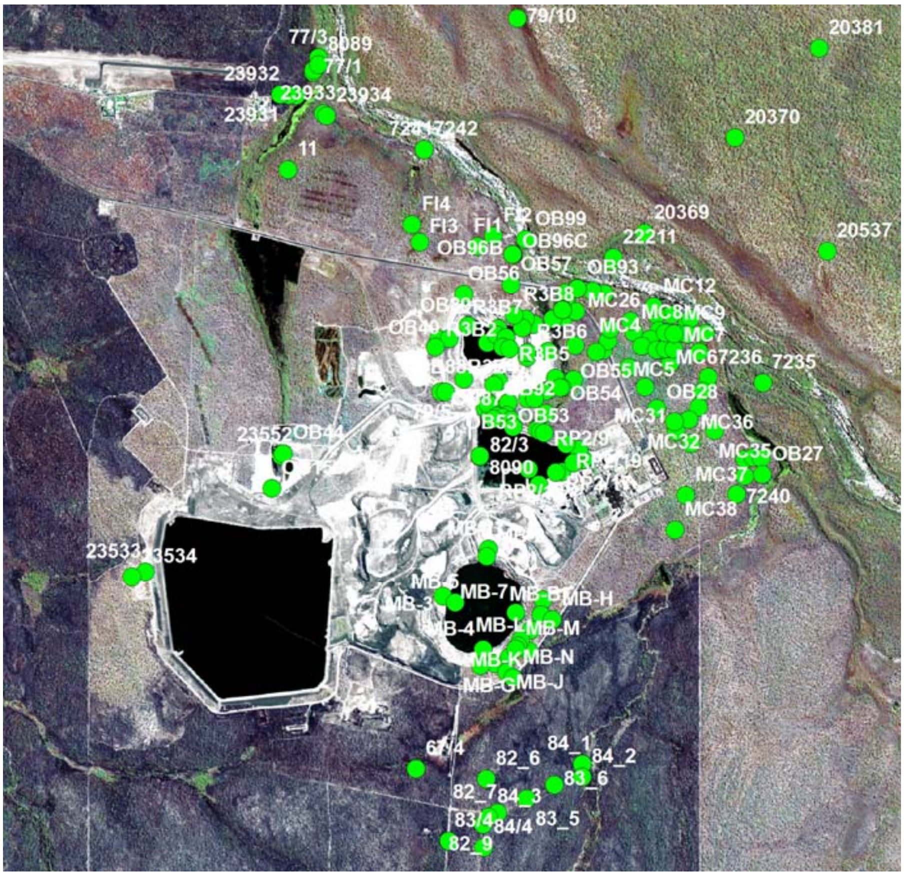

Figure 2 Ranger uranium mine monitoring bore sites associated with Pit 3

The aim for this financial year is the establishment of comprehensive groundwater quality datasets, with agreement reached on data quality and statistical assessments of available data to identify any changes in groundwater quality over the time monitored. The information gained will form the basis of recommendations for possible changes to the scope and extent of far-field groundwater quality monitoring at Ranger. The information will also be used in the assessment of current monitoring programs for ***eriss*** and DRDPIFR and help streamline and identify any deficiencies in these programs.

Acknowledgments

The Northern Territory Department of Regional Development, Primary Industry, Fisheries and Resources is acknowledged for collection of the bore water samples and providing the aliquots for analysis.

Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

P Medley, A Bollhöfer & J Brazier

Introduction

Surface water samples in the vicinity of the Ranger and Jabiluka project areas are regularly measured for their radium-226 (^{226}Ra) activity concentrations to check for any significant increase in ^{226}Ra levels downstream of the impacted areas. This is due to the potential risk of increased exposure to radiation via the biophysical pathway due to mining activities. Mussels particularly bioaccumulate ^{226}Ra which may then be incorporated into the human body upon consumption.

Water samples are collected weekly in Magela Creek (Ranger) from both upstream and downstream sites, and monthly from the Ngarradj Creek (Jabiluka) downstream site. Samples are not collected from these locations during periods of no contiguous surface water flow (ie during the dry season).

All Ngarradj Creek samples are analysed for total ^{226}Ra (ie dissolved and particulate phases combined) by ***eriss***'s environmental radioactivity laboratory using a method described in Medley et al (2005).

Before the 2006–07 wet season the OSS had decided to combine weekly samples obtained from Magela Creek to give monthly averages. Analyses of the complete data set and combined wet season samples from previous years has shown that combining weekly samples will provide valid monthly radium results, but reduce the costs for radioanalysis significantly.

Progress to date

Magela Creek

The ^{226}Ra activity concentration data are compared to previous wet seasons in Figure 1.

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

A limit of 10 mBq/L increase above natural (upstream) background in total ^{226}Ra concentration in surface waters downstream of Ranger has been defined for human radiological protection purposes (Klessa 2001) and is based on the potential dose received from the ingestion of ^{226}Ra in the freshwater mussel *Velesunio angasi* (Martin et al 1998). Each wet season the difference value is calculated by substracting the upstream median from the downstream median (Sauerland et al 2005). This difference is called the wet season median difference (shown by the solid green lines in Figures 1 and 2) and should not be more than the limit of 10 mBq/L. The wet season median difference for the entire 2001–08 wet season data set is approximately zero ('*All years*' column Table 1). The data for the seven sampling seasons indicate that ^{226}Ra levels in Magela Creek are due to the natural occurence of radium in the environment (upstream data set) and that ^{226}Ra activity concentrations in Magela Creek water are not elevated (wet season median difference of zero) downstream of Ranger uranium mine.

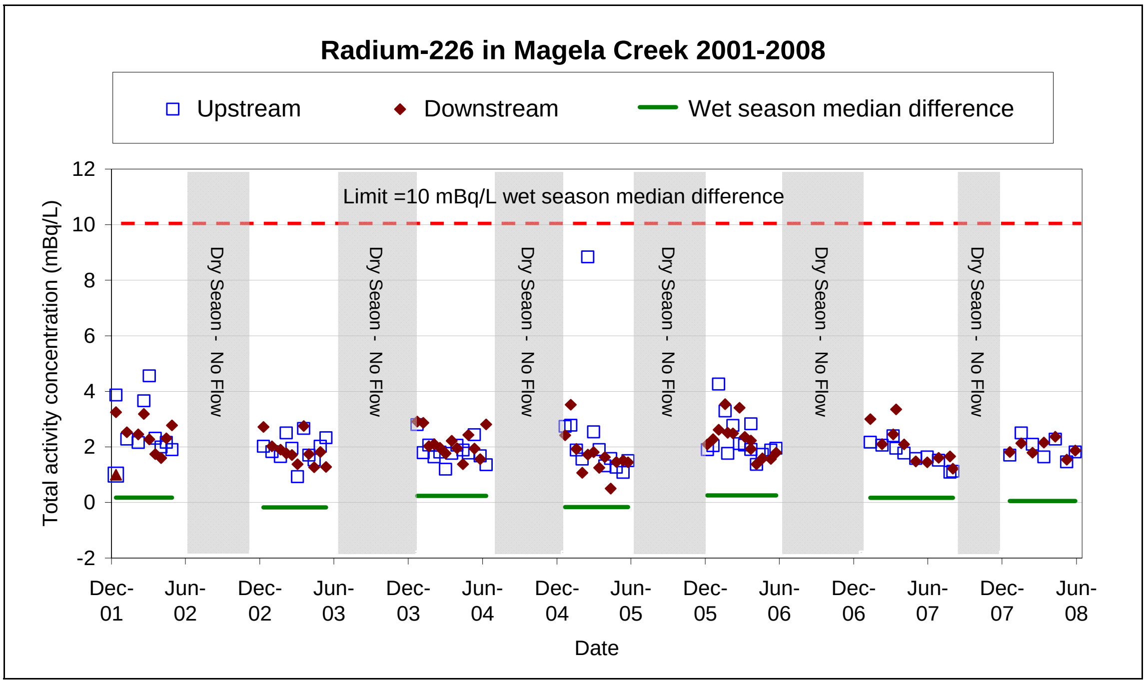

Figure 1 Radium-226 in Magela Creek for the 2001–08 wet seasons

Table 1 Median and standard deviations of the ^{226}Ra activity concentration for individual wet seasons (2003–08) and for the entire study period (2001–08)

Statistics for total ^{226}Ra activity concentrations [mBq/L]								
Magela Creek		All years 2001-08	2002–03	2003–04	2004–05	2005–06	2006–07	2007–08
Median and standard deviation	upstream	2.0 (± 1.1)	2.0 (± 0.5)	1.8 (± 0.4)	1.7 (± 2.1)	2.0 (± 0.8)	1.7 (± 0.4)	1.8 (± 0.4)
	downstream	2.0 (± 0.6)	1.8 (± 0.5)	2.0 (± 0.5)	1.6 (± 0.7)	2.3 (± 0.7)	1.9 (± 0.7)	1.9 (± 0.3)
Wet season median difference		0.0	0.0	0.2	- 0.2	0.1	0.1	0.1
Ngarradj								
Median and standard deviation	upstream	1.2 (± 0.5)	1.4 (0.6)	1.1 (± 0.4)	1.3 (± 0.3)	1.0 (± 0.4)	1.2 (± 0.4)	N/A
	downstream	1.1 (± 1.8)	1.1 (1.5)	0.9 (± 0.9)	1.0 (± 0.6)	0.5 (± 0.5)	1.0 (± 0.3)	1.0 (± 0.4)
Wet season median difference		0.0	- 0.1	- 0.3	- 0.1	0.0	0.1	N/A

Ngarradj Creek

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

Since monitoring data from 2003 onwards have shown that there has been no significant difference between upstream and downstream values, and moreover since the absolute values are in any case very low and barely above detection limit , monitoring at the upstream site has been discontinued while Jabiluka remains in long-term care and maintenance. From the 2007–08 wet season onwards, comparisons of the downstream data will be made with previous season's data for this location to check that there are no significant upward deviations from this control record.

^{226}Ra results (monthly samples) for the 2007–2008 wet season at the Ngarradj Creek downstream site are comparable to the very low values of previous years (Figure 2), indicating that the downstream environment remains unimpacted. A t-test shows that there is no statistically significant difference between the 2007–08 data and the previous wet seasons.

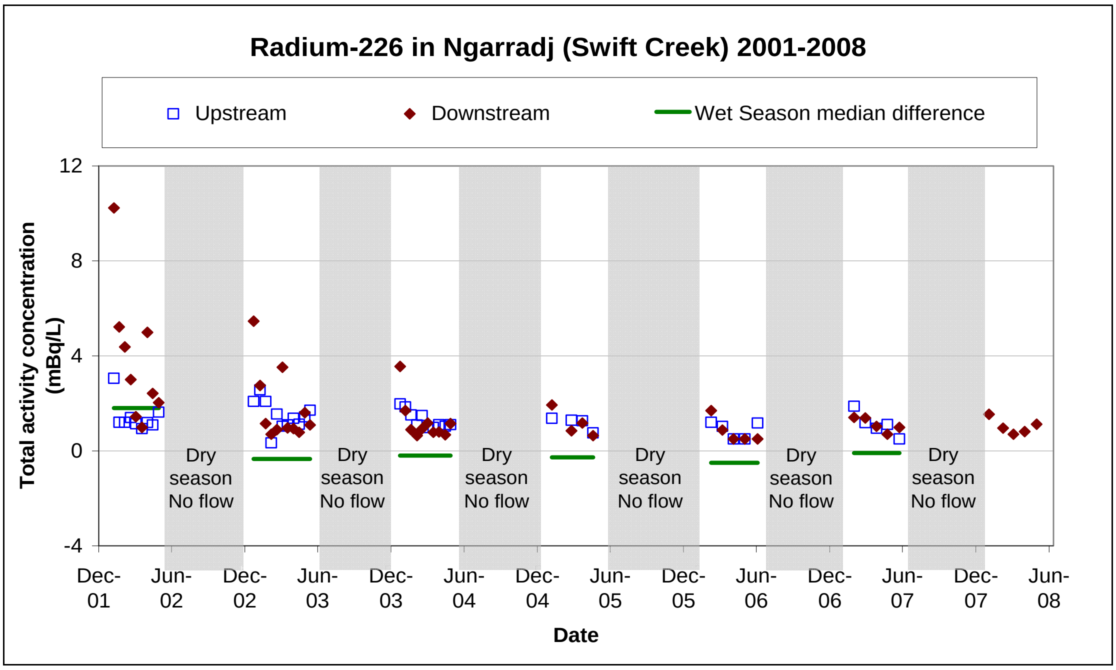

Figure 2 ^{226}Radium in Ngarradj for the 2001–08 wet seasons

Steps for completion

The ^{226}Ra monitoring in Magela and Ngarradj Creeks will be continued for the 2008–09 wet season.

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Suspended sediment, metal and radionuclide loads in Magela and Gulungul Creeks

P Medley & K Turner

Introduction

This project aims to measure the activity concentrations of uranium (U) and radium (Ra) in samples of fine suspended sediment through the wet season and at various positions on flow hydrographs, to determine if it is possible to use the continuous turbidity record to derive seasonal loads of sediment-bound U and Ra (and other trace metals) in Magela and Gulungul Creeks.

The only previous attempt to estimate the annual U load in Magela Creek was that by Hart et al (1987) and the result had a precision of ±100%. In addition there has been no attempt made to determine Ra loads. Hence it is apparent that there is a significant knowledge gap with respect to providing a reliable value for this important parameter.

Correlations between turbidity and suspended sediment concentration have been derived for Gulungul Creek (Moliere et al 2008a) and Magela Creek (Moliere et al 2008b). This project will focus on determining if it is possible to extend this further, by using continuous turbidity traces to estimate seasonal loads of U and Ra in suspended sediment. In addition, monitoring levels of ^{226}Ra (and ^{228}Ra) in suspended sediment in Magela Creek upstream and downstream of the mine site could be used to support the bioaccumulation monitoring project regarding the uptake of Ra in aquatic biota. Low level inputs of U and other heavy metals into Gulungul Creek were detected during the 2003–04 and 2005–06 wet seasons (Sauerland et al 2006, Mellor et al 2007). Earlier research indicated the possibility that these inputs could be related to migration of U from black (acid sulfate) soils in the Gulungul catchment with ^{234}U/^{238}U activity ratios of approximately 1. This work will complement the previous studies on identifying the source(s) of uranium in Gulungul Creek and will complement the work being done on measurements of turbidity and suspended sediment load.

Method

Automatic samplers were used to collect water samples at upstream and downstream sites along Magela and Gulungul Creeks during runoff events in order to quantify suspended sediment concentrations and associated contaminant and radionuclide activity concentrations over a range of flow conditions. ^{226}Ra/^{228}Ra activity ratios in samples collected from Magela Creek are to be measured, in addition to the ^{226}Ra activity and uranium concentration, via the determination of ^{228}Th by alpha spectrometry using a a method developed by Medley (2007). ^{226}Ra activity concentrations and ^{234}U/^{238}U activity ratios are to be measured in samples collected from Gulungul Creek. Trace metals and major ions were analysed, following a reverse aqua regia digest, using ICP-MS and ICP-OES respectively. Over 50 samples from Magela Creek and 20 samples from Gulungul Creek were sent for trace metal and major ion analysis, and the data are awaiting interpretation once analysis of the radiochemical suite has been completed.

Magela Creek

Of the 50 samples sent for ICPMS analysis, 24 samples were sub-sampled for radium analysis covering a range of flow conditions in Magela Creek. Uranium concentration in these samples was determined via ICPMS. Although radionuclide concentrations in suspended sediment is the key parameter to be investigated, initial radioanalyses were performed on the total sample (dissolved fraction and suspended sediment combined). This was mainly due to the limitations on sample volume that can be collected, and the long sample holding time (> 48 h) after collection by the autosampler, resulting in potential for significant loss from the dissolved fraction (in particular for radium) by adsorption on the suspended particles.

Results from radium analyses of the total sample fractions indicate that ^{226}Ra activities in all samples will suffice to allow ^{226}Ra to be determined in the particulate fraction remaining from ICPMS analyses on sample aliquots. However, given the detection limits for ^{228}Ra determination, via ingrowth of ^{228}Th, of ~5 mBq and the fact that ^{228}Ra activity is lower in the fine fraction of sediment in Magela Creek (see results of the mussel longitudinal study reported under KKN 1.3.1 *Stream monitoring program in Magela Creek*), ^{228}Ra results will likely be below detectable limits for most of the samples analysed. An ingrowth period of 12 months is required before ^{228}Th can be analysed via alpha spectrometry and thus radium results are likely be reported in 2009.

Gulungul

Samples were collected based on discharge events rather than turbidity, as increases in turbidity are invariably associated with the rising stage of the hydrograph (Moliere 2005). Four samples were selected from Gulungul Creek for the 2007–08 wet season for radionuclide analysis, and all were obtained from the upstream site in February (Figure 1). Due to an uneventful wet season, no sample collections were triggered at the downstream site.

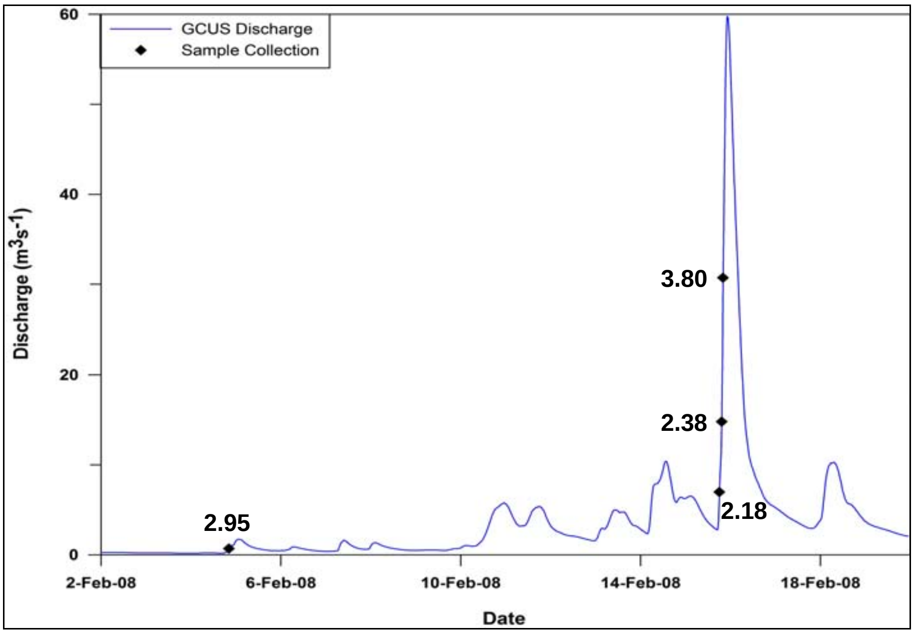

Figure 1 Flow hydrograph of Gulungul Creek, showing discharge at the upstream site, sample collection events and ^{238}U activity concentration [mBq·l^{-1}] measured in the samples

The samples were analysed for ^{226}Ra and for uranium activity ratios and analysis was performed on total (ie unfiltered) samples for the same reasons given above for samples from Magela Creek. All three samples collected on February 16 were collected at an NTU of ~ 30.

The ^{234}U/^{238}U activity ratio in the sample displaying the highest ^{238}U activity concentration was approximately 1, while for the other samples the ^{234}U/^{238}U activity ratio was > 1, but slightly lower than previous grab samples collected from the Gulungul Creek upstream site (Table 1).

Table 1 Mean ^{234}U/^{238}U ratios in samples from grab sampling in 2004 and 2005 and discharge event triggered auto sampling

Sample identification	Mean ^{234}U/^{238}U	Standard deviation
Average of samples analysed from 2004	1.38	0.18
Average of samples analysed from 2005	1.43	0.10
3 samples collected from event triggered autosampler	1.16	0.04
Final sample collected from event triggered autosampler	0.97*	0.09*

* This is the result from a single sample. The standard deviation is based on counting statistics from alpha spectrometry alone.

The lowest ^{234}U/^{238}U activity ratio occurred at the highest discharge level at the site, at an NTU of ~30, which coincides with the highest uranium concentration measured in the total fraction. Ivanovich & Harmon (1982) indicate that in some sediments ^{234}U/^{238}U ratios may be less than unity and therefore mixing of sediment with creek water is considered a likely source of U in these samples. Uranium concentrations and activity ratios will be measured in the particulate fraction of the remaining aliquots from ICPMS and related to turbidity data to investigate the relationship between turbidity and particulate uranium concentration.

Steps for completion

Radioanalysis of particulate fractions in all samples analysed thus far from event triggered sampling of Magela (^{226}Ra) and Gulungul (^{226}Ra and ^{234}U/^{238}U) creeks. Analysis and interpretation of trace metal, radionuclide (uranium and radium) and major ion data for both Magela and Gulungul Creeks.

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Results from the routine stream monitoring program in Magela Creek catchment, 2007–08

Introduction

C Humphrey, A Bollhöfer & D Jones

Progress under this KKN for the stream monitoring program in the Magela Creek catchment is reported by way of (i) results of the routine monitoring program conducted for the 2007–08 period, and (ii) monitoring support tasks for the same period, including research and development, reviews and reporting. The latter tasks are reported separately in 'Ranger stream monitoring: Research and development', p65, this volume.

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

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

Early detection of short or longer-term changes

- *Water physico-chemistry:*
  - Grab samples for water quality meaurement: includes pH, electrical conductivity (EC), suspended solids, uranium, magnesium, calcium, manganese and sulfate (weekly sampling during the wet season) and radium (samples collected weekly but combined to make monthly composites),
  - Continuous monitoring: use of multi-probe loggers for continuous measurement of pH, EC, turbidity, temperature and dissolved oxygen in Magela Creek, and EC and turbidity in Gulungul Creek;
- *Toxicity (including creekside) monitoring* of reproduction in freshwater snails (four-day tests conducted at fortnightly intervals);
- *Bioaccumulation* – concentrations of chemicals (including radionuclides) in the tissues of freshwater mussels and fish 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, fish sampled biannually in the late dry season).

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).

The results from the stream monitoring program and monitoring support tasks and the outcomes of reviews of research programs are summarised in the accompanying papers hereafter.

Chemical and physical monitoring

J Brazier

Routine weekly sampling program in Magela Creek

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

Radium-226 activity concentrations in Magela Creek for 2007–08 are reported separately to ARRTC (see 'Surface water radiological monitoring in the vicinity of Ranger and Jabiluka', pp36–39, in this volume). The first samples for SSD's 2007–08 wet season surface water monitoring program were collected from Magela Creek on 28 November 2007. Weekly sampling continued throughout the season while the creek was flowing, with the last sample collected on 2 July 2008. On 9 July 2008, key stakeholders agreed that continuous surface flow had ceased in Magela Creek and monitoring of the creek was no longer required.

Overall, water quality was comparable with previous years with the seasonal behaviour of all variables consistent with patterns observed in the last five years. The only exception to previous years is the episodic impact on turbidity resulting from erosion of soil from landslips in the upper Magela catchment (well upstream of the mine) into the creek. These landslips occurred last wet season as a result of a record three-day period of torrential rain that occurred in the Magela catchment during late February/early March (This event is discussed in detail in the Supervising Scientist Annual Report 2006–07.) In 2008, during times of heavy rainfall in the part of the catchment containing the landslips, soil was washed into Magela Creek and caused increases in turbidity and a distinctive red colour change to the water at both the upstream and downstream monitoring sites. This occurred on five occasions and the details are reported in 'Turbidity and suspended sediment management guidelines and trigger values for Magela Creek', pp178–82 and 'Definition of sediment sources and their effect on contemporary catchment erosion rates in the Alligator Rivers Region', pp199–205, in this volume).

In late December, rainfall increased in the Magela catchment and the subsequent increase in flow resulted in decreased magnesium concentration, electrical conductivity and pH, and increased turbidity at both the upstream and downstream sites. With rainfall continuing during the last week of December and into the first week of January 2008, Ranger Retention Pond 1 (RP1) commenced seasonal discharge via Coonjimba Billabong into Magela. This discharge occurred approximately a month earlier than in previous years. The input of RP1 water resulted in a corresponding increase in concentrations of uranium, magnesium, sulfate and conductivity at the downstream site. All variables measured by SSD for this period were within guideline values or limits, with the maximum value of uranium reaching only 3% of the 6 µg/L limit.

ERA reported a uranium concentration of 0.93 µg/L on the 7 January 2008 at the downstream site. This value corresponded to the 'Action trigger' for uranium. This is the stakeholder-agreed value beyond which an investigation or contingency plan must be initiated (Figure 1). Water sampling by ERA (investigative and routine) and SSD (routine) showed that over the following week uranium concentrations decreased to levels consistent with previous wet seasons at less than 2% of the limit (Figure 1).

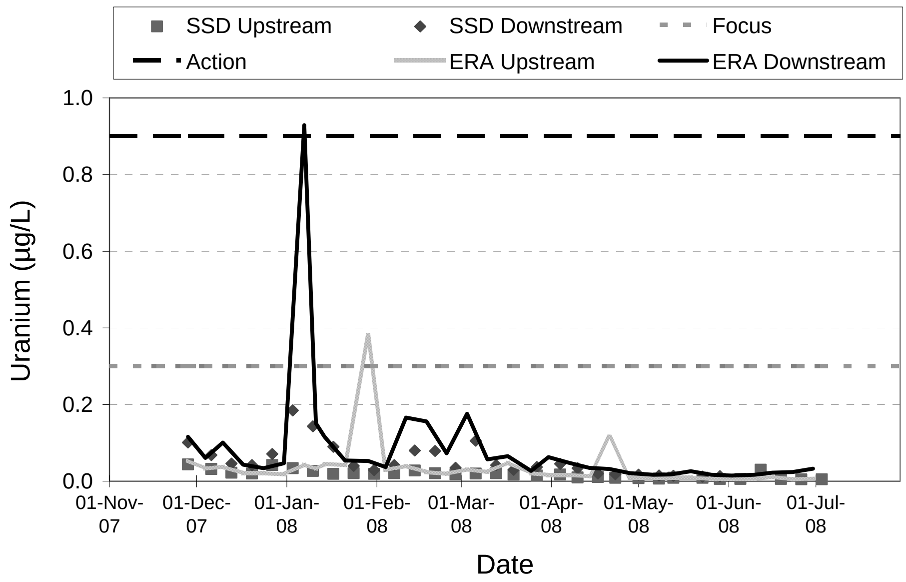

Figure 1 Uranium concentrations measured in Magela Creek by SSD and ERA between November 2007 and July 2008

The routine water sampling program caught the first leading edge of soil movement from the landslip area (discussed above) on the 24 January 2008. An increased turbidity value of 8 NTU was measured at the upstream site (Figure 2). Accompanying continuous monitoring data from Magela Creek showrd that this turbidity event lasted 24 hours, reaching a maximum turbidity value of nearly 40 NTU at the upstream and downstream sites. All other variables measured at the upstream site during this turbidity event were comparable to previous grab sample data.

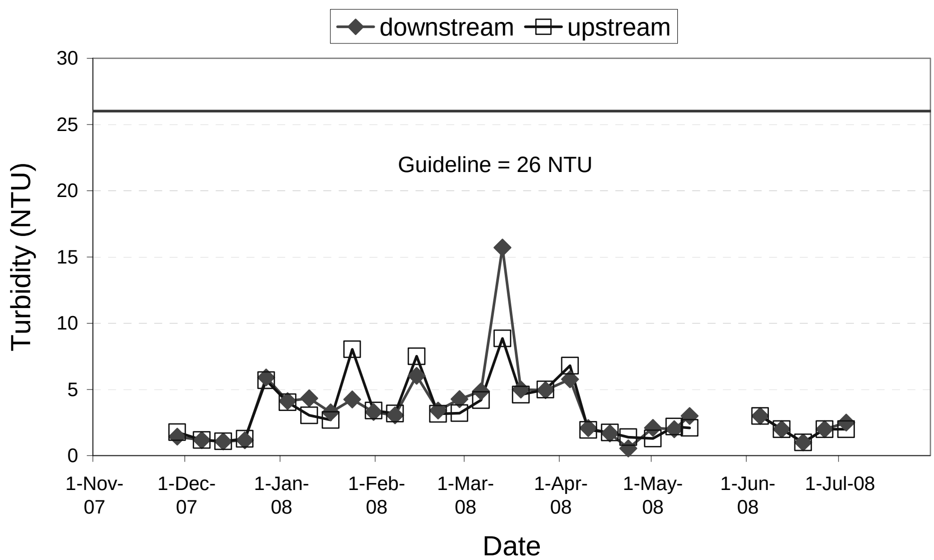

Figure 2 Turbidity concentrations in Magela Creek (SSD data) between November 2007 and July 2008

On the 13 March 2008, turbidity was elevated at both the upstream and downstream site with a higher value recorded at the downstream site compared to the upstream site (Figure 2). Continuous monitoring data show that the value of 15 NTU recorded at the downstream site was at the peak of a turbidity event which lasted 18 hours. Although continuous monitoring data were not available from the upstream site during this event (owing to equipment malfunction), it is inferred that the 8 NTU recorded in the grab sample from the upstream site represents the receding edge of the same turbidity event that was at its peak at the downstream site. This inference is further supported by the fact that though the water level in Magela Creek rose in the 36 hours prior to sampling, local rainfall in Jabiru for the previous 48 hours was only 17 mm (Source: Bureau of Meteorology), suggesting that storm events in the upper Magela catchment (well upstream of the mine) were the source of this suspended sediment load. SSD is currently investigating the contribution of sediment from this source to the annual load of fine suspended sediment in Magela Creek (see 'Turbidity and suspended sediment management guidelines and trigger values for Magela Creek', pp178–82, this volume).

While RP1 ceased flowing over the weir on 8 April 2008, water continued to be siphoned from RP1 into Magela Creek via Coonjimba Billabong. SSD continuous monitoring sondes showed EC steadily increasing from early April at the downstream site (Figure 3) which is not comparable with previous seasons (where upstream and downstream EC values converge as mine site influences decrease). ERA were advised on 11 April 2008 and discharge from RP1 siphoning was discontinued at 1830 hours on 11 April. EC began decreasing at the Magela Creek downstream site approximately 7 hours later. Electrical conductivity, magnesium, sulfate and uranium concentrations continued to decrease over the following weeks at the downstream site in Magela Creek.

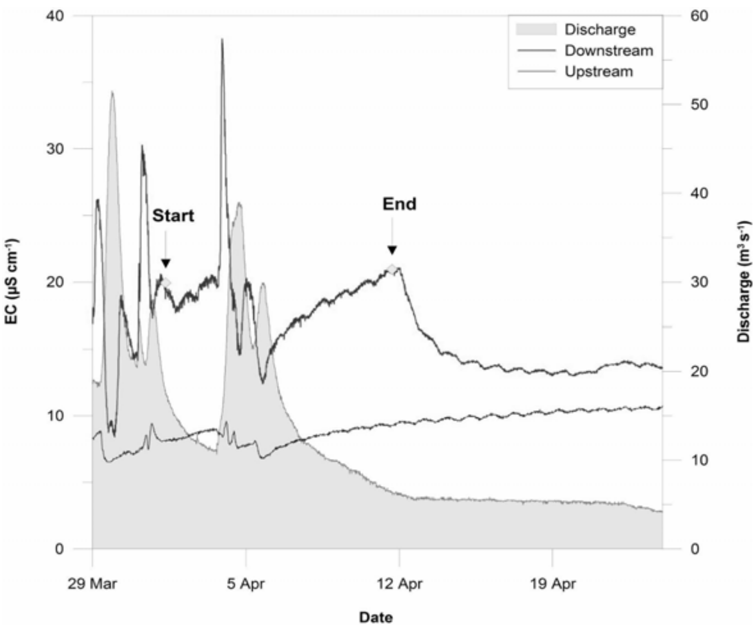

Figure 3 Continuous monitoring of EC and stream discharge in Magela Creek April 2008. Period of RP1 siphoning indicated by 'Start' and 'End' shown on EC trace.

Uranium concentrations were low for the majority of this season, being less than 2% of the limit since February 2008. This behaviour is comparable to previous wet seasons (Figure 4).

Chemical and physical monitoring of Gulungul Creek

The first water chemistry samples were collected from Gulungul Creek for the Supervising Scientist's 2007–08 wet season surface water monitoring program on 27 December 2007, immediately after commencement of surface flow. The last samples were collected on 12 June 2008. Key stakeholders agreed on 19 June 2008 that surface water flow in Gulungul Creek had ceased and monitoring of the creek was no longer required after this date.

All variables at both upstream and downstream sites were comparable to the routine monitoring results from the last five years and show good water quality overall, providing reassurance that water quality in the creek has not been significantly impacted by mining activities.

Turbidity measured on 14 February 2008 at the upstream and downstream sites was the highest measured for the 2007–08 wet season and coincided with increased rainfall experienced in the catchment over that week (Figure 5).

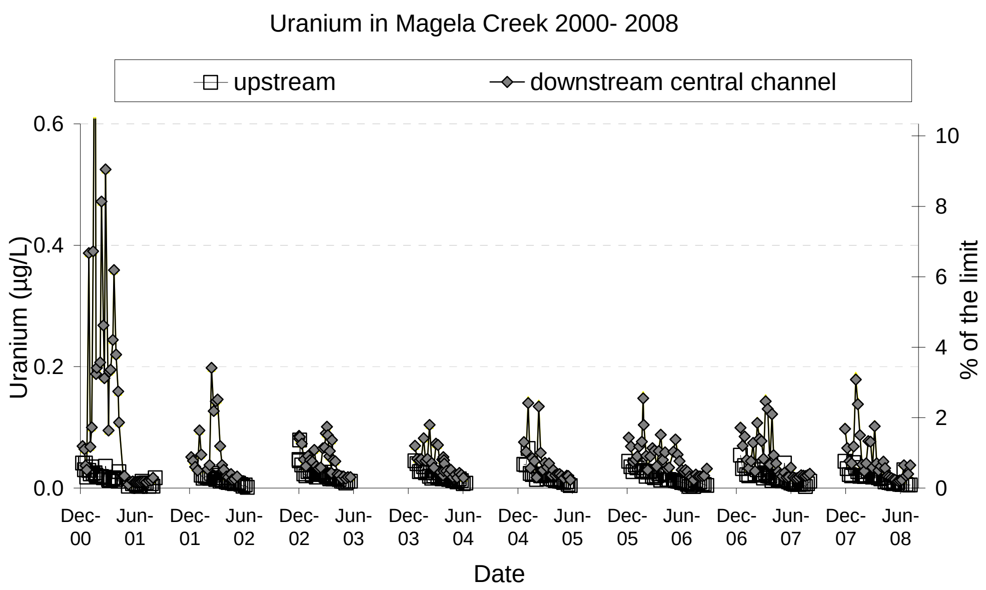

Figure 4 Uranium concentrations in Magela Creek since the 2000–01 wet season (SSD data)

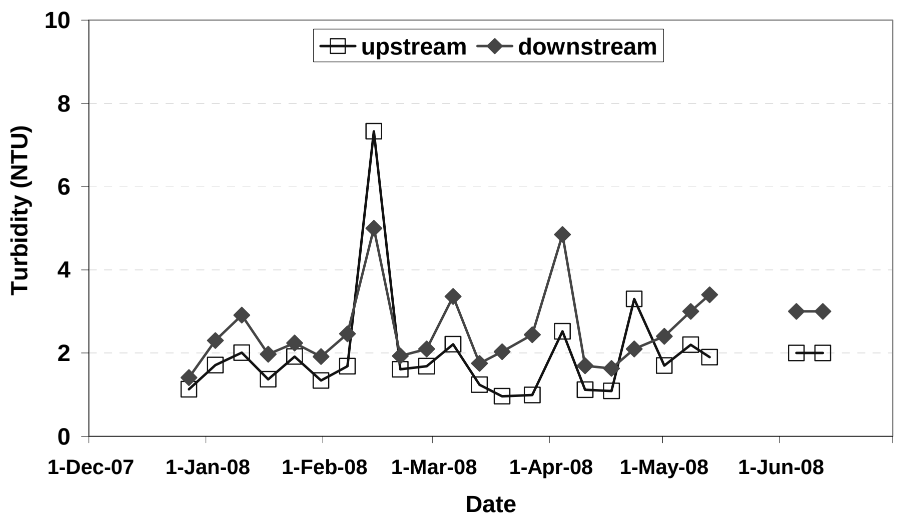

Figure 5 Turbidity measurements in Gulungul Creek for the 2007–08 wet season

Though uranium concentrations were higher at the downstream site, they remained at less than 4% of the limit and were comparable to previous wet seasons (Figure 6). Figure 7 shows that in general there was good agreement between SSD and ERA data for uranium through the 2007–2008 wet season. Any differences between the two data sets were generally the result of different stream conditions pertaining on the different sampling date regimes followed by SSD and ERA.

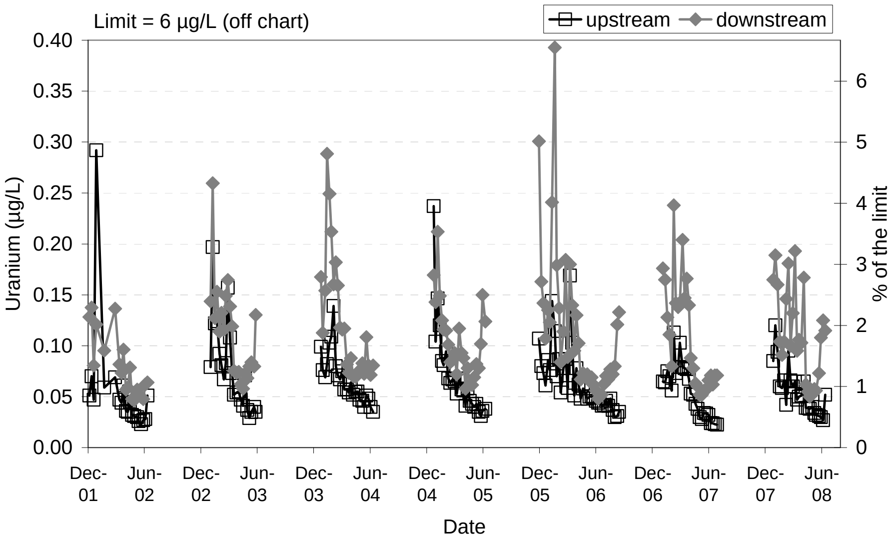

Figure 6 Uranium concentrations in Gulungul Creek between 2000 and 2008 (SSD data)

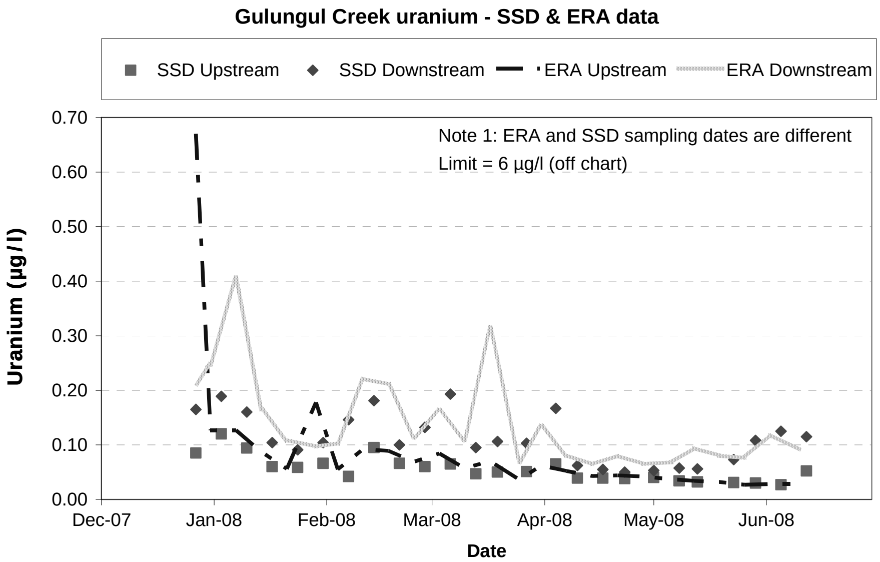

Figure 7 Uranium concentrations measured in Gulungul Creek by SSD and ERA during the 2007–08 wet season

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Toxicity monitoring in Magela Creek

C Humphrey, C Davies & D Buckle

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

Creekside monitoring, where test organisms are held in tanks on the creek bank, has been the primary method used for toxicity monitoring since the 1991–1992 wet season (when toxicity monitoring commenced). Trials of an in situ monitoring method, where test organisms are held in containers located in the creek itself, commenced in the 2005–06 wet season. The purpose of these trials was to evaluate whether this method could replace creekside monitoring as the primary field toxicity method. The findings from this 3 year trial are presented below (see 'Development of in situ toxicity monitoring methods for Magela Creek, 86–90, this volume).

Creekside testing in 2007–08 compared egg production in freshwater snails (*Amerianna cumingi*) upstream of the minesite (control site) and from the creek just below gauging station G8210009, some 5 km downstream of the mine (test site, Magela downstream). At each of the two sites, duplicate containers hold replicate (8) snail pairs (thus 16 pairs of snails exposed per site).

An initial in situ-only test commenced on 2 December 2007, as there was insufficient water depth beneath the creekside monitoring pumping stations to initiate the creekside testing at that time. The first creekside test commenced on 17 December and a further seven four-day tests were conducted every other week between December 2007 and March 2008 to provide reference data for comparison with the in situ method.

The results of the initial in situ test and eight subsequent creekside tests in 2007–08 for snail egg production are plotted as part of a continuous time series of actual and derived 'difference' data in Figure 1. Descriptions of creekside methods and data quality issues are provided in the Supervising Scientist Annual Report 2001–02 and on the SSD website <http://www.environment.gov.au/ssd/monitoring/magela-bio.html>.

Snail egg production at upstream and downstream sites was generally similar across all eight creekside tests, although egg numbers were lower than usual at both upstream and downstream sites during the sixth test. Overall, the pattern of egg production across all creekside tests was similar to that observed in previous wet seasons. Importantly, the upstream-downstream difference values plot around the running mean (since 1991–92 wet season) and are within the maximum and minimum values recorded over this time.

For this reporting year, improvements have been made to statistical analysis of the toxicity monitoring dataset for the purposes of impact detection. While a t-test has been employed in the past to test for differences in the upstream-downstream difference values between two time periods (in particular, before and after an event or wet season of interest), this testing can be extended to a two-factor ANOVA (Analysis of Variance), with Before/After (BA; fixed) and Season (nested within BA; fixed) as factors. The particular ANOVA model used here is likely to be a statistically more powerful test than the t-test, while ANOVA generally is a more efficient test in its simplicity in data preparation and testing of data assumptions. Further, while the first factor (BA) serves the same function as the t-test, the second factor (Season) can be used to determine whether, within the Before and After periods, any set of difference values for a wet season are significantly different. Though this latter test may provide no more additional information in the case of a comparison of the (current) season of interest versus all previous wet seasons, in a comparison of several 'before' and 'after' seasons that are of particular interest, it can potentially identify variability among test responses. Further analysis of the season (BA) factor (ie Tukey's pairwise comparison) would then determine if significant differences occur only in the 'after' period (variability) that could indicate impact, even though mean difference values before and after are not significantly different.

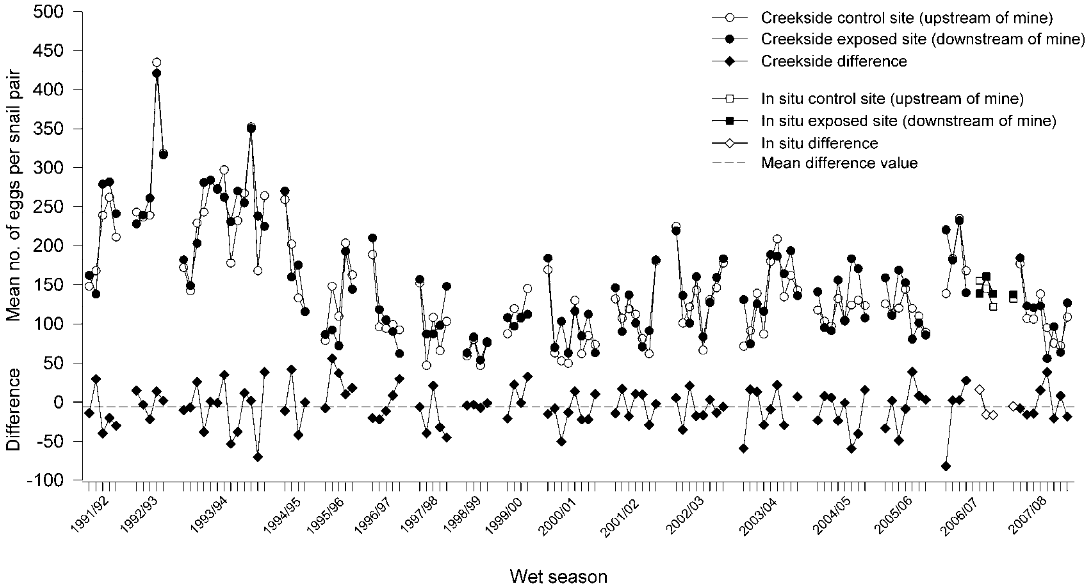

Figure 1 Creekside monitoring results for freshwater snail egg production for wet seasons between 1992 and 2008. Note the last three tests in 2006–07 and the first test for 2007–08 used the in situ monitoring method only.

Applying ANOVA testing to the 2007–08 results, upstream-downstream difference values for snail egg production data were found not to differ from difference values measured in previous wet seasons ($p = 0.709$). Moreover, no differences were observed among the difference values for particular wet seasons within the Before (pre-2008) and After periods ($p = 0.665$). From the creekside results, it is concluded that no adverse effects on freshwater snails from inputs of Ranger minesite waters to Magela Creek occurred during the 2007–08 wet season. This is further supported by the additional in situ monitoring results presented below.

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Bioaccumulation in fish and freshwater mussels from Mudginberri Billabong

J Brazier, A Bollhöfer, B Ryan & C Humphrey

Mudginberri Billabong is the first major permanent waterbody downstream (12 km) of the Ranger mine (see map 3). Local Aboriginal people harvest aquatic food items, in particular fish and mussels, from the billabong and hence it is essential that they are fit for human consumption. Consequently, concentrations of metals and/or radionuclides in the tissues and organs of aquatic biota attributable to mine-waste input to Magela Creek from Ranger must remain within acceptable levels. Enhanced body burdens and bioavailability of mine-derived solutes in biota could also potentially reach limits that may harm the organisms themselves. Hence the bioaccumulation monitoring program serves an ecosystem protection role in addition to the human health aspect.

Mussel bioaccumulation data were obtained intermittently from Mudginberri Billabong from 1980 to 2001. From 2002, regular (annual) sampling from Mudginberri and a control site in the nearby Nourlangie catchment (Sandy Billabong) was initiated. Data prior to 2000 have been discussed and are available in previous Supervising Scientist annual reports. Therefore, only data from 2000 onwards (where methods are standardised and control sites have been included) will be discussed in this report.

Forktail catfish have been identified as the most reliable species to monitor for uranium uptake, primarily because there is a reasonable historical dataset, they are sufficiently abundant in numbers in both billabongs and they are a popular food for the local Aboriginal people. Collection of forktail catfish, sediment and water occurs every two years at both Mudginberri and Sandy Billabongs.

Bioaccumulation of uranium and radium in freshwater mussels

Uranium concentrations in freshwater mussels, water and sediment samples collected concurrently from Mudginberri and Sandy Billabongs are shown in Figure 1. The concentrations of uranium in mussels from both Mudginberri and Sandy Billabongs are very similar from 2000 onwards, with no evidence of an increasing trend in concentration over time and little evidence of an increasing trend in concentration with mussel age (the latter a feature of radium concentrations in mussel soft tissues, see discussion below). Uranium in mussels is reported to have a short biological half-life (Allison & Simpson 1989), a conclusion that is supported by the data in Figure 1, with the uranium concentrations in mussel flesh being low.

The lack of any increase in concentration of U in mussel tissues through time, with essentially constant levels observed between 1989 and 1995 (previous reports), and consistently low levels from 2000 to the last sample taken in May 2007, indicates absence of any mining influence. Mussels were not collected from the control site at Sandy Billabong in 2007 as sampling occurred in May (instead of the normal October period) to coincide with a research project investigating the distribution of radium and uranium in mussels along the length of the Magela Creek, upstream and downstream of the mine. Results for this project are discussed below.

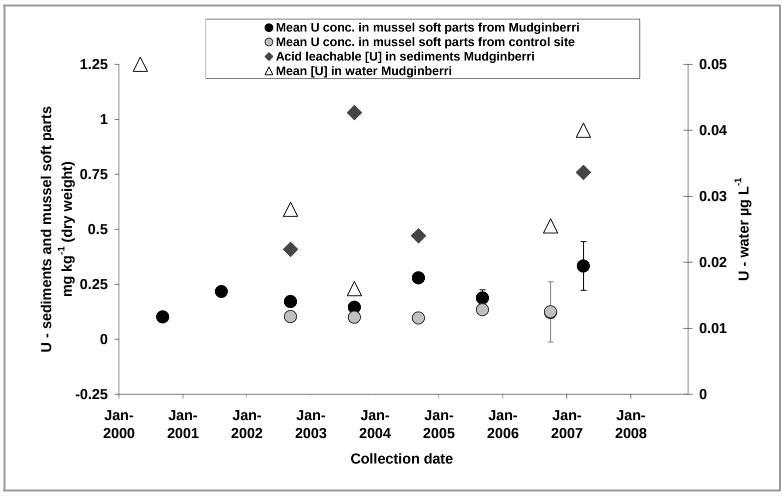

Figure 1 Mean concentrations of U measured in mussel soft-parts, sediment and water samples collected from Mudginberri Billabong and control billabongs since 2000

Concentrations of Ra in mussels are age-dependent (Figure 2) and also appear to be related to growth rates, water chemistry and location (and associated sediment characteristics) within a billabong. When comparing data from amongst years and billabongs (Figure 2), concentrations of Ra in mussels from Mudginberri Billabong are higher, age-for-age, than in mussels from Sandy Billabong. This may be attributable to three factors: (i) naturally higher catchment concentrations of Ra in Magela Creek compared with Nourlangie Creek catchment, (ii) lower concentrations of calcium (Ca) in Mudginberri Billabong waters compared with Sandy (Ca can act as an antagonist to the uptake of Ra by aquatic organisms); and (iii) finer sediment particle sizes in Mudginberri compared with Sandy (finer sediments tend to contain higher Ra concentrations) (Ryan et al 2005). To address whether the Magela catchment has naturally higher radium concentrations, a study was undertaken in May 2008 where mussels, sediment and water were collected along Magela Creek, both upstream and downstream of the mine. Radium, uranium and other key analytes were measured, as well as assessment of the importance of sediment particle size. Results and discussions are presented in 'A longitudinal study of radionuclide and metal uptake in mussels from Magela Creek and Mudginberri Billabong', pp91–97, this volume).

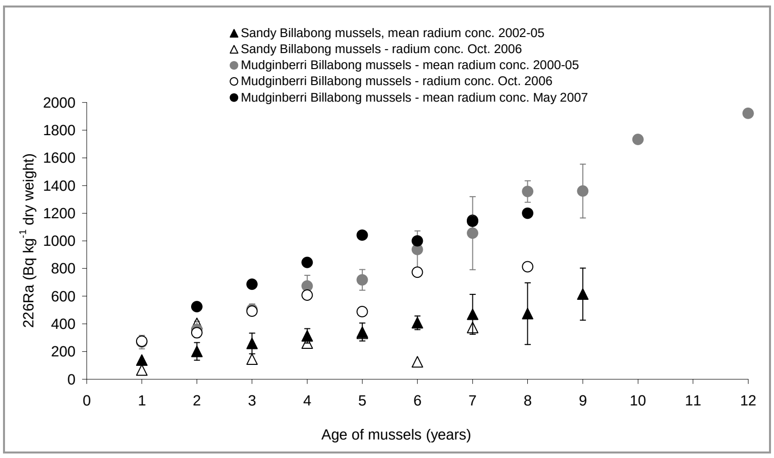

Figure 2 ^{226}Ra activity concentrations in the dried flesh of freshwater mussels collected from Mudginberri Billabong 2000–2007 and Sandy Billabong 2002–2006. The error bars are ± 1 standard deviation.

The average annual committed effective doses calculated for a 10-year old child who eats 2 kg of mussel flesh, based upon average concentrations of ^{226}Ra and ^{210}Pb from Mudginberri Billabong mussels collected between 2000 and 2005, amounts to 0.24 mSv. The average for Sandy Billabong for the same time period amounts to 0.13 mSv. Even in the unlikely case that the difference in doses between the two billabongs was exclusively mine-related, the mine contribution would still amount to only 10% of the public dose guideline limit (ICRP 1996).

The generally consistent relationship between age and Ra concentration observed for mussels amongst years and for each billabong (Figure 2) currently provides a robust baseline against which any future mine-related change in Ra concentrations can be detected. The use of further statistical methods to determine differences in regression relationships will be explored as a means for quantifying any such future change.

Bioaccumulation of uranium in fish

Time series concentrations of uranium in the flesh of forktail catfish collected from Mudginberri and Sandy Billabongs are summarised in Figure 3, together with U concentrations measured in water and sediment for 2000–2007 collections.

The concentrations of U in the flesh of forktail catfish are low (<0.02 mg/kg) with no significant variation over time. This is consistent with the low concentrations of uranium in sediments (<1.4 mg kg^{-1}) and water (<0.05 µg L^{-1}) of Mudginberri Billabong for the corresponding period (Figure 3).

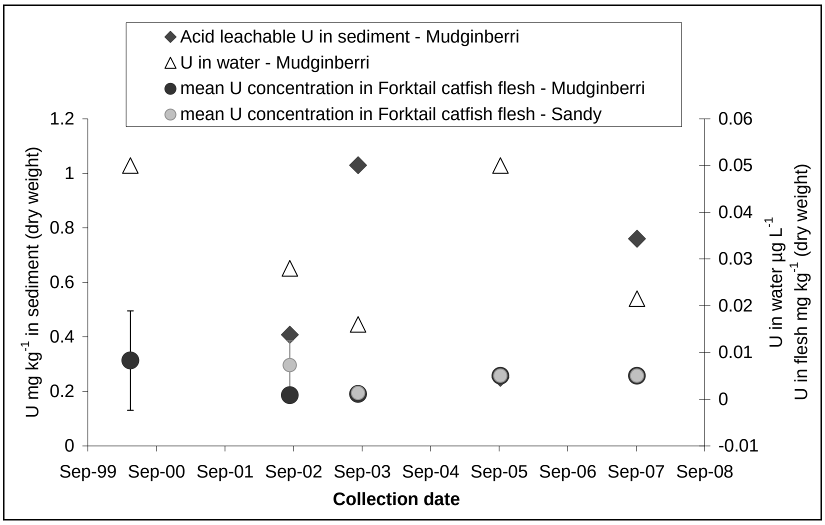

Figure 3 Mean concentrations of U measured in the flesh of forktail catfish, sediment and water samples collected from Mudginberri and Sandy Billabongs, since 2000. Error bars represent standard error. The lack of error bars in some years for forktail flesh samples indicates that the uranium concentration measured was below the detection limit for all samples.

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Monitoring using macroinvertebrate community structure

C Humphrey, L Chandler & J Hanley

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

Samples were collected from each site at the end of each wet season (between April and May). For each sampling occasion and for each pair of sites for a particular stream, dissimilarity indices are calculated. These indices are a measure of the extent to which macroinvertebrate communities of the two sites differ from one another. A value of 'zero' indicates macroinvertebrate communities identical in structure while a value of 'one' indicates totally dissimilar communities, sharing no common taxa.

Disturbed sites may be associated with significantly 'higher' dissimilarity values compared with undisturbed sites. Compilation of the full macroinvertebrate data set, from 1988 to 2008, have been completed with results shown in Figure 1. This figure plots the paired-site dissimilarity values using family-level (log-transformed) data, for the two 'exposed' streams and the two 'control' streams.

In previous years and for each site pair, a single dissimilarity value had been calculated from community structure data that represented a pooled average across the 3 to 5 site replicates. However, this approach, with associated analyses based upon pooled data, does not maximise information contained in the individual site replicates. In particular, by randomly pairing-off the available upstream and downstream replicates, more powerful analyses, potentially, are available that can be used to test whether or not macroinvertebrate community structure has altered significantly at the exposed sites for the recent wet season of interest.

Thus for the first time, the paired-site dissimilarity indices are plotted by way of replicate site data (Figure 1) while multi-factor ANOVA has been extended to include analysis of the replicate (paired-site) dissimilarity values. For 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.

Inferences that may be drawn from the data shown in Figure 1 are weakened because there are no baseline (pre-1980) data upon which to assess whether or not significant changes have occurred as a consequence of mining. Notwithstanding, a four-factor ANOVA based upon replicate, paired-site dissimilarity values and using the factors Before/After (BA; fixed), Control/Impact (CI; fixed), Year (nested within BA; fixed) and Site (nested within CI; random) showed no significant difference between the control and exposed streams in the change (in dissimilarity) from values from earlier years (back to 1998) to those from 2008, (ie the BA x CI interaction is not significant). While the Year x Site (BA CI) interaction is significant in the same analysis ($p = 0.014$), this simply indicates that dissimilarity values for the different streams – regardless of their status (Before, After, Control, Impact) – show differences through time.

The dissimilarity plots shown in Figure 1 corroborate these results, showing reasonable constancy in the mean dissimilarity values for each stream across all years.

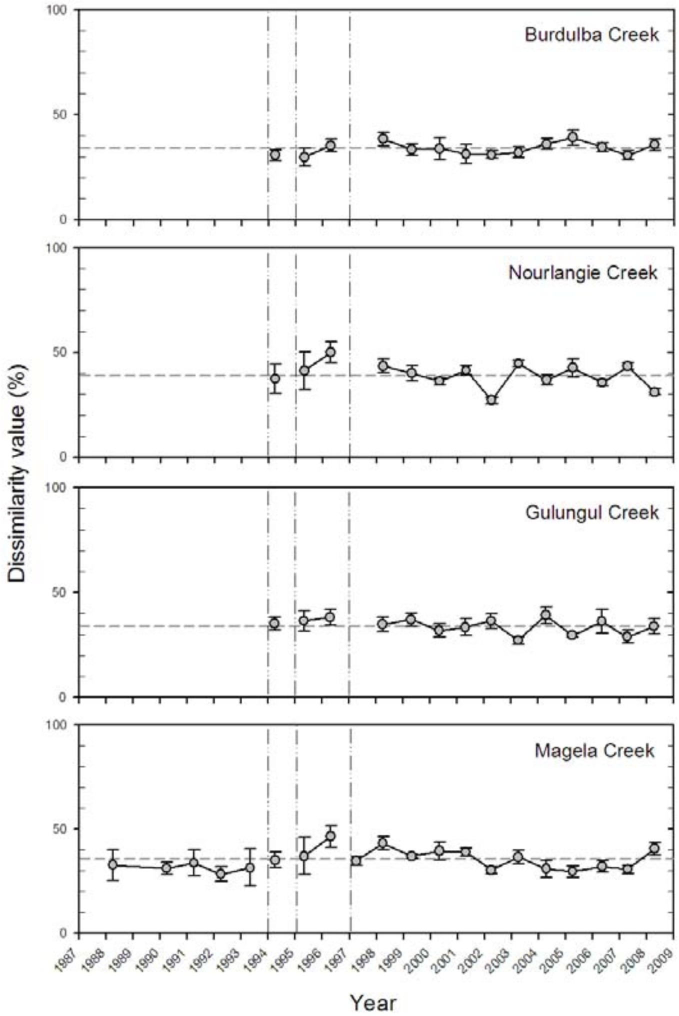

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2008. The dashed vertical lines delineate periods for which a different sampling and/or sample processing method was used. Dashed horizontal lines indicate mean dissimilarity across years.

Dissimilarity values represent means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of upstream-downstream replicate samples within each stream.

Dissimilarity indices such as those used in Figure 1 may also be ‘mapped’ using multivariate ordination techniques to depict the relationship of the community sampled at any one site and sampling occasion with all other possible samples. Samples close to one another in the ordination indicate a similar community structure. Figure 2 depicts the ordination derived using the *pooled* (average) within-site macroinvertebrate data (unlike the replicate data used to construct the dissimilarity plot from Figure 1). Data points are displayed in terms of the sites sampled in Magela and Gulungul Creeks downstream of Ranger for each year of study (to 2008), together with all other control sites sampled up to 2008 (Magela and Gulungul Creeks) or prior. Because the data-points associated with these two sites are generally interspersed among the points representing the control sites, this indicates that these ‘exposed’ sites have macroinvertebrate communities that are similar to those occurring at control sites. This was verified using ANOSIM testing (ANalysis Of SIMilarity, effectively an analogue of the univariate ANOVA), a statistical approach used to determine if exposed sites (Magela and Gulungul downstream) are significantly different from control sites in multivariate space. ANOSIM conducted on (i) pooled (within-site) data from all years, and (ii) replicate data from 2008 only, showed no significant separation of exposed and control sites for the respective comparisons (P>0.05).

Collectively, these graphical and statistical results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining in the period 1994 to 2008 have not adversely affected macroinvertebrate communities.

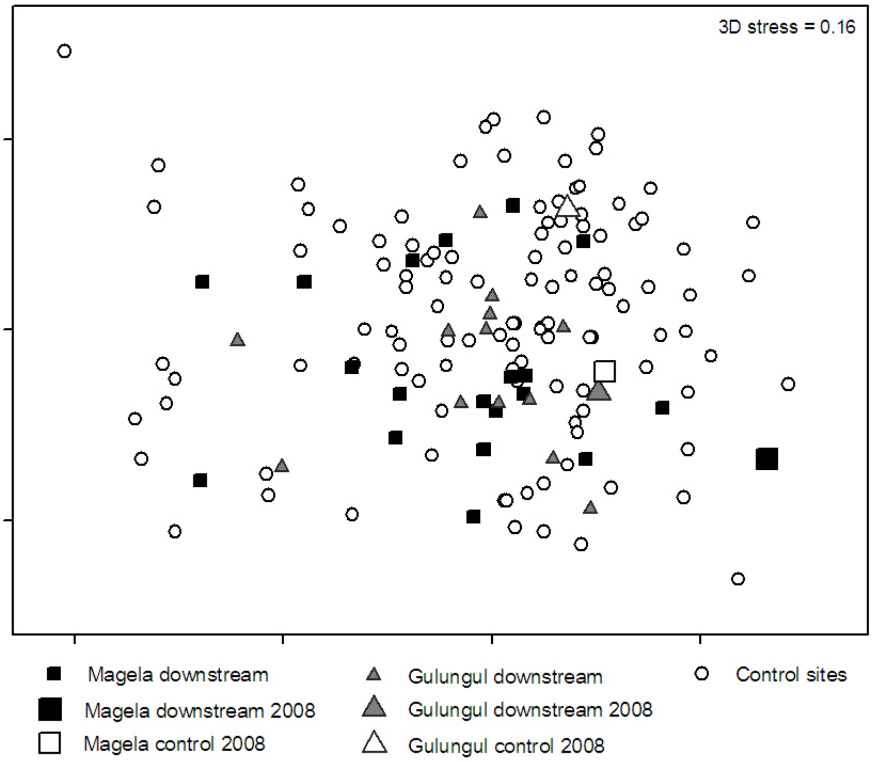

Figure 2 Axis 1 & 2 of a three dimensional ordination plot of macroinvertebrate communities sampled from sites in several streams in the vicinity of Ranger mine for the period 1988 to 2008. Data from Magela and Gulungul Creeks for 2008 are indicated by the larger symbols.

A related study of macroinvertebrate communities, sampled from shallow lowland billabongs in May 2006, is aimed at providing a biological basis for developing water quality closure criteria for the billabongs immediately adjacent to Ranger (see description in 'Developing water quality closure criteria for Ranger billabongs using macroinvertebrate community data', pp130–135, this volume).

Reference

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Monitoring using fish community structure

D Buckle, C Humphrey & C Davies

Assessment of fish communities in billabongs is conducted between late April and July each sampling year. Data are gathered using non-destructive sampling methods, from ‘exposed’ and ‘control’ sites in deep channel billabongs annually, and shallow, lowland billabongs dominated by aquatic plants, biennially. Details of the sampling methods and sites were provided in the Supervising Scientist’s Annual Report for 2003–04.

For both deep channel and shallow lowland billabongs, comparisons are made between a directly exposed billabong in Magela Creek catchment downstream of the mine versus a control billabong from an independent catchment (Nourlangie Creek and Wirnmuyurr Creek). The similarity of fish communities in exposed sites to those in control sites is determined using multivariate dissimilarity indices, calculated for each sampling occasion. Dissimilarity indices are described and defined in the previous paper, ‘Monitoring using macroinvertebrate community structure’. A significant change or trend in the dissimilarity values over time could imply mining impact.

Channel billabongs

The scope of the monitoring program for fish communities in channel billabongs was reviewed in October 2006. An outcome of the review was to reduce the number of visual counts along each of the five billabong transects from five to four. The reduction to four counts was justified on the basis that both the mean, paired-billabong dissimilarity measure and the statistical power in data derived from four counts per transect were not significantly reduced. (The reduction in sampling effort provides time to train new visual observers, thereby reducing observer bias over time.) With the review recommendations implemented in 2008, the complete dataset (1994–2008) has now been standardised to four counts per transect.

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

In the Supervising Scientist Annual Report for 2003–2004, the decline in the paired-site dissimilarity measures over time was noted. While that decline in dissimilarity still remains significant ($P \leq 0.0001$), a recent re-examination of the dataset indicates that the decline is potentially confounded by a change in the field observation method that occurred in 2001. In that year, the original observation canoe was replaced by a slightly larger and more stable observation boat that provided greater protection from the increasing number of saltwater crocodiles observed in the channel billabongs. This method change corresponds with a significant reduction in the dissimilarity measures (ANOVA result $P = 0.004$) between Mudginberri and Sandy Billabong fish communities.

Associated with this difference in dissimilarity, has been a significant increase in the time taken to sample each transect (ANOVA result $P = 0.002$), due to the slightly less manoeuvrable boat. Longer observation times may influence the paired-site dissimilarity between Mudginberri and Sandy Billabongs as the probability of encountering and recording the more cryptic species would be enhanced. While comparative observations made between the two methods during the introduction of the visual boat in 2001 show very little difference between the paired-billabong community dissimilarity values (average dissimilarity 25.2 and 24.8, canoe and boat respectively), an investigation of the data is nevertheless underway to determine whether the method change is a factor explaining the lower dissimilarities, or whether real changes in fish communities over time, and unrelated to sampling method, is the cause.

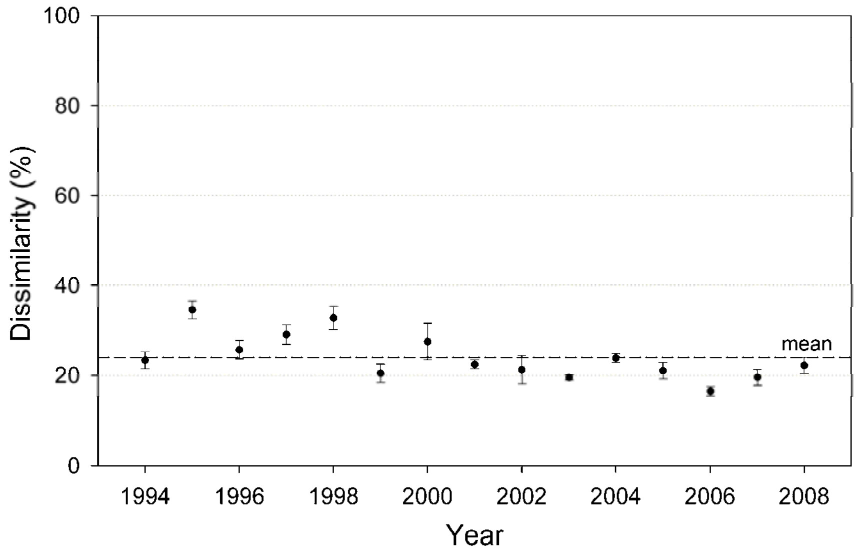

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of the Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

Abundances of chequered rainbowfish (*Melanotaenia splendida inornata*) have the most influence on the paired-site community dissimilarities calculated between Mudginberri and Sandy Billabongs (Supervising Scientist Annual Report 2004–05, Section 3.6.1). This species and its identified decline in abundance since 1989 in Mudginberri Billabong (SSAR 2004–05, Section 3.6.1) do not appear to be influenced by the visual method change introduced in 2001. Chequered rainbowfish are a schooling species and are easily observed using both visual methods; not surprisingly, therefore, there is no relationship between transect time and rainbowfish abundance. Furthermore, the significant decline in chequered rainbowfish abundance ($P \leq 0.001$) occurred *during* the canoe observation period (1989–2000, $P = 0.02$) and primarily in the earlier years. The continuingly lower rainbowfish abundances recorded in latter years have simply maintained the overall decline since 1989.

The decline in rainbowfish does not appear to be related to any change in water quality over time as a consequence of water management practices at Ranger uranium mine. The net input of magnesium (Mg) from Ranger has been used as a reasonably reliable surrogate measure of mine waste-water contaminant concentrations in Magela Creek (see SSAR 2004–05, Section 3.6.1 for further information). For wet seasons from 1988–89 to 2007–08, no significant relationship has been observed between the mine contribution of Mg and corresponding rainbowfish abundance in Mudginberri Billabong. This is not surprising as concentrations of U and Mg in Magela Creek arising from mine waste water discharges are at least two orders of magnitude lower than those known to adversely affect larval fishes, including in the case of U, chequered rainbowfish (SSAR 2003–2004, Section 3.4.1 & SSAR 2004–2005, Section 3.4).

The significance of previously-identified environmental correlates of the rainbowfish decline, ‘wet season stream discharge’ (negative), ‘natural, wet season stream solute concentration’ (positive) and ‘length of previous dry season’ (positive) (SSAR 2004–05, Section 3.6.1) is either marginal or no longer held (regression analysis P = 0.07, P = 0.22 and P = 0.05 respectively). However, further correlation and regression analysis using total monthly discharge in Magela Creek shows that larger flows in either January or February are followed by reduced numbers of chequered rainbowfish in Mudginberri Billabong (p = 0.027 and p = 0.015 respectively). These monthly discharge correlates are refinements of total wet season discharge (from above) and as with the latter measure, are negatively correlated with rainbowfish numbers in Mudginberri.

Previously, it was suggested that low rainbowfish abundances associated with high stream flows may be the result of low solute concentrations in the creek waters which may suppress survival of fish larvae (SSAR 2004–05, Section 3.6.1). An alternative explanation for this relationship may be the greater dispersion (and hence ‘dilution’) of migrating fish that is possible during periods of high stream flows. Early research showed that the main stimulus for the upstream migration of chequered rainbowfish in Magela Creek was wet season flood events immediately preceding the migrations (***eriss*** 1998, Section 2216). Typically, wet seasons of high stream discharge consist of numerous flood events that would provide increased stimuli for rainbowfish to migrate. Thus, the reduced rainbowfish numbers in Mudginberri Billabong following larger January and February flows might suggest that early rainbowfish migrations during these wetter months are more successful and result in greater dispersion of rainbowfish to dry season refuges upstream of the billabong. In wet seasons in which January and February rainfall is relatively low, the upstream migrations of rainbowfish may be reduced and limited, resulting in more fish utilising this dry season refuge site, located lower in the catchment.

Another possible cause for the decline in chequered rainbowfish identified in the SSAR 2004–05 (Section 3.6.1) was ‘Habitat conditions on Magela Creek floodplain’. This refers to the increased spread of a number of grass species on the floodplain since the removal of feral buffalo in the early to mid 1980s. Of particular interest is the exotic Para grass (*Urochloa mutica*) which has expanded rapidly on Magela floodplain over the same period as Chequered rainbowfish have declined in Mudginberri Billabong. These choking grasses have potentially reduced rainbowfish habitat and breeding grounds and hence subsequent recruitment success to upstream dry season refuge areas.

Ongoing annual monitoring of fish communities in Mudginberri and Sandy Billabongs is required to elucidate the cause of the decreased dissimilaritybetween the billabongs and the rainbowfish decline in Mudginberri Billabong since 1989.

Shallow lowland billabongs

Monitoring of fish communities in shallow billabongs is conducted every other year (see SSAR 2006–07). The last assessment of fish communities in shallow lowland billabongs was conducted in May 2007 with results reported in the Supervising Scientist Annual Report for 2006–07. The next assessment will be conducted during recessional flows sometime between late April and June 2009.

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

Introduction

C Humphrey, A Bollhöfer & D Jones

Progress under this study of the stream monitoring program for Magela Creek catchment is reported by way of (i) results of the monitoring program conducted in the 2007–08 period, and (ii) monitoring support tasks for the same period, including research and development, reviews and reporting. Results under Part (i) are reported in 'Results from the routine stream monitoring program in Magela Creek catchment, 2007–08', pp45–63, this volume.

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

1 the future of the SSD's weekly grab sampling program;

2 development of continuous monitoring;

3 development of in situ toxicity monitoring; and

4 longitudinal study of bioaccumulation in mussels in Magela Creek.

Prior to reading these summaries, it is advisable to read the introductory section of the accompanying Part (i) paper describing the rationale of the monitoring program and hence the context for the research and development outlined below.

Future of the weekly water chemistry grab sampling program in Magela Creek catchment

J Brazier, C Humphrey & D Buckle

Proposed relocation of sampling sites in Magela Creek

Ongoing optimisation of existing monitoring methods and, in some cases, development of new methods is necessary to ensure that best practice continues to be employed for detection of possible impacts arising from the Ranger mining operation. To this end, some significant changes will be made in the 2008–09 wet season to the monitoring program that has been in place since 2001. These modifications will enhance the ability of SSD to independently detect changes whilst reducing replication of monitoring activities that are already carried out by Department of Regional Development, Primary Industry, Fisheries and Resources (DRDPIFR) and Energy Resources of Australia Ltd (ERA).

A key operational focus in the 2008–09 wet season will be the closer integration of the continuous and grab sampling water quality monitoring and in situ toxicity (biological) monitoring approaches. Thus, the routine water chemistry grab samples will be collected at SSD's continuous monitoring and in situ toxicity monitoring sites to provide better overlap amongst these methods. These changes to the routine grab sample water quality monitoring program will complement monitoring by ERA and DRDPIFR rather than essentially duplicate them, as has been done in the past.

Weekly grab sampling for measurement of key mine site analytes, including physicochemical parameters, will continue as for previous seasons but will relocate to the pontoon sites used by SSD for continuous and in situ toxicity monitoring.

The upstream pontoon (GTD site) is located approximately 700 m downstream of the current MCUS statutory monitoring location, and the downstream pontoon (009D site) is located approximately 400 m downstream of 009 (locations marked on Figure 1).

To examine the potential effect of changing the locations of the grab sampling sites on the ability of SSD's program to detect impacts from the minesite, chemical data gathered weekly between the 2001 and 2008 wet seasons as part of the creekside monitoring program (locations of the new routine grab sampling sites) were compared with corresponding data from the compliance and reference sites used for the routine grab sample monitoring program between 2001 and 2008. [1] Weekly grab samples were collected for the creekside monitoring program over the full life of this program, with creekside monitoring being conducted every second week. Only weekly routine water chemistry data that overlapped with the period of deployment of creekside monitoring (December to April each wet season) were used for this comparison. However, it should be noted that the days on which the water samples were collected by the two programs (viz water quality grab sampling and creekside monitoring) were not necessarily the same. Between 2001 and 2008, there were 70 sample points for the creekside monitoring program and 110 sampling points for the routine grab sample program that were able to be used for this analysis (reflecting the greater frequency of sampling for the grab sample program).

[1] Creekside monitoring is a form of toxicity monitoring that has been replaced with in situ toxicity monitoring. The samples collected for water chemistry during creekside testing were drawn from waters held in header tanks pumped from floating pontoons in the creek.

A. Upstream monitoring sites on Magela Creek near Ranger

B. Downstream monitoring sites on Magela Creek

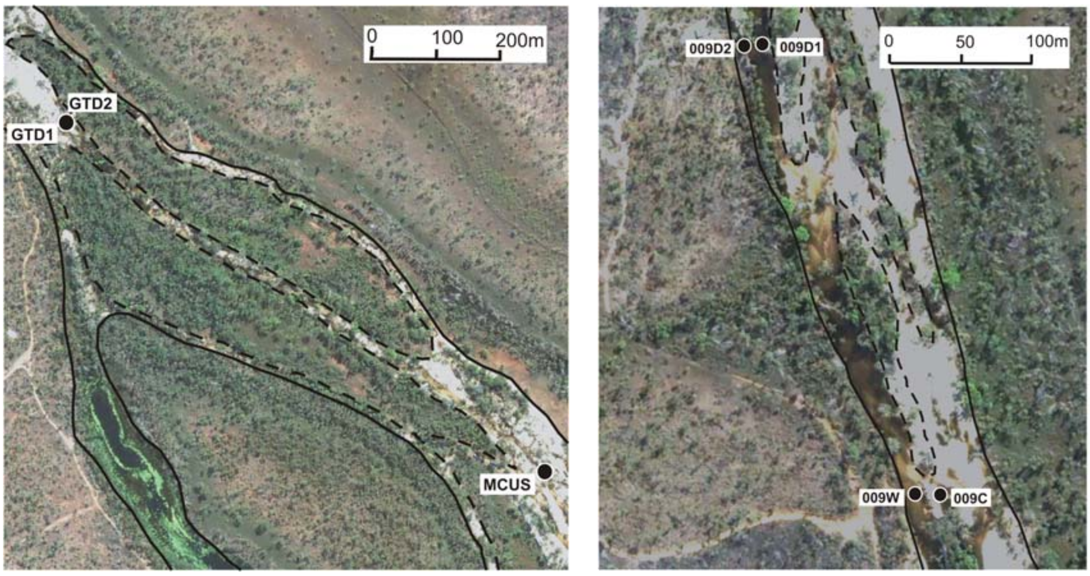

Figure 1 Upstream and downstream monitoring sites used in the SSD's water chemistry (grab sampling and continuous) and toxicity monitoring programs. Channel boundaries are indicated by the continuous or broken (water-level-dependent) lines.

Figure 2 (a–c) shows comparative data for magnesium, sulfate and uranium concentrations, respectively, in the filtered (<0.45 µm) fraction collected for creekside monitoring and the routine water chemistry program. The upstream sites compare well for magnesium, sulfate and uranium. Though the range of uranium concentrations measured at the upstream creekside monitoring site (GTD1/2, Figure 1, essentially a single site as the two creekside pumps are located on the one pontoon) is greater compared with the corresponding range at the upstream statutory monitoring site (MCUS), the concentrations measured at either location are extremely low; there was only one sampling day in 2003 when the uranium concentration exceeded 0.1 µg/L (GTD, Figure 2C; still well below even the focus level for U in Magela Creek at 0.3 µg/L). Confirming these observations, ANOVA testing showed that concentrations were statistically indistinguishable amongst the three sampling locations for any of the analytes, magnesium, sulfate and uranium ($P>0.05$).

The downstream statutory compliance site (009) is located at a position in Magela Creek where there is a single (unbraided) channel (Figure 1). ERA collects a grab sample from a location close to the centre of the channel (009C). Historically, the SSD has collected from two locations – one at 009C and one closer to the western bank at 009W (Figure 1). Approximately 50 m below 009, the creek divides into three channels. The two downstream pontoons associated with the continuous and toxicity monitoring programs are located in the west channel about 400 m downstream of the compliance site. One pontoon is located at 009 D1 on the eastern side of the west channel, and the other is located at 009 D2 closer to the west bank of the west channel (Figure 1).

As a result of both the channel splitting below the compliance site and incomplete lateral mixing of mine waters (leading to a concentration gradient from west to east in Magela Creek at the 009 site), magnesium, sulfate and uranium in water samples collected from nearer the western bank (009W) are similar in concentration to the same measured variables at the pontoons (creekside, 009D) but appear higher in concentration to the same analytes measured at the central channel compliance site (009C) (Figure 2).

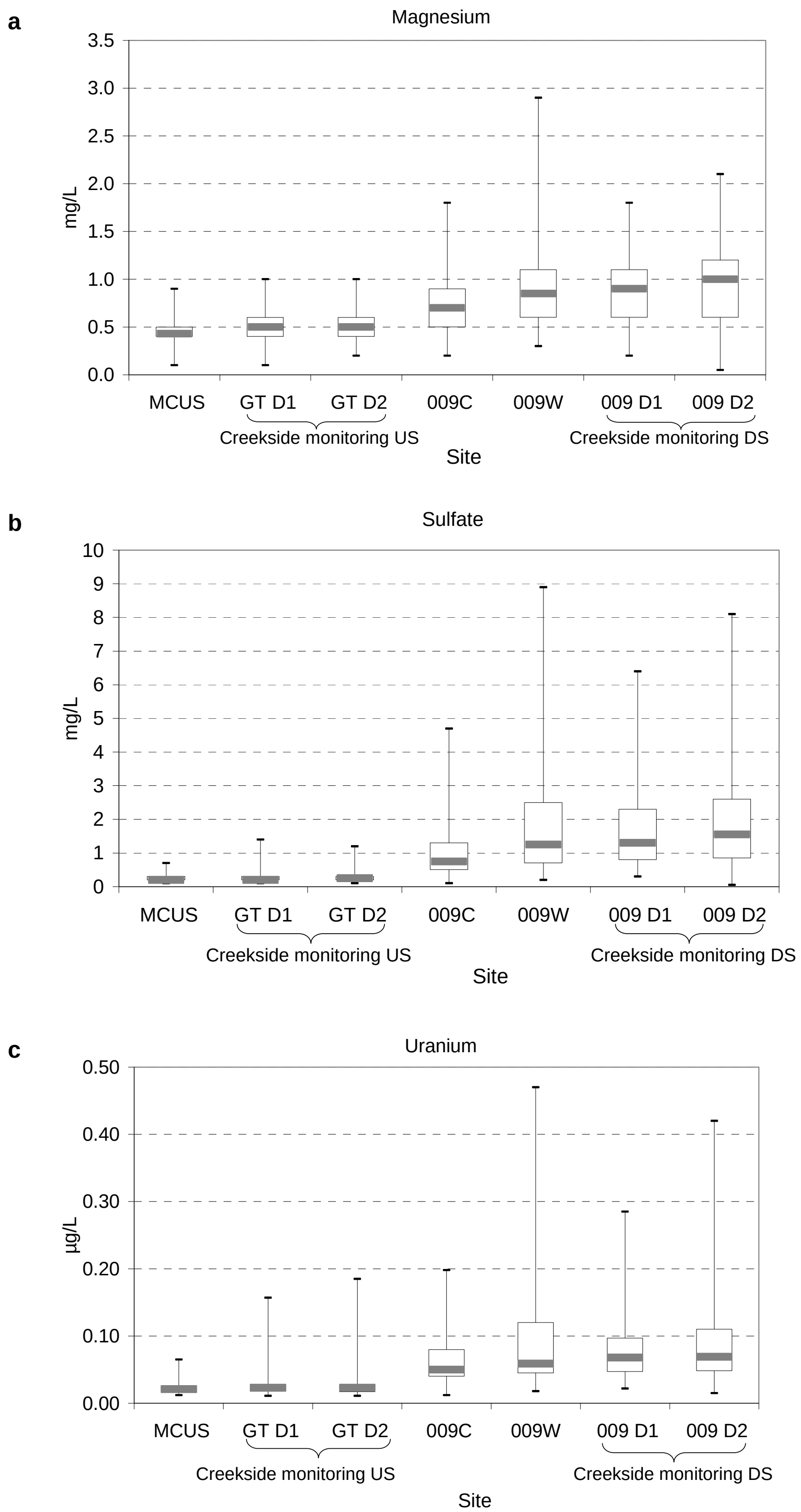

Figure 2 Box plots of concentrations measured between 2001 and 2008 for the upstream routine statutory monitoring site (MCUS), upstream creekside monitoring site (GT D1 and GT D2), downstream statutory compliance monitoring site (009C), downstream site adjacent to 009C but closer to the west bank (009W), and downstream creekside monitoring sites (009 D1 and 009 D2). Boxplots show mean, range, and 25th and 75th percentile. See Figure 1 for site locations.

These observations were investigated further with three-factor ANOVA based upon log transformed concentration data for magnesium, sulfate and uranium using the factors 'Year', 'Side of stream' (west vs east/central) and 'Longitudinal location' (upstream vs downstream) (all factors fixed). Results of the ANOVA are shown in Table 1. While Year was significant for uranium only and longitudinal location significant for sulfate only (though near-significant for U and Mg), Side of stream was significantly different for all three analytes (Table 1), confirming the distinct lateral (west to east) concentration gradient.

Table 1 Results of three factor ANOVA and Tukey pairwise tests examining differences in water quality amongst monitoring locations at the Magela Creek downstream site. None of the ANOVA interactions* was significant for any of the analytes. Emboldened, italicised Tukey comparisons indicate significant pairwise differences.

Analyte	ANOVA factor			Tukey pairwise comparison	
	Year	Side of stream	Longitudinal location	Site pair	P value
Uranium	0.000	0.023	0.075	009C–009W 009C–009D1 ***009C–009D2***	0.0782 0.2438 ***0.0250***
Magnesium	0.309	0.009	0.065	***009C–009W*** 009C–009D1 ***009C–009D2***	***0.0197*** 0.1614 ***0.0074***
Sulfate	0.083	0.001	0.005	***009C–009W*** ***009C–009D1*** ***009C–009D2***	***0.0003*** ***0.0050*** ***0.0001***

* Year and Side of stream; Year and Longitudinal location; Side of stream and Longitudinal location; Year, Side of stream and Longitudinal location

The interaction between 'Side of stream' and 'Longitudinal location' was examined more closely using the Tukey's multiple comparison test. This test provided a pairwise comparison of all four sites, enabling greater interpretation of the significant or near-significant 'Side of stream' and 'Longitudinal location' factors (Table 1). None of the Tukey pairwise comparisons 009W–009D1, 009W–009D2 nor 009D1–009D2 showed significant differences for any of the three analytes, confirming the similarity in water quality in these west-side waters. For all three variables, concentrations were significantly different between the central channel compliance site (009C) and the downstream western pontoon site (009D2), the site pair at the extremes of the lateral concentration gradient. For magnesium, this significant lateral concentration difference with 009C values extended to the adjacent 009W site, while for sulphate it extended to both 009W and 009D1 sites, reflecting, presumably, the close proximity of the downstream monitoring sites to the main source of $MgSO_4$ in Magela Creek, and thus less distance available for mixing (RP1 via Coonjimba Billabong), not far upstream.

Whilst the concentrations measured at the 009 D2 pontoon location are statistically higher than values at the compliance site 009C further upstream, the actual magnitude of the difference is only minor, and is not regarded as sufficient to impact on the decision to relocate the grab sampling site, particularly since sampling in the west channel at the location of the current pontoons will result in a more conservative assessment of the contribution of the mine site to solutes in Magela Creek. Further, given that there is no statistical difference between the D1 and D2 locations, it has been decided to base all sampling (both continuous and grab) on and from the D2 pontoon located closest to the creek bank. This offers improved access conditions, and lower OH&S risk, as well as allowing for the provision of two datasondes on the pontoon, providing redundancy in the case of equipment malfunction.

In further support of the decision to relocate grab sampling to the west channel at the location of the current pontoons, there have only been two occasions in the last eight years of grab sample monitoring where higher uranium concentrations were recorded in the central channel (009C) than in the west channel (009W). This occurred for two consecutive weekly samples for water collected on the 12 and 19 February 2002 when the west channel had a uranium concentration of 0.049 and 0.031 µg/L compared to 0.198 and 0.127 µg/L in the central channel, respectively. On all other occasions, the west channel has had higher or similar EC and uranium concentrations compared to the central channel (continuous monitoring and grab sample data). This analysis supports the view that water quality between 009C and the downstream pontoon sites operated by SSD is sufficiently similar that the integrity of the program will be retained with the proposed relocation of the grab sampling site. In particular, the collection of weekly samples from the west channel should enhance SSD's ability to detect inputs of solutes from the mine site.

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

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

To provide a further integrity check on the field measurement, the field technician will compare the readings taken from the field meter with those being recorded at the same time by the continuous monitoring sonde. If there is good agreement (allowing for known systematic offsets in the continuous readouts), then the field measurement will be recorded as valid and reported to stakeholders. If there is significant disagreement, then a sample will be collected and measured back in the laboratory.

The research emphasis for the water quality monitoring program during the 2008–09 wet season will be placed on event-based sampling to determine if it will be possible to establish a correlation between EC and U at higher EC values. A desk top study will also be conducted, using historical data, to investigate the relationship between unfiltered and (<0.45 µm) filtered uranium concentrations with turbidity and EC to better understand the possible limitations of event-based sampling (for example, losses of U from solution by absorption to particles prior to retrieval of the autosampler bottles from the field).

Future of the weekly grab sampling program – Gulungul Creek

Weekly grab sampling for routine analysis of water chemistry variables will be discontinued at the upstream site commencing with the 2008–09 wet season, as this does not represent a useful reference site (ie water chemistry measured at this site may reflect upstream (natural) catchment influences that compromise its effectiveness for assesing downstream impacts from the mine). Weekly monitoring will continue at the downstream site. The continuous monitoring of EC and turbidity will be maintained at both the downstream and upstream sites.

Continuous monitoring of water quality in Magela Creek

K Turner

Background

Continuous monitoring of surface waters at a number of locations within, and outside of, the Ranger uranium mine site is conducted by Supervising Scientist Division and Energy Resources of Australia Ltd (ERA). SDD monitors sites in Magela Creek and ERA monitors sites in the Georgetown Creek and RP1 catchments that convey solutes from the site into Magela Creek. The data are used in the assessment of potential impacts arising from activities carried out at the minesite.

In situ sensors are maintained by SSD at key sites in Magela Creek upstream and downstream of the mine (Supervising Scientist Annual Report 2006–2007). An important attribute of SSD's continuous monitoring network is its ability to quickly distinguish between natural events in the creek system and those that occur in response to inputs from the minesite. The outputs from the SSD monitoring sensors are available online in the Darwin office via telemetry, and automatic alerts are sent to mobile phones in the event that pre-set levels are exceeded.

SSD has carried out its continuous monitoring program for the past four years, collecting in situ water quality (electrical conductivity, turbidity and pH) data in Magela Creek at intervals of 5–15 minutes. This acquisition complements SSD's current routine water chemistry program that involves the collection of weekly grab samples for comprehensive chemical analysis. In this paper the continuous data have been used in two contexts and, accordingly, are reported separately:

(a) Within the present report, data throughout the 2007–08 wet season have been used to aid interpretation of the routine water quality measurements (see 'Chemical and physical monitoring, in this volume', pp46–50, this volume). In particular, continuous monitoring data have been used to better understand the flow event dynamics of magnesium (Mg) concentrations. Magnesium is the dominant major ion solute in water produced from mining activities at Ranger and is potentially an important contributor to aquatic toxicity in Magela Creek.

(b) In a separate report in this volume (see 'Development of Magela catchment area solute budget using continuous monitoring systems', pp75–85, this volume) Mg concentrations predicted from the continuous EC data are used to derive the loads of Mg transported along Magela Creek, apportioning the relative contributions arising from point and diffuse sources by comparing total loads of mine-derived solutes estimated in Magela Creek with continuous data data collected from the two major catchment flowlines on the Ranger site.

The continuous data are also being used to investigate the transport of U and other metals associated with sediment. The outcomes from this work will be reported in the future.

Use of continuous EC data to monitor Mg concentrations in Magela Creek

The derivation and description of strong linear relationships between Mg concentration and and electrical conductivity (EC) are described in 'Development of Magela Catchment area solute budget using continuous monitoring' (pp75–85, this volume). Continuous Mg data (predicted from the EC trace) for the 2007–08 wet season (Figure 1) show good agreement with the Mg concentrations measured concurrently in grab samples collected as part of the routine water chemistry monitoring program from the statutory upstream and downstream sites (MCUS and G8210009, respectively).

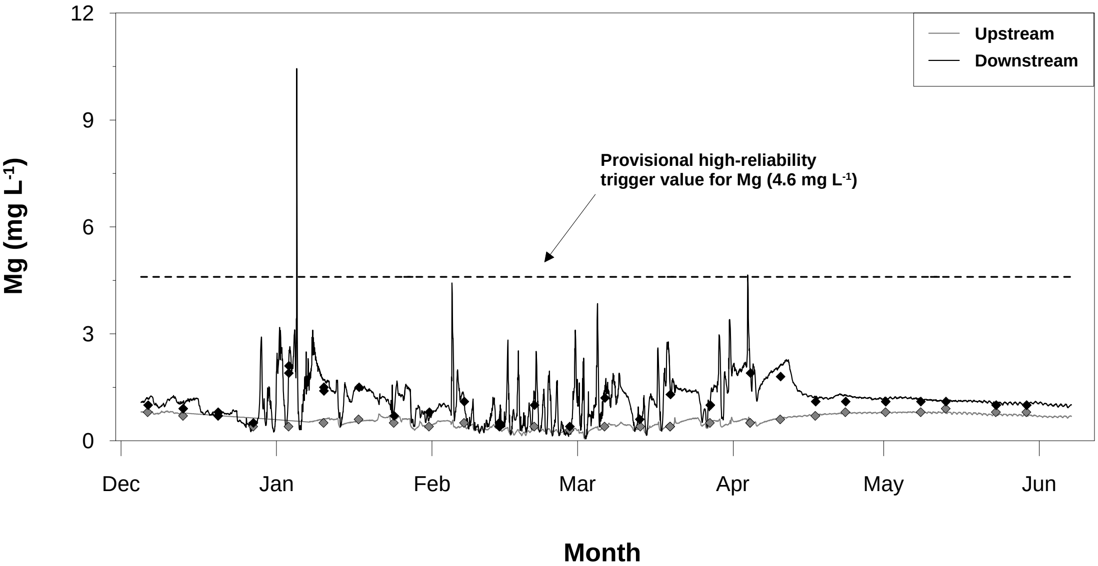

Figure 1 Predicted continuous Mg concentration (lines) with overlain measured grab sample values (diamond points) for upstream and downstream of the Ranger mine for the 2007–08 wet season

The Mg concentration traces provide a more complete description of the dynamic changes in water quality of the creek system, illustrating that transiently-elevated levels of Mg can pass undetected by the much less frequent routine water chemistry measurements (Figure 1). This is a consequence, in part, of the scheduled timing (0800 to 1200 h) of grab sample collection in relation to the diurnal variation in EC at the downstream site. There is a strong diurnal variation in flow in Magela Creek (Figure 2), occurring as a result of diurnal variation in rainfall in the upper reaches of the creek catchment. As a result, background EC in Magela Creek (ie at MCUS) is also diurnal as input of rainwater via surface runoff from the upstream escarpment dilutes solute concentrations in the base flow, which is more strongly influenced by higher EC groundwater.

The diurnal behaviour of EC is confounded downstream of the mine because input of higher EC mine waters via the Coonjimba Creek and Corridor Creek tributaries (RP1 and GC2 catchments respectively) is dependent on the level of water in Magela Creek, amongst other factors. At rising and high flows in Magela Creek, mine-derived waters become backed-up in Coonjimba and Georgetown Billabongs (see maps 2 & 3) and are only discharged to Magela Creek when flow in the main creek recedes, resulting in increased EC at the downstream site during these periods of falling or lower flows in Magela Creek. This results in an inverse relationship between flow and EC in Magela Creek (Figure 2) where the diurnal behaviour of the net EC (mean downstream EC minus mean upstream EC) directly attributable to input of mine waters is illustrated. This demonstrates how low frequency (relative to the time constant for the primary driving process), fixed-time grab sampling may be inadequate for tracking water quality in streams with large variations in flow.

Of particular note is that the continuous monitoring data at the downstream site showed that the provisional high-reliability trigger value for Mg (4.6 mg L^{-1}) was exceeded 7 times over the past three wet seasons. None of these short duration excursions were detected by the routine weekly monitoring program. The highest inferred Mg concentration during the past three wet seasons was 11.3 mg L^{-1}, nearly 2.5 times the trigger value. However, the results from the biological (toxicity) monitoring program (reported below) indicated no adverse ecological effects arising from such short duration elevations of Mg concentrations.

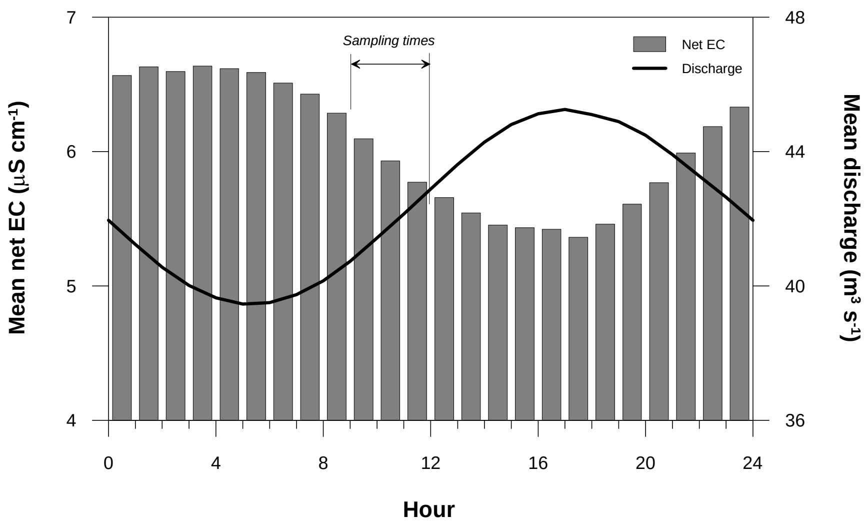

Figure 2 Mean hourly net EC (mean downstream EC minus mean upstream EC) in Magela Creek over the 2005–06, 2006–07 and 2007–08 wet seasons (grey bars). Also shown is the mean hourly Magela Creek discharge over the entire period of record (black line). The typical day time window (sampling times) during which grab samples are collected as part of SSD's routine water chemistry monitoring program is shown for comparison.

The lack of a biological response is not surprising due to the very short duration of the exceedances, ranging from 7 minutes to 4 hours. The ecotoxicological tests that were used to derive the Mg trigger value are based on chronic exposure periods of 3 to 6 days, substantially longer than any of the in situ exposures that have occurred in Magela Creek. Metal concentrations required to cause adverse effects from short exposures are typically higher than for longer, continuous exposures. This means that the Mg trigger value based on chronic exposure endpoints is likely to be substantially overprotective for short duration pulses. In the context of the whole wet season, the Mg exceedances are likely to have negligible impact on the system as the Mg concentration was only above the trigger value (and by only a small factor) for an average of 0.5% of the duration of each wet season (Figure 3).

Nevertheless, SSD is using short duration pulse exposure toxicity testing methods to quantify the effect of such short duration elevated exposures to Mg on the most sensitive members of its suite of toxicity test species. The findings from this work will be reported in future papers.

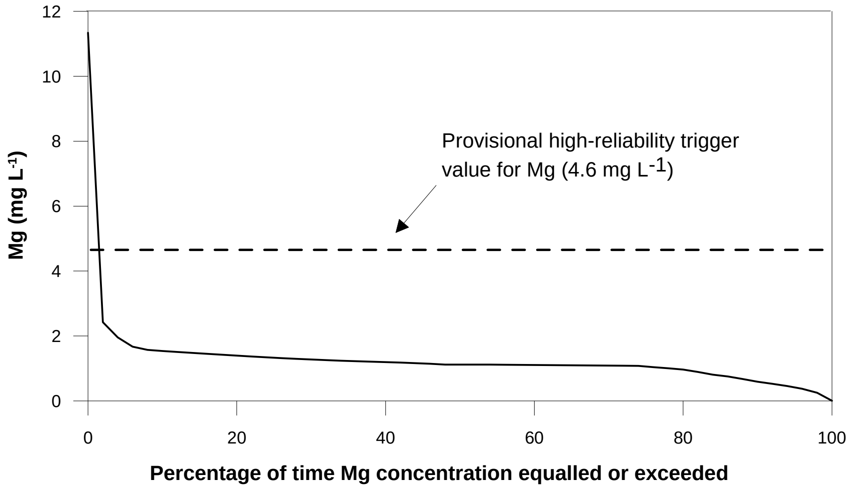

Figure 3 Frequency distribution curve for Mg concentrations predicted from the continuous (5-15 min frequency) EC record in Magela Creek downstream of the mine, showing the percentage of the time that the Mg concentrations equalled or exceeded values within the range of Mg concentrations between 2005 and 2008. The provisional trigger value (Ca-protected) for Mg is shown for comparison.

Development of Magela Catchment area solute budget using continuous monitoring systems

K Turner & D Jones

Background

Continuous monitoring of surface waters around the Ranger uranium mine is conducted by both SSD and ERA and the data is used in the assessment of potential impacts arising from activities carried out at the mine site. In situ sensors are maintained by SSD at key sites in the receiving waters of Magela Creek upstream and downstream of the mine, and by ERA in two mine site tributaries, Coonjimba Creek and Corridor Creek (see Map 2). Two key attributes of SSD's continuous monitoring network include the ability to remotely monitor events in the creek system to:

1 quickly distinguish between natural events and those that occur in response to inputs from the mine site;

2 compare EC values and predicted Mg concentration values with ecotoxicology-derived guideline values for Mg; and

3 quantify the loads of solutes and sediment in the Magela Creek system, with the aim of identifying of contaminants arising from mining activities.

The first 2 points have previously been described in detail (Supervising Scientist 2007, Turner et al 2008a & 2008b) and are summarised in the companion surface water monitoring paper under this KKN. The third point, including results from ongoing investigations designed to better understand solute transport and loads in Magela Creek, is addressed here.

Ranger uranium mine is located within the 500 km^2 Magela Creek catchment which comprises sandstone escarpment in the southwest region and lowland flood plains to the northeast. Magela Creek is a seasonal system that is characterised by waters of very low electrical conductivity (EC) and slightly acidic pH, reflecting the highly weathered soils and large area of inert sandstone escarpment in the catchment (Hart et al 1982). The effect of groundwater input on surface water quality is dependent on catchment rainfall and runoff and has been discussed elsewhere (Klessa 2005, Supervising Scientist 2003).

Magnesium sulfate ($MgSO_4$) is the major constituent of the mine-derived waters, arising from weathering of magnesite, chlorite and minor sulphides in waste rock and low-grade ore stockpiles located within the Coonjimba and Corridor Creek catchments (Map 2) (leGras & Boyden 2001). Work carried out by Van dam et al (2006, 2008) has shown that magnesium (Mg) is potentially an important contributor to toxicity in Magela Creek water, while in contrast sulfate was shown to be of very low toxicity.

During the wet season months, mine-derived waters containing elevated (relative to upstream Magela Creek) concentrations of Mg are passively released into Magela Creek via the Coonjimba Creek and Corridor Creek catchments, which include Coonjimba Billabong and Georgetown Billabong respectively (see Map 2). These tributaries only connect with Magela Creek during the wet season months, at which time their water quality is dominated by inflow of surface runoff from waste rock dumps and low grade ore stockpiles located on the minesite.

Additional (non-point sources of Mg) include wet season induced leaching of RP2-derived water applied to the footprints of the land application areas (LAAs) during the preceding dry season. This water infiltrates the soil profile, and a proportion is flushed out during the subsequent wet season. Previous analysis of the solute delivery from the LAAs suggested that most of the Mg is washed out during the following wet, whilst sulfate behaves non-conservatively. The accuracy of this earlier analysis of solute export from the LAAs was confounded by having to rely on weekly grab samples rather than continuous monitoring data. This latter situation has been addressed since the 2005–06 wet season by SSD's continuous monitoring of EC in Magela Creek.

Turner et al (2008a & b) showed that background Mg concentrations in Magela Creek exhibit diurnal behaviour driven by diurnal variation in base flow. Magnesium variation is confounded downstream of the mine as input of higher Mg mine waters from the mine site tributaries occurs via Coonjimba and Georgetown billabongs which are backflow billabongs that only flush into the creek channel under low flow conditions. The flow dependent nature of Mg concentrations in Magela Creek highlights the need to consider Mg loads as well as Mg concentrations.

Previous studies have included estimation of event-based solute loads and subsequent derivation of total annual solute loads in Magela Creek (Hart et al 1982, 1986, Klessa 2005). However, these previous load estimates were based on solute measurements in grab samples taken over a only small portion of the period of total annual flow and therefore contained a high degree of uncertainty. The continuous monitoring network has enabled calculation of Mg loads based on continuous data (5–20 minutes) improving the accuracy of load estimations. Ultimately, by comparing the total mass of solutes transported downstream of the mine in Magela Creek with the mass of solutes from point and diffuse sources, a dynamic assessment the intra- and inter-seasonal fluxes of salts in the system will be possible.

Methods

Since the 2005 wet season, continuous in situ EC data have been collected in Magela Creek at 5 to 20 minute intervals using sensors (Datasondes) located at sites upstream and downstream of Ranger mine (near Magela u/s and Magela d/s, respectively (Map 2)). The upstream Datasonde is located a short distance upstream of the Georgetown-Magela Creek confluence and is free from mine surface water influence. The downstream Datasondes are located on either side of the western-most channel in Magela Creek and are referred to as downstream east (DSE) and downstream west (DSW) according to their respective locations in the channel.

Datasondes managed by ERA measure water level and EC data in Ranger minesite tributaries, specifically at the RP1 spillway in Coonjimba Creek and a site called GC2 in Corridor Creek (Map 2). Magela Creek discharge data are collected by the Department of Natural Resources of the Environment and The Arts (NRETA) from the gauging station G8210009, located approximately 500 m upstream of SSD's Magela d/s site (Map 2).

Weekly surface water samples have been collected by SSD from Magela Creek upstream and downstream of Ranger mine (near Magela u/s and Magela d/s respectively (Map 2)) and by ERA at the RP1 and GC2 sites since 2001. The samples are analysed for dissolved (<0.45 µm filtered) solute concentrations using either inductively coupled plasma mass spectrometry (ICP-MS) or inductively coupled plasma atomic emission spectroscopy (ICP-AES). In situ EC is measured at the time of sample collection.

Electrical conductivity – Mg relationships

Relationships between EC and Mg at each of the sites have been derived by correlating Mg concentrations in grab water samples with concurrent EC measurements. There are statistically very strong linear relationships between Mg concentration and EC at the four continuous monitoring locations (Figure 1).

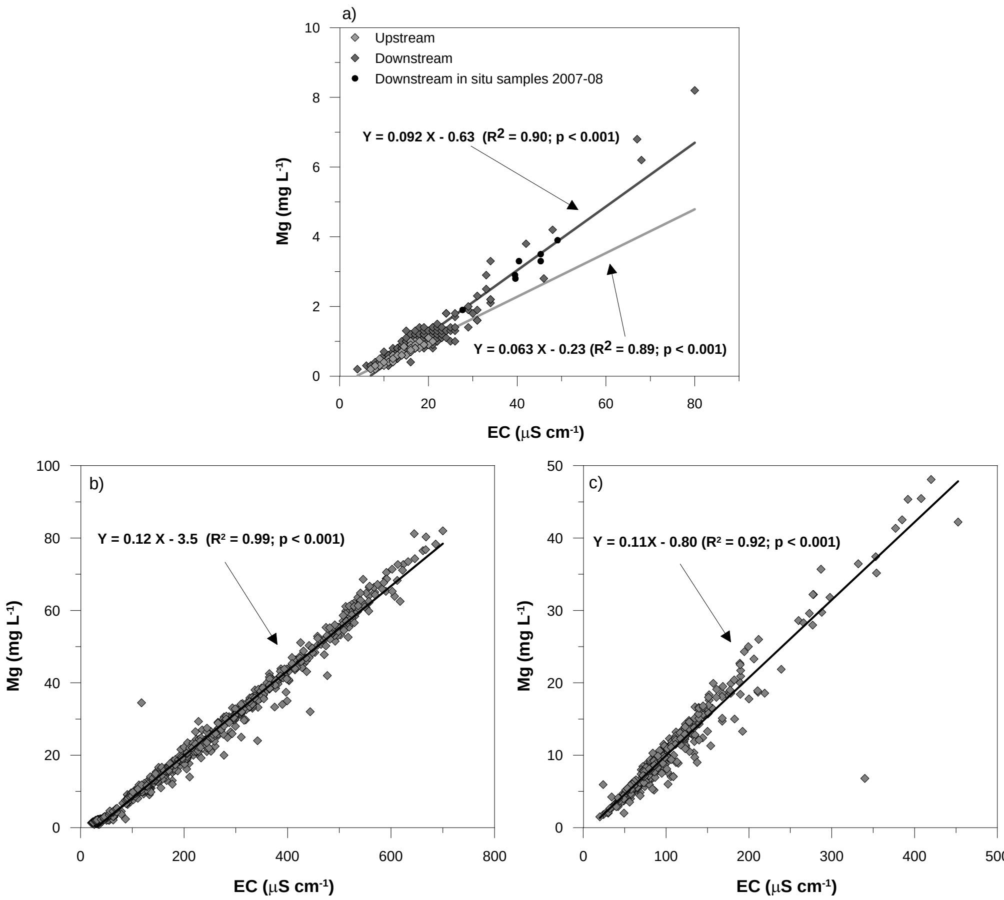

Figure 1 Relationships between EC and Mg concentration for the a) upstream and downstream sites on Magela Creek, b) RP1 in the Coonjimba Creek catchment and c) GC2 in the Corridor Creek catchment

Different EC-Mg relationships exist for the upstream and downstream sites in Magela Creek and RP1 and GC2 as a result of different Mg sources, concentration ranges and relative contributions of the constituent major ions present at each of the sites. Klessa (2005) showed that upstream of the mine, the major cation composition of Magela Creek water is Mg and Na dominant (Mg=Na>Ca>K) with sulfate, Cl^- and HCO_{3^-} being the co-dominant anion species. In contrast, the mine site water bodies are $MgSO_4$ dominant. Table 1 summarises the parameters characterising the relationships between EC and Mg for each of the sites.

Table 1 Summary of EC-Mg relationships for Magela Creek upstream and downstream of the mine and at the RP1 and GC2 sites on minesite tributaries

Sites	Slope	r^2	p
Upstream	0.063	0.89	<0.001
Downstream	0.092	0.90	<0.001
RP1	0.12	0.99	<0.001
GC2	0.11	0.92	<0.001

The slope of the regression for RP1 and GC2 are essentially the same, indicating that the major ion compositions are similar at both sites. The lower slope in Magela Creek water upstream of the mine indicates a greater influence of major ions other than Mg. The major ion composition of Magela Creek water downstream of the mine is a mix between upstream (background) and released mine derived waters, giving a slope that lies between that of the minesites and the upstream site, albeit much closer to the slopes for RP1 and GC2.

While there is a strong correlation between measured EC and Mg at the Magela Creek downstream site, the relationship lacks data points in the upper region of the range depicted, increasing the uncertainty about the reliability of the data fit at the upper end. To validate the relationship, higher EC (≥ 40 µS/cm) water samples were specifically collected during the 2007–08 wet season using an in situ automated sampler, triggered by EC readings from the Datasonde. Whilst the higher Mg concentrations measured in these samples fitted the original regression well and did not alter the slope of the derived EC-Mg relationship (Figure 1), the Mg concentrations in the samples collected only spanned the range of 40–50 µS/cm, thus the top, right end of the correlation remains sparsely populated. It is important that there is high confidence in the predicted Mg concentration in the upper region since it is in this region where Mg could approach environmentally significant concentrations (ie with respect to the derived guideline value for Mg). It has been suggested that predicted Mg concentrations should be reported as daily (or even 96 hr) means to better align the data with the duration of toxicological test methods (van Dam, in press). The distribution of daily mean EC versus flow at the Magela Creek downstream site is shown in Figure 2. The data illustrate that since the 2005-06 wet season, the daily mean EC rarely exceeds 50 µS/cm. All exceedences occurred at flows between 10 – 50 m^3/s, under which conditions the backflow billabongs (Coonjimba and Georgetown) flush mine derived waters into Magela Creek, increasing the Mg signal at the downstream site.

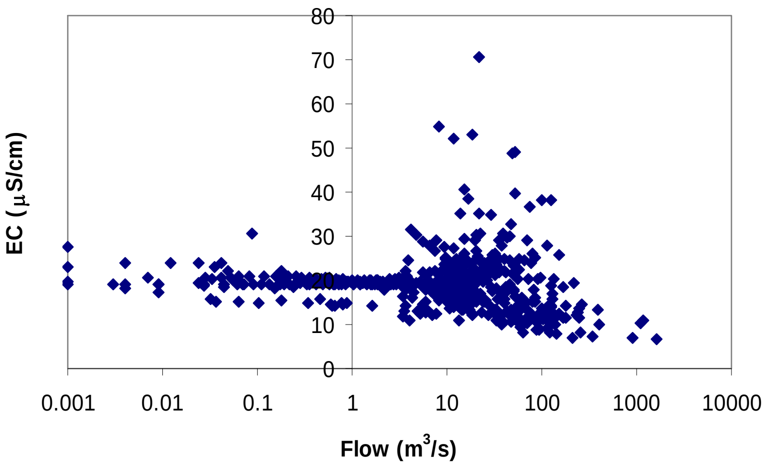

Figure 2 Distribution of daily mean EC values at the Magela Creek downstream site against flow at G8210009

Thus, for conditions where EC ≤ 50 µS/cm, the existing fitted EC-Mg relationship is considered to be very reliable for predicting Mg concentration using the continuous EC record. For conditions where EC ≥ 50 µS/cm, there is a greater level of uncertainty in the predictive power of the regression but these conditions occur very infrequently. Collection of high EC samples during the 2008–09 wet season will further address this issue.

To ensure the quality of predicted Mg data using the above EC-Mg relationships, an essentially similar procedure is used at all sites to check the reliability of the continuous EC data. This is done by comparing the EC values measured in situ or in grab samples with the instantaneous values measured by the datasonde. The continuous EC data collected form Magela Creek is considered to be reliable, with $r^2 = 0.93$ for the upstream site and $r^2 = 0.84$ for the downstream site. The continuous EC data measured in RP1 are also assessed as being very reliable with $r^2 = 0.94$. However, the fit for GC2 has an r^2 value of 0.71, indicating that the relationship at this site is not as robust, and implying a lower degree of confidence in the Mg concentrations predicted from the continuous EC trace.

Magnesium loads

The Mg concentrations predicted from the continuous EC data collected over the past wet seasons have been used together with discharge data to estimate Mg loads input to Magela Creek via the mine site tributaries as well as loads transported along Magela Creek. Magnesium load is calculated using equation (1), where *t* is time (s), *i* is a defined period of time (in this case, 10 min), [*Mg*] is instantaneous magnesium concentration (mg L^{-1}) and *Q* is instantaneous discharge (L s^{-1}).

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

By multiplying Mg concentration by the corresponding discharge at each 10 min interval and then summing each of these load increments over time, the total mass of Mg over a specified interval can be calculated.

Minesite

Point sources

The Mg concentrations predicted using the continuous EC measurements in the mine-site tributaries have been used together with the measured flows at these locations to calculate Mg loads moving down these catchment lines through the wet season (Table 2). Loads for the 2005-06 and 2006-07 wet seasons have been presented here. Complete data has not yet been provided by ERA for the 2007-08 wet season.

Table 2 Estimated Mg loads (t) exported from Coonjimba and Corridor Creeks for the 2005–06 and 2006–07 wet seasons as measured using flow data and Mg concentrations predicted from continuous EC records at RP1 and GC2, respectively

Year	RP1	GC2
2005–06	56	15
2006–07	116	18

The Mg loads in Table 2 are within the range of previously reported values for GC2 and RP1 (ERA 2006, 2007).

Diffuse sources

To provide an estimate of the Mg load potentially available for export from the LAAs via shallow groundwater flow during the wet season, the Mg load added to each of the land application areas can be calculated by multiplying the volume of applied pond water by the concentrations of Mg contained within it for each time interval over a given time series (cf Equation 1). However, the time series data for the past three years have not yet been obtained from ERA, so the total annual load has been estimated by multiplying the total annual volume of pond water applied by the mean dry season (June to November) Mg concentration of the applied waters during 2005 and 2006 (Table 3). Note that wetland treatment does not significantly change the concentrations of major ion solutes. In contrast, pond water treatment through the water treatment plant will reduce the loads of solutes, and significant volumes of pond water have been treated in the past two years. The volume of water that has been put through the treatment plant is well documented and can be accounted for in the annual volume of water that is directed to the land application areas.

Table 3 Dry season irrigation data[1] for the Magela, Jabiru East and Djalkmara LAAs

	Magela LAA			Jabiru East[2]			Djalkmara			*Total*
Year	Volume (ML)	Mean [Mg] (mg/L)	Mg Load (t)	Volume (ML)	Mean [Mg] (mg/L)	Mg Load (t)	Volume (ML)	Mean [Mg] (mg/L)	Mg Load (t)	***Mg load***
2005	438	180	79	–	–	–	262	190	50	***129***
2006	389	190	74	327	190	62	260	215	60	***196***

1 Applied volumes taken from Ranger Wet Season Reports (2005 & 2006); 2 Jabiru East first became operational in the 2006–07 wet season

Magela Creek

Magnesium loads in Magela Creek have been calculated over the past three wet seasons using the continuous EC data measured at the upstream and downstream sites and discharge measured at G8210009. The data for the 2007–08 wet season are shown in Figure 3.

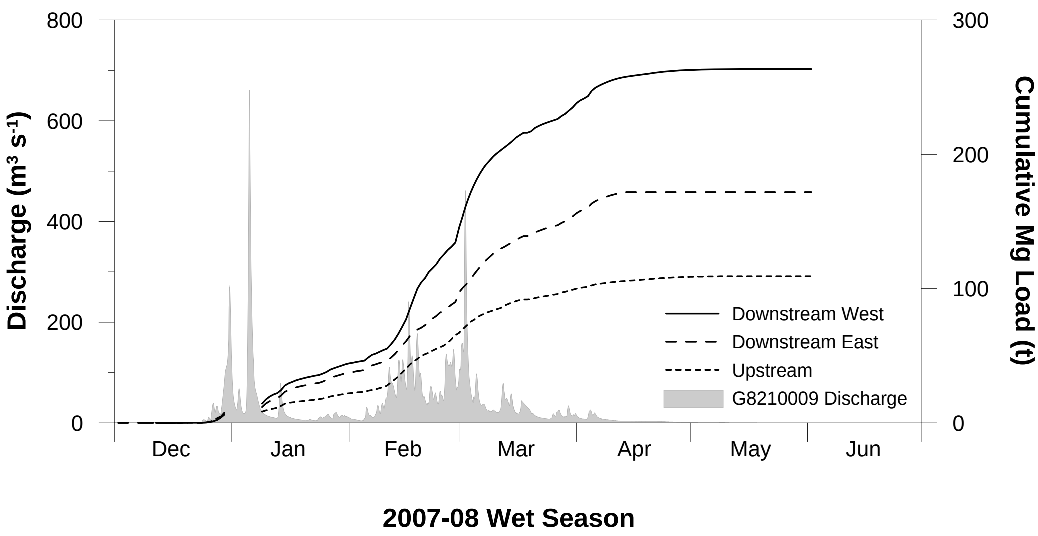

Figure 3 Cumulative Mg loads measured at the upstream and at both downstream sites during the 2007–08 wet season, along with discharge measured at G8210009

The difference between the load calculated at the upstream site and the downstream sites corresponds to input of mine-derived solutes. The difference between the loads derived for DSE and DSW shown in Figure 3 reflects the hydraulic skewing of the cross channel distribution of mine derived solutes after they enter the Creek. The continuous monitoring infrastructure is located in an anastomosed section of the stream that has three distinct channels, each separated by sand banks (Figure 4). Mine inputs to Magela Creek occur from the western bank and the channeled nature and hydraulic conditions of Magela Creek result in incomplete lateral mixing of the mine waters, especially those entering the creek from Coonjimba Billabong, a few hundred metres upstream. This incomplete lateral mixing of solutes contributed from the mine was also noted by Noller (1994). This results in the formation of a concentration gradient across the stream cross section, with the bulk of the load flowing down the west channel. Figure 4 shows the EC gradient across the cross-section of the creek measured on 1 April 2008 under low flow conditions (total flow at G8210009 was 10 cumecs).

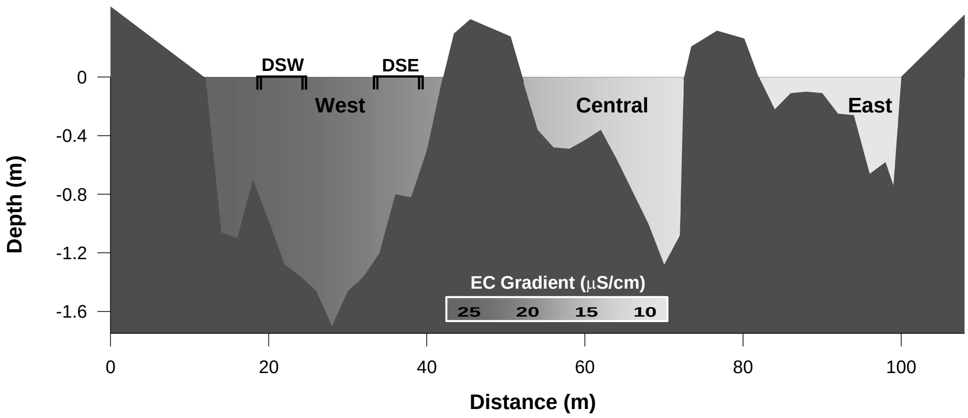

Figure 4 Cross section of Magela Creek at the downstream continuous monitoring site showing three distinct channels separated by sandbanks. The extent of shading illustrates the EC gradient (ranging from 25 µs/cm at the west bank to 10 µS/cm in the central channel) measured on 1 April 2008 when flow in Magela Creek was 10 cumecs.

Under low flow conditions (10–50 m^3/s), higher concentrations of mine-derived waters preferentially occur in the western-most channel and lower EC, catchment-derived (background) waters predominate in the eastern most channel. This potentially flow-dependent lateral distribution of mine-derived solutes (including Mg) has implications for deriving the overall solute load for the Creek as the apportioning of total stream discharge between the three channels has not been well defined as a function of flow. Since the predicted Mg concentrations are derived from EC measured in the west channel, and multiplied by flow across the full cross section, it is possible that the loads derived using this procedure are overestimates.

Load balance at Magela Creek downstream

The total annual Mg load at the Magela Creek downstream site (*DS*) in any given wet season should behave, in principle, according to equation 2. *US* is the natural background Mg load for the Magela Creek catchment upstream of the mine site. *RP1* is the Mg load input from the Coonjimba Creek catchment including RP1. *GC2* is the Mg load input from the Corridor Creek catchment and *ROC* is the Mg load input from the rest of the catchment including wet season washout of shallow groundwater from the LAAs on the mine site that are adjacent to Magela Creek (Magela LAA, Djalkmara LAAs) (Map 2). Note that LAAs on minesite tributaries (Jabiru East LAA, RP1 LAA, Corridor Creek LAA) are assumed to report to Coonjimba Creek or Corridor Creek upstream of the monitoring points RP1 and GC2, respectively, and hence are accounted for in the loads estimated at these point sources (Map 2).

$$DS = US + RP1 + GC2 + ROC \qquad (2)$$

At this point in the study, there are essentially two unknowns in the above equation. The Mg load estimated at the downstream site is a potential overestimate since it is derived using EC data from the west channel. The extent of interseasonal washout of Mg from the soil profile in the LAAs is not absolutely known, although work by Willet et al (1993) indicates that Mg has only a low affinity for the soil profile. If this is the case then the bulk of the Mg in pond water applied during a dry season is potentially available to be flushed from the soil profile in the subsequent wet season. For the purposes of constructing a load balance to compare with estimated downstream loads, it has been assumed that there is very little loss of Mg by adsorption to the soil profiles within the LAAs.

The sum of wet season Mg loads contributed to Magela Creek from individual point sources and diffuse inputs is compared in Table 4 with the solute loads calculated at the downstream west channel site. It can be seen that there is extremely good agreement between the two values, implying that use of the west channel (DSW) data results in 100% capture of the annual solute load from the mine site.

Table 4 Summary of loads imported to Magela Creek from point (upstream of the mine, RP1 and GC2) and diffuse (LAAs) inputs and loads estimated at the downstream site

Time period	Upstream	RP1	GC2	Diffuse	Sum	Downstream (*% recovery*)
2005–06	183	56	15	129	383	402 (*104%*)
2006–07	153	116	18	196	483	519 (*107%*)

This result indicates that while the loads measured at the downstream site were originally suspected to be overestimates (see continuous monitoring paper for ARRTC 20), the level of overestimation (assuming Mg is conservative) does not appear to be significant based on the data from the 2005–06 and 2006–07 wet seasons. Hence the downstream west bank site (DSW) appears to be a good location for deriving total loads of solutes exported by the mine site. Based on these findings it is concluded that it is reasonable to use the total flow discharge measured at G821009 in conjunction with the concentrations measured in the western channel to measure total solute load of Mg in Magela Creek because:

- at low flow, EC (solute concentration) in the western channel is significantly higher than the other channels however the majority of the total discharge is contained within the western channel so the load transported down the other channels is negligible;
- during high flows, the total solute concentration in the western channel decreases (inverse relationship between flow and EC shown in Figure 2) and the majority of the solute load exported at the downstream site under these conditions is from upstream inputs, thus there is good lateral mixing across all channels.

Diffuse inputs

A long-standing question has been when during the wet season the bulk of the antecedent dry season Mg is flushed out into Magela Creek. The annual Mg load estimated using the data from the downstream site on Magela Creek has been shown to account for the total annual mass of Mg exported from point and diffuse sources. If it is assumed that mass balance closure in maintained at any point in time through the season, it is now possible to solve equation 2 for the *ROC* load at any given time. Figure 5 shows the total monthly diffuse Mg input plotted against total monthly rainfall measured at the Jabiru airport.

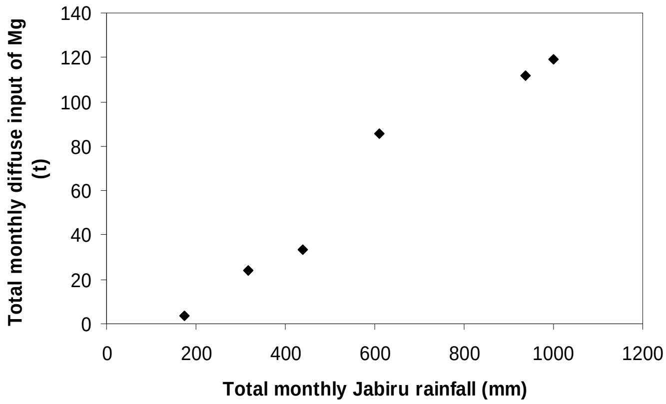

Figure 5 Total monthly rainfall measured at Jabiru airport versus total monthly input of Mg to Magela Creek via diffuse sources for the 2005–06 and 2006–07 wet seasons. Data used for this analysis is from Feb to May each year.

This linear relationship suggests that diffuse source exports of Mg to Magela Creek are a direct function of rainfall – and consistent with the previously proposed hypothesis of annual wet season flushing of the soil profiles in the LAAs. Cumulative diffuse inputs to Magela Creek over the 2005–06 and 2006–07 wet seasons are shown in Figure 6 along with cumulative rainfall measured at Jabiru airport.

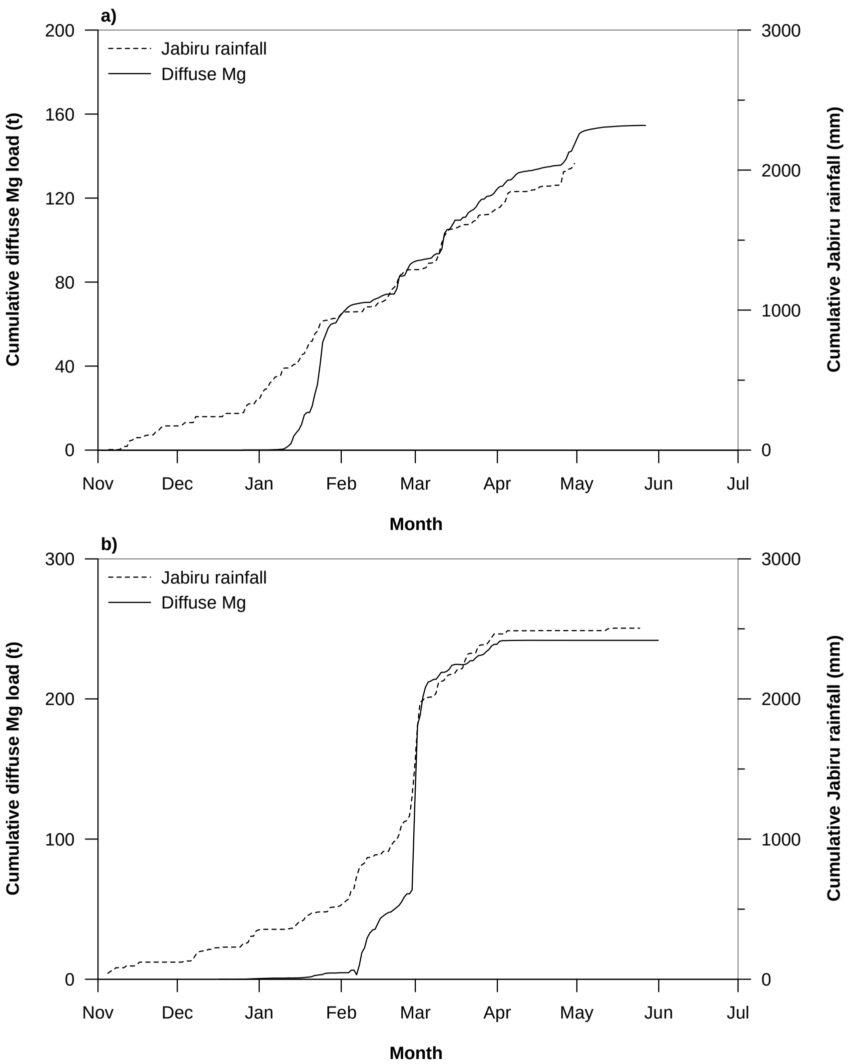

Figure 6 Cumulative diffuse Mg load (downstream load minus sum of point source inputs) entering Magela Creek over the a) 2005–06 and b) 2006–07 wet seasons, along with cumulative rainfall measured at Jabiru airport

The visual correspondence between the cumulative traces further supports the proposal that the process is driven by flushing of the salts in response to infiltration of rainwater into the soil. The time series plots in Figure 7, and the non-zero intercept in Figure 6 suggest that approximately 255 mm of rainfall (260 mm and 250 mm for the 2005–06 and 2006–07 wet seasons, respectively) is required before significant inputs of Mg from diffuse (LAA) sources start to enter Magela Creek.

Summary

Prior to consideration of the time series data for both the Magela and mine catchments there was substantial uncertainty involving the estimation of Mg loads at the downstream site.. It was considered that as a result of cross channel skewing of concentrations, use of the west channel solute data in conjunction with total stream flow may have resulted in a substantial overestimation of the actual load. However, it now appears that in practice these factors cancel out, resulting in the west channel concentration data providing quite a reasonable estimate of total solute load in Magela Creek. This in turn has enabled assessment (subject to some assumptions) of the loads exported to Magela Creek via diffuse sources.

The analysis to date suggests that the total mass of Mg applied to the LAAs each dry season is flushed into Magela Creek during the subsequent wet season. However, further work is needed to provide complete verification of this. Thus during the 2008–09 wet season the proportion of flow in the west channel will be defined as a function of stage height by carrying out cross channel hydrological gauging in the western channel. This work will be done to produce the quantitative evidence necessary to provide final confirmation of the preliminary conclusion that has been made above. The issue of the contribution of Mg from diffuse sources will be investigated in more detail by collating and analysing the available time series concentration data for Mg in groundwater bores in the LAAs along with further assessment using the continuous monitoring data.

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Development of in situ toxicity monitoring methods for Magela Creek

C Humphrey D Buckle & C Davies

The mainstay of the biologically-based, toxicity monitoring approach since 1991 has been creekside monitoring, in which a continuous flow of water from the adjacent Magela Creek is pumped through tanks containing test animals and held under creekside shelters. There are a number of practical constraints with this method, including high staff resourcing demands, reliance on complex powered pumping systems (in an area of high electrical storm activity) and vulnerability to extreme flood events.

The problems associated with the creekside program have led to an evaluation of the viability of in situ testing – deploying floating containers in Magela Creek containing the same test organism (freshwater snail, *Amerianna cumingi*) currently used for the creekside monitoring program. Potential advantages of this method include improved water flow-through and contact conditions for the test organisms, portability, the ability to run an essentially continuous biological monitoring program and greatly reduced resourcing (staff, infrastructure) and maintenance. In addition the reduced staff resourcing needs means that more staff time will be available for other components of the monitoring program, including the continuous monitoring component, and for interpretation of the data. These advantages make the in situ method appealing for future monitoring at Ranger, and potentially, for use at other minesites in the Northern Territory and elsewhere.

While in situ testing has previously been investigated as a technique for biological monitoring in Magela Creek (Annual Research Summaries 1987–88, 1988–89, 1989–90 and 1990–91), the method had remained undeveloped until recently because of perceived occupational health and safety advantages of the creekside procedure (in particular, ready accessibility and safety of staff for this land-based method). However, refinement of the techniques and improved safety and access procedures for work in the creek have allayed many of these earlier concerns.

Over a decade of creekside monitoring, test data have been obtained since 1991–92 using the established creekside protocols and infrastructure. It is thus critical to ensure that the proposed in situ method yields comparable results before it can be phased in as the sole toxicity monitoring procedure in the future. A three-year period was set aside for assessment of in situ testing, including method development. Work commenced in the 2005–06 wet season to refine the testing procedure. Concurrent creekside and in situ tests commenced during the 2006–07 wet season until the early March (2007) flood severely damaged creekside infrastructure. After this, only the in situ method could be used for the rest of the season. With the exception of the first test, concurrent testing resumed during the 2007–08 wet season. The first test relied on in situ monitoring alone, due to insufficient water depth beneath the creekside monitoring pumping stations (located in the creek channel).

The ease with which the trial in situ monitoring program was able to be reinstated after the major flood event in early March 2007 (that essentially destroyed the creekside systems for the rest of the wet season, see Supervising Scientist Annual Report 2006–2007 (Section 2.2.3, Toxicity monitoring) and be deployed as well during low flows in December 2007, clearly highlights the benefit of a method that is both independent of complex and vulnerable infrastructure, and the extremes of flow conditions in the creek.

Reproductive output (egg production) in the freshwater snail, *Amerianna cumingi*, was the main focus for in situ monitoring evaluation. A potentially important aspect of in situ method development was the nature and frequency of feeding of the deployed snails. At both the upstream and downstream locations, two feeding treatments were tested for each of the four-day tests: daily feeding (as per creekside testing) and feeding only at the commencement of the tests (once-only feeding). If the once-only feeding treatment provided comparable results to daily feeding, this would lead to an even greater reduction in the resources necessary to run in situ monitoring.

For each in situ feeding treatment and at each of the upstream and downstream sites, in common with creekside testing, there were duplicate containers each holding replicate (8) snail pairs (thus 16 pairs of snails exposed per site). Mean number of eggs per snail pair and upstream–downstream difference values for both in situ feeding methods and creekside testing are shown in Figure 1.

Differences in the mean number of eggs per snail pair amongst in situ and creekside treatments were tested for using Analysis of Variance (ANOVA). Factors tested were Treatments (‘creekside’, ‘in situ daily feed’, and ‘in situ once only feed’), Seasons (2006–07 and 2007–08), ‘Runs’ (or individual tests – nested within Season), Sites and Duplicates (nested within Treatment, Season, Site and Run). This test found significant differences amongst Treatments, Seasons and Runs ($P \leq 0.001$ for all) as well as Duplicates ($P = 0.012$). The significant ‘Runs’ and ‘Duplicates’ effect indicates variability of egg counts among the different tests, and between the duplicates in each treatment, location and test occasion. The significant ‘Season’ effect is most likely due to the higher snail egg counts in season 2006–07 (Figure 1). Neither the significant ‘Runs’, ‘Duplicates’ nor ‘Season’ effect is of relevance to the in situ treatments and creekside comparison.

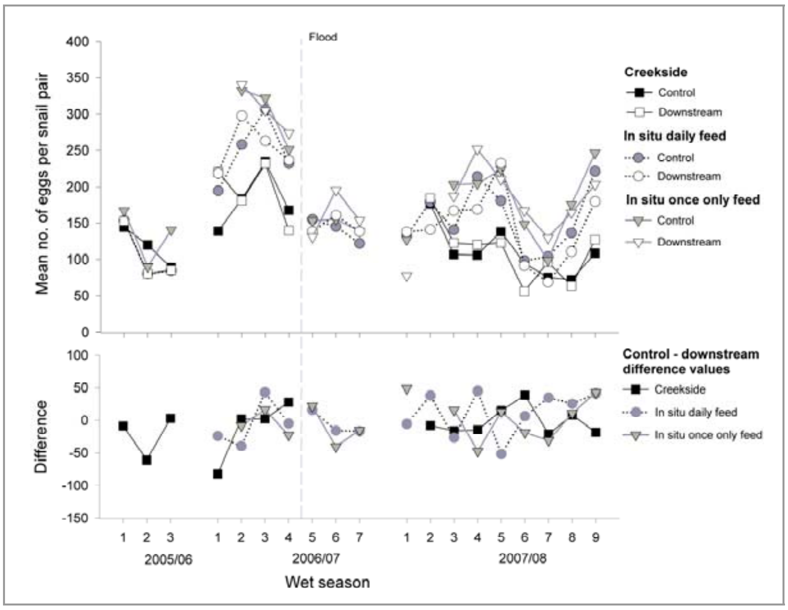

Figure 1 Comparison of snail egg production for creekside monitoring and in situ monitoring, 2005–06, 2006–07 and 2007–08 wet seasons. Note that in 2005–06, in situ testing was confined to the control site while the once-only feeding treatment was not included in the first test of 2006–07.

Of greater interest for the comparisons is the significant difference in egg production observed amongst the different testing conditions (ie ‘Treatment’ effect). The lack of interaction between ‘Treatment’ and other factors indicates the difference in snail egg production amongst treatments was consistent across all spatial and temporal scales examined. Further analysis, using a Tukey's multiple comparison test between pairwise treatments (creekside, in situ daily feed and in situ once only feed), showed that snail egg production rates for the three treatments were all significantly different from one another. The in situ once-only feed treatment had the highest mean snail egg production rates closely followed by in situ daily feed, while egg production rates under creekside conditions were considerably reduced (Figure 1).

Water quality data, as measured during the comparative tests by water temperature and electrical conductivity (EC), were examined to determine the possible influence of these variables on differences in egg production rates. Similar water temperature values were recorded for the creekside and in situ treatments (Figure 2), indicating that the creekside system, pumping intermittently to storage header tanks, provided a temperature range comparable to that in the creek. This is important because early research by Jones (1992) showed that snail egg production is strongly (and positively) linked with water temperature. EC is a good indicator of creek water quality, particularly of mine-site-derived solutes (see Continuous monitoring description above), and again, values between corresponding creekside and in situ treatments matched closely (Figure 3). These results indicate that water temperature and EC are unlikely contributors to the significant differences observed in snail egg production amongst creekside and the two in situ treatments. Furthermore, 'in situ once only feed' had higher snail egg production compared with 'in situ daily feed' even though both are subject to the same in situ flow through of water from Magela Creek.

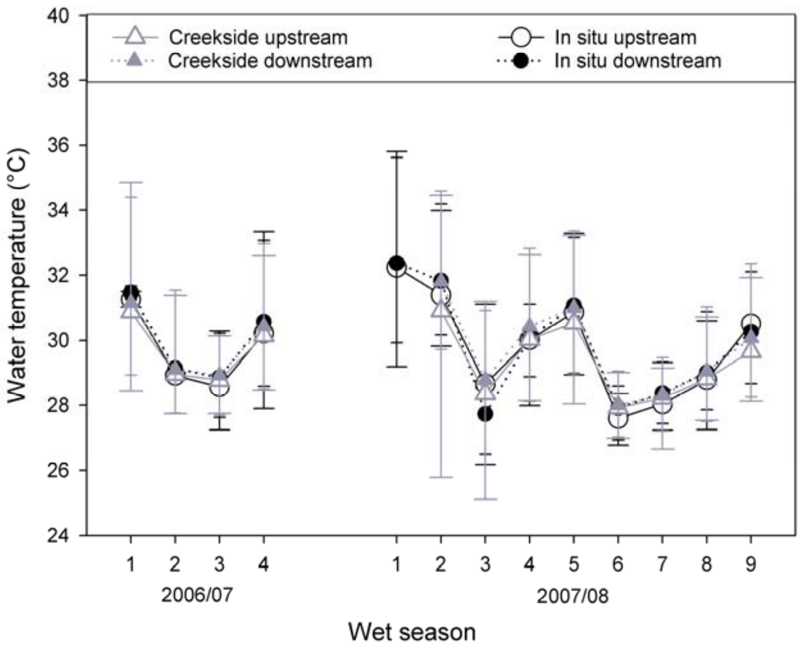

Figure 2 Comparison of test water temperatures recorded between creekside and in situ tests over the 2006–07 and 2007–08 wet seasons. Symbols and vertical bars depict the mean and range (maximum and minimum) in temperatures, respectively, at least hourly over the 96 hr test period.

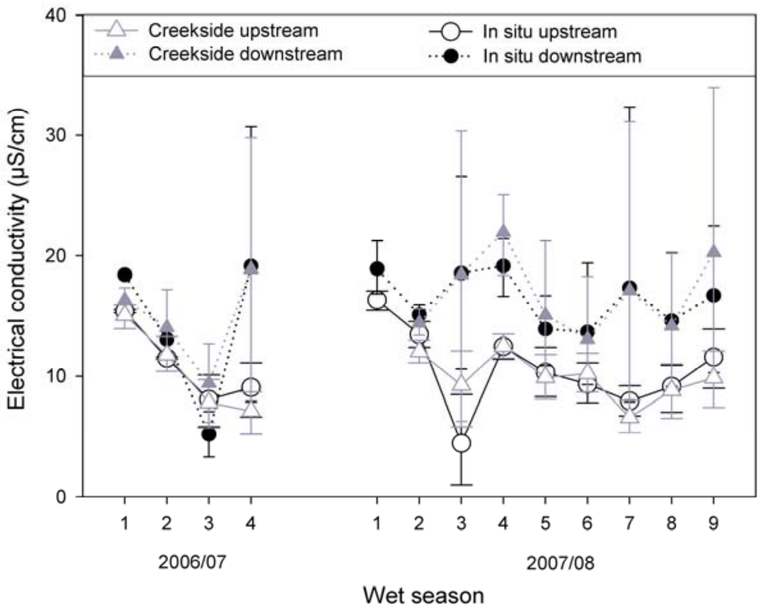

Figure 3 Comparison of test water electrical conductivity (EC) recorded between creekside and in situ tests over the 2006–07 and 2007–08 wet seasons. Symbols and vertical bars depict the mean and range (maximum and minimum) in EC, respectively, at least hourly over the 96 hr test period.

Enhanced snail egg production under in situ compared with creekside conditions is more likely due to the greater accumulation of epiphytes (attached algae) and settled detritus that was observed to occur in the in situ containers. Jones (1992) found that provision of epiphytes and settled detritus as dietary items for snails resulted in significant increases in egg production over other food sources (though this diet was not practical to be adopted as a standard feeding regime). Similar detrital and algal accumulation was not observed in creekside containers, most likely due to the considerably reduced flow-through of creek waters in the creekside test containers compared to flow-through in the in situ containers, and also to the removal of coarse water-borne particles by a filtration system contained in the creekside header tanks.

This same dietary explanation may also be the basis for the increase in egg production observed for the 'in situ once only feed' compared with the 'in situ daily feed' treatment (Figure 1). Daily feeding (of lettuce discs) to snails in creekside and 'in situ daily feed' treatments involves agitation of the egg-laying chambers in the test waters to wash off settled particles (including potential food items); this aspect of daily cleaning is absent from the 'in situ once only feed' regime, enabling epiphytes and settled detritus to further accumulate. Once-only feeding also involves much less disturbance and possible damage to snails. Any damage/disturbance to snails is known to result in reduced egg production.

While greater snail egg production was observed in the in situ treatments, the critical test response variable and end-point is the upstream-downstream difference value (Figure 1) (see 'Ranger routine stream monitoring – Toxicity monitoring in Magela Creek', pp51–52, this volume). Analysis of the upstream–downstream difference values for the creekside and in situ treatments using a three factor ANOVA, incorporating Treatments, Seasons and Runs (nested within seasons) showed no significant differences. Thus, the upstream-downstream difference values produced by the historical creekside monitoring program do not differ from those values derived from in situ daily feed or in situ once only feed over the two seasons of comparative testing.

The in situ toxicity monitoring approach will provide a more robust testing method compared with the creekside procedure since it exposes test organisms to a more continuous flow through of Magela Creek water and also reduces both staff resources and the reliance on maintenance-intensive, complex infrastructure. In addition, the simplicity of the in situ system means that it can be easily set up at other locations if needed. ‘In situ once only feed’ testing has additional advantages over ‘in situ daily feed’ of even greater reduced staff resources and improved testing conditions (that is, less disturbance) for the test organism.

Based on the conclusive results presented above the ‘in situ once only feed’ deployment of the freshwater snail, *Amerianna cumingi*, will replace the creekside toxicity testing procedure from the 2008–09 wet season onwards.

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

J Brazier, A Bollhöfer, C Humphrey & B Ryan

Background

The Supervising Scientist Division develops radiological, chemical, ecotoxicological and biological techniques to monitor and assess impacts upon ecosystems and humans that arise from uranium mining activities in the Alligator Rivers Region. An important component of the monitoring program for the Ranger mine measures uptake of selected metals and radionuclides in freshwater mussels, *Velesunio angasi*. Among the suite of radionuclides measured, radium-226 (^{226}Ra) is of particular interest as ^{226}Ra in mussels has been identified as the major contributor to radiological dose from ingestion of bush foods by local indigenous people. This comparatively large contribution occurs because (a) freshwater mussels are an integral component of the diet of the Mudginberri community located downstream from the mine (b) the high concentration factor of 19 000 for radium in freshwater mussels, and (c) the large dose conversion factor for ^{226}Ra of 0.28 µSv·Bq^{-1}.

Any significant increase in metal and radionuclide concentrations in aquatic biota measured through time (or compared to an appropriate reference site) also provides the potential for early warning of a developing issue with bioavailability of mine-derived solutes. Consequently, the ongoing measurement of metal and radionuclide concentrations in mussels provides both ecosystem protection and human health protection functions.

Mussels are routinely collected from Mudginberri Billabong at the end of each dry season. They are obtained from the inlet of the billabong since this is where Aboriginal people typically collect their mussels. Sandy Billabong in the adjacent Nourlangie Creek catchment is sampled as a control site. It has been shown that radium activity concentrations in mussels from Mudginberri Billabong are higher, age-for-age, than in mussels from Sandy Billabong (see Figure 2, in ‘Bioaccumulation in fish and freshwater mussels from Mudginberri Billabong’, pp53–56, this volume).

To test the hypothesis that Ranger mine is not contributing to these higher loads, and that the Magela Creek catchment has naturally higher concentrations of ^{226}Ra compared with the Nourlangie Creek catchment, mussels, sediment and water were collected in May 2007 along Magela Creek from well upstream of the mine down to Mudginberri Billabong. The following sites were sampled: the outlet of Bowerbird Billabong well upstream (~ 20 km) of the mine, along the Magela Creek channel immediately upstream and 5 km downstream of the mine (MCUS, G8210009 respectively), at the entry zone of Georgetown Billabong into Magela Creek (GTC), and in Mudginberri Billabong located 12 km downstream of the mine. Figure 1 shows the location of all sites sampled in this study, as well as three other sites CJB, GTB and Corndorl referred to in the text from other studies. In addition to radium activity concentrations, other selected parameters including calcium and uranium and lead isotope ratios, were also measured in mussel soft tissues, sediment and water from each of the sites.

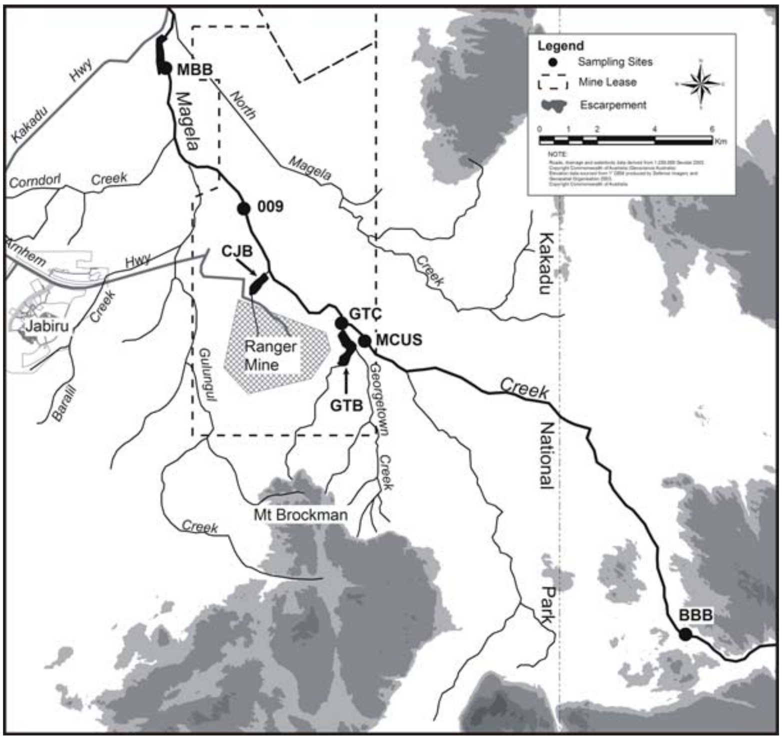

Figure 1 Location of sampling sites along Magela Creek. BBB, Bowerbird Billabong; MCUS, Magela Creek upstream; GTC, Georgetown confluence; G8210009, Magela Creek downstream; MBB, Mudginberri Billabong; GTB, Georgetown Billabong; CJB, Coonjimba Billabong

Potential effects of variation in mussel weights upon measured contaminant concentrations

Mussels were collected by hand and immediately placed into acid washed containers holding water from the sample location. They were then transported to the Darwin laboratory where they were purged over 6–7 days, before being measured for length and width, weighed and dissected to remove the flesh. Samples were oven dried and reweighed to determine the dry weight. The age of each mussel was determined by placing the shell over an incandescent light source and counting the number of annual growth bands (annuli). The dried and ground flesh of each mussel was combined according to age class and site, and the average dry weight per age class determined.

Figure 2 shows the dry weight of mussels plotted against mussel age for four sampling sites along the creek, and the average weight and age data from the May 2007 and all previous collections, from Mudginberri Billabong. Also shown is the average weight and age data from all previous Sandy Billabong collections. The rate of mussel growth and the maximum theoretical mass that a mussel could achieve at the end of its life (m_∞ – marked by the horizontal dashed line in each of the panels of Figure 2) differ among the sites. For example, mussels from Bowerbird Billabong grow slower but reach a larger mass compared to Mudginberri or Sandy Billabong mussels. Mussels from the latter two billabongs show similar growth patterns and reach a theoretical maximum mussel dry mass of ~1 gram.

The results shows that the rate of mussel growth in the same creek system is quite variable, which in turn may influence radionuclide and metal uptake. Moreover, because mussel soft tissue weights are known to vary seasonally and even inter-annually for populations from the same site in response to temporal changes in physiology, food availability and other ambient conditions, the plots depicted in Figure 2 will also change accordingly.

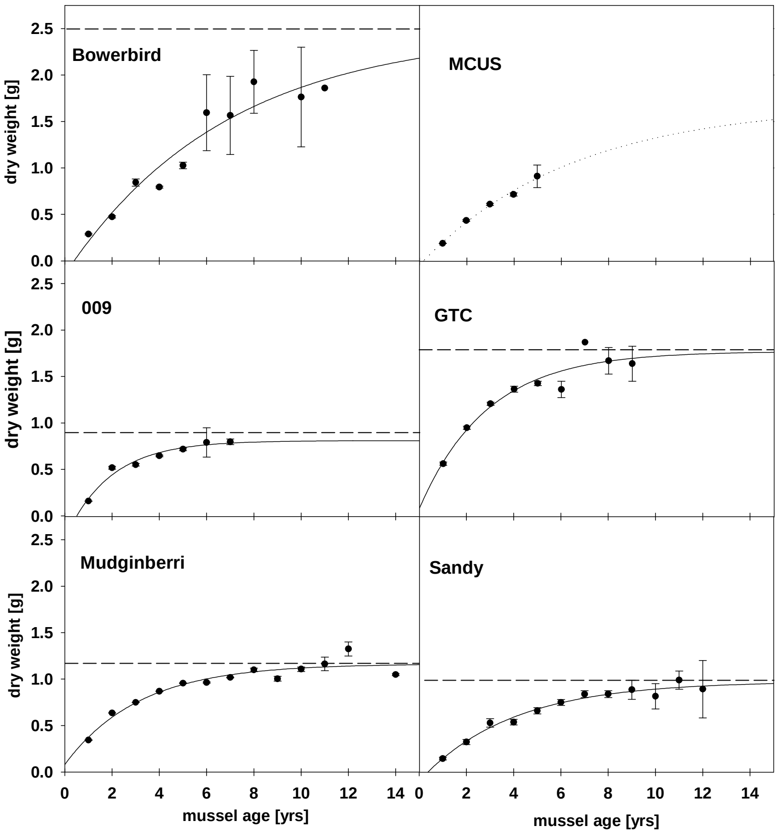

Figure 2 Mussel dry weight plotted versus age at the sampling locations along the Magela Creek channel and from the control site, Sandy Billabong. The solid line represents the results of a fit assuming that the mussel weight-age relationship follows a form of the von Bertalanffy Growth Equation. Dashed lines indicate the ultimate mussel dry mass, m_∞ (not enough data for MCUS).

Population or temporal differences in concentrations of metals and radionuclides in mussels may simply reflect physiological or environmental conditions unique to a site, which in terms of impact assessment, may be unrelated to potential sources of contaminant uptake. Consideration must be given to mussel physiology, time of year for sampling and other ambient conditions when choosing a sampling regime (including control sites), as the measurement of metal and radionuclide concentration alone may be insufficient to be able to unambiguously infer a mine influence at the exposed site. A sufficient time series of data are required from sampling sites to make reliable inferences about impacts.

Radium in mussels and sediments

Each mussel age class was measured for the radioisotopes of lead (^{210}Pb), thorium (^{228}Th) and radium (^{226}Ra & ^{228}Ra) by gamma spectrometry. Mussels $\leq$ 1 year of age, or an age class with similarly insufficient mass for analysis by gamma spectrometry, were analysed by alpha spectrometry for ^{226}Ra. Measurement of the same radioisotopes was made on sediment and water samples. ^{226}Ra and ^{228}Ra are members of the uranium and thorium decay series, respectively. Hence the activity ratio of the two isotopes provides a measure of the relative contribution of uranium and thorium-rich sources, respectively, to the radium activity concentration in a sample. The lower the ^{228}Ra/^{226}Ra activity ratio in sediments or mussels, the higher is the contribution of radium derived from a uranium rich source.

Figure 3 shows the ^{228}Ra/^{226}Ra activity ratio measured in sieved (<63 µm) sediment, and the average ratios in 1–4 year old mussels collected along Magela Creek. ^{228}Ra/^{226}Ra ratios in sieved sediment are slightly lower than ratios in total sediment at all sites, although the difference in Mudginberri Billabong is small due to the relatively larger proportion of fine silts and clays present in these billabong sediments (12%) compared with Magela Creek (3–4%).

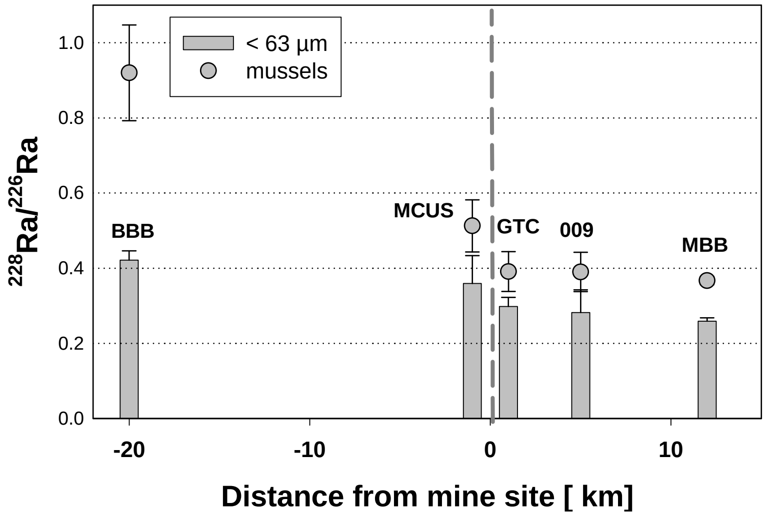

Figure 3 $^{228}Ra/^{226}Ra$ activity ratio measured in sieved (<63 µm) sediment and in mussels collected along Magela Creek. BBB, Bowerbird Billabong; MCUS, Magela Creek upstream; GTC, Georgetown confluence; G8210009, Magela Creek downstream; MBB, Mudginberri Billabong

The $^{228}Ra/^{226}Ra$ activity ratio in the <63 µm sediment fraction declines gradually along the catchment gradient. An even more pronounced decline is seen for the mussel flesh. The decline is gradual, rather than a step function that would otherwise be expected for a point (U mine) source. This indicates a small but steadily increasing relative contribution of a uranium-rich source relative to thorium along the catchment gradient.

In contrast to the $^{228}Ra/^{226}Ra$ activity ratios, the radium activity concentrations measured in mussel flesh and sediments exhibit no clear pattern. Radium loads are, actually, highest in mussels collected from the outlet of Bowerbird Billabong in the upper reaches of the catchment, well upstream of possible mining influence. Comparatively higher total radium concentration in water and sieved sediment at that sampling location compared to other sampling sites along the creek channel, and the different growth pattern observed for mussels collected at the outlet of Bowerbird Billabong, may explain this finding. Future work will investigate whether similar variations occur not only at a catchment-wide scale but within a single waterbody, such as Mudginberri Billabong.

Uranium in mussels and sediments

Composited dried mussel flesh from each age class was acid digested and measured for uranium concentration by Inductively Coupled Plasma Mass Spectrometry (ICPMS). In addition, sediment (from whole and < 63 µm fraction) was digested in a weak acid solution and analysed for uranium concentration using ICPMS. This extraction procedure indicates the proportion of uranium that is potentially bioavailable.

In contrast to radium, there is no age dependency for concentration of uranium in mussel flesh. Uranium concentrations are generally higher in mussels downstream of the Ranger mine compared to upstream of the mine, and are highest closest to the mine at GTC. Comparison of average uranium concentrations in mussel tissue (1–5 year olds) from this collection with uranium data from 1980 (Allison & Simpson 1989) show similar levels at Mudginberri and Bowerbird Billabongs (Figure 4).

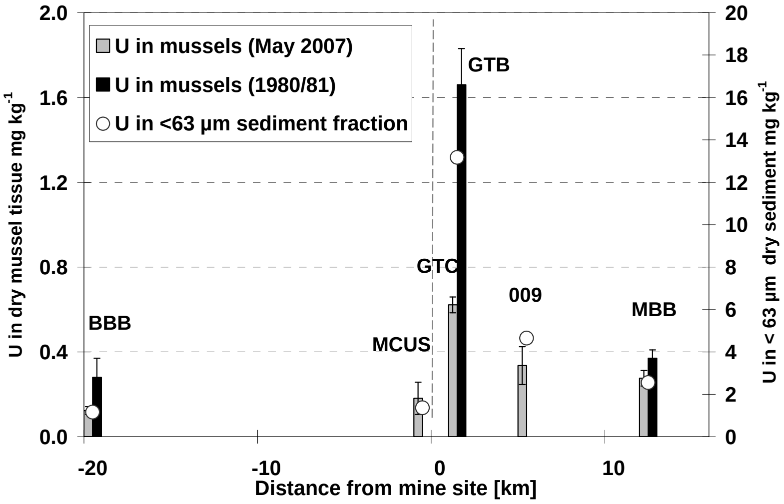

Figure 4 Comparison of May 2007 uranium concentrations in mussels along Magela Creek, upstream and downstream of the mine with historical (1980) data. Uranium concentration in the < 63 μm fraction that is potentially bioavailable (1 M HCl acid digest) is also shown.

The concentration of U in mussels in Georgetown Billabong in 1980 is much higher than for May 2007, but it should be noted that the recent collection was not from the billabong but instead from the channel outlet which is influenced to a much greater extent by the adjacent Magela Creek waters unaffected by the Ranger mine. Since the 1980 analyses were a pre-mining baseline assessment, it suggests that the higher uranium concentrations found in Georgetown Billabong mussels at that time were related to natural erosional contributions from the surface expression of ore body number 1 located in the Georgetown Creek catchment.

Stable lead (Pb) isotope ratios in mussels and sediments

^{206}Pb and ^{207}Pb are the stable end-members of the uranium decay series (^{238}U and ^{235}U, respectively) while ^{208}Pb is the stable end-member of thorium decay (^{232}Th). The ratio of the abundance of the ^{206}Pb and ^{208}Pb isotopes normalised to ^{207}Pb provides a very sensitive measure of the relative contribution of uranium and thorium-rich sources, respectively, to the total lead concentration in the sample. Similar to the radium activity ratios, high ^{206}Pb/^{207}Pb and low ^{208}Pb/^{207}Pb ratios, respectively, indicate a contribution of a uraniferous source. These lead isotopes are physically and chemically alike and therefore are not altered differentially by environmental processes. As a consequence, any changes in isotopic composition are a result of mixing of lead from different sources and so, if the source lead isotopic compositions are known, then it is possible to ascribe the extent of contribution of each of the upstream sources to samples collected from a particular site.

Lead isotope ratios were measured by ICPMS on dried tissue from mussels collected along the Magela catchment and compared to the isotope ratios of the < 63 μm sediment fraction. In addition, isotope ratios were measured in sediments (< 63 μm) from Coonjimba Billabong (which receives water from Retention Pond 1 and then drains into Magela Creek just upstream of G8210009), Gulungul Billabong (which drains into Magela downstream of G8210009 and eventually into Mudginberri) and Corndorl Billabong (which also flows into Magela Creek between G8210009 and Mudginberri Billabong). The results are summarised in a three isotope plot, showing ^{206}Pb/^{207}Pb versus ^{208}Pb/^{207}Pb isotope ratios, in Figure 5.

Figure 5 illustrates that the isotopic composition of lead in mussels and sediment along the Magela catchment is largely a mix of lead from two sources: lead with an upper Magela catchment signature, reflected in Bowerbird Billabong data, and lead with a signature similar to the Ranger orebody. The contribution from the Ranger orebody is amplified in the mussels, as shown by generally higher $^{206}Pb/^{207}Pb$ and lower $^{208}Pb/^{207}Pb$ isotope ratios compared to the respective sediment.

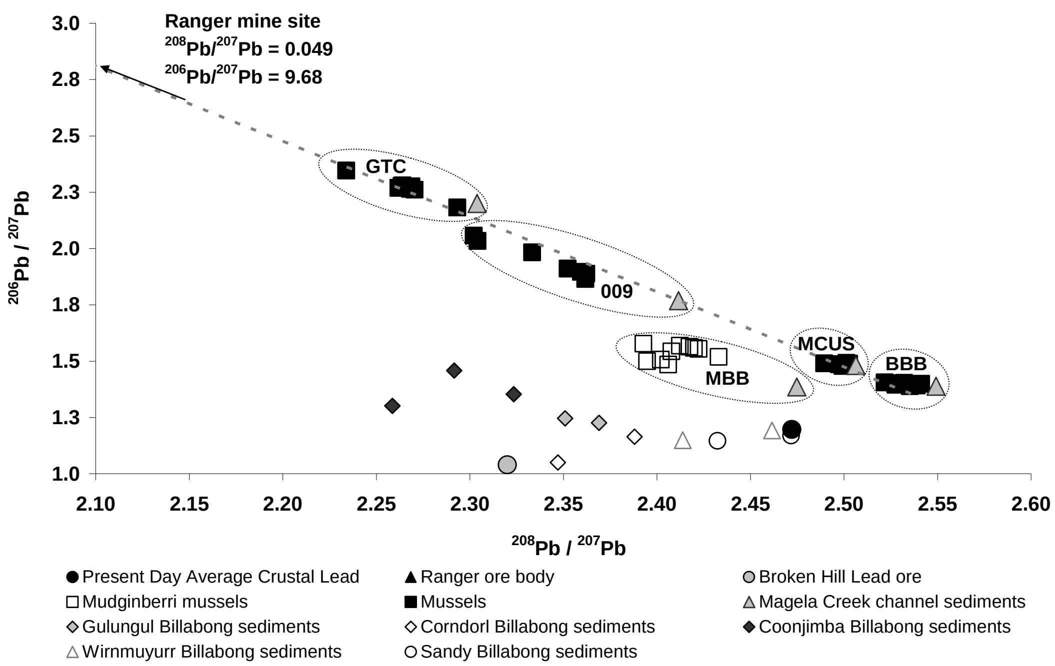

Figure 5 $^{206}Pb/^{207}Pb$ plotted versus $^{208}Pb/^{207}Pb$ isotope ratios measured in mussel tissue and < 63 μm sediment fraction (1 N HCl digest) from Magela Creek. The dashed trendline assumes mixing of radiogenic lead with the Ranger ore signature (not shown as outside of axes range) and lead with an Upper Magela catchment signature (BBB). Each site's sediment and mussel lead isotope signature is circled with the site label.

An additional lead source appears to influence the Mudginberri mussels and sediments, causing an offset from the trendline towards lower $^{206}Pb/^{207}Pb$ isotope ratios. Potential sources contributing to this lower ratio are in the Gulungul or Corndorl catchments, exhibiting lead isotope ratios closer to common lead with a Broken Hill lead ore signature. Importantly, sediments and mussels at MCUS exhibit a higher $^{206}Pb/^{207}Pb$ isotope ratio than at Bowerbird suggesting that the orebody signature is increasing gradually along the catchment downstream, and hence is natural to the catchment, rather than the result of a new point source caused by the operational Ranger mine.

Conclusion

Variations in radium activity concentrations found in mussels along the catchment are driven by a range of factors unrelated to current mining activity at Ranger (eg growth rates, body weights, competing chemistry, local geology), the precise contribution of which are yet to be fully understood. Consequently, a simple upstream-downstream comparison of radium activity concentrations in mussels is not an appropriate method to unambiguously detect a mine signal. Activity and isotope ratios, respectively, are better suited for source identification.

This study has found that changes in $^{228}Ra/^{226}Ra$ activity and lead isotope ratios along Magela Creek are largely natural features of the catchment, rather than a mining-related feature. In particular, the uranium concentrations in the mussels at all sites, are comparable to pre-mining values from 1980; and the gradual increase of uraniferous signature in sediments and mussels along the creek channel implies a catchment-wide contribution rather than a contemporary mining-related impact. Overall, the results show that the aquatic environment is not being impacted by mining activities.

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Part 2: Ranger – Rehabilitation

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

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

Introduction

The bulk of this project, as it relates to Ranger landform evolution simulations, was reported at the 20th ARRTC meeting in October 2007. During 2007–08 a report was completed describing the progress of the co-operative study conducted with Professor Tom Coulthard and the University of Hull to utilise the CAESAR (Cellular Automaton Evolutionary Slope and River model; Coulthard 2001, Van de Wiel et al 2007) landscape evolution model to assess the impact of extreme rainfall events on the proposed Ranger rehabilitated landform.

In addition, during 2007–08 further work was undertaken in collaboration with Dr Greg Hancock (University of Newcastle, NSW) using the CAESAR model to assess the impact of extreme rainfall events on a natural sub-catchment of Tin Camp Creek (TCC) in western Arnhem Land. The catchment of TCC has a similar geology (ie schist-based) to that of the pre-mining land surface at the Ranger uranium mine and it may provide a suitable analogue for specifying the long-term (closure critertia) erosion rates to be expected following rehabilitation of the site (Uren 1992) and Moliere et al (2002) showed that erosion rates and landform morphology at TCC were a good estimate of what the Ranger landform may evolve towards for slopes where channels have developed where flow has concentrated.

The Siberia landform evolution model has been used to provide average long-term erosion and gully incision rates of rehabilitated mine landforms within a catchment with varied surface treatments (Evans & Willgoose 2000, Hancock et al 2008, Lowry et al 2006). However, the average long-term erosion assessments conducted prior to these studies have not implicitly addressed the impact of an extreme rainfall event or a series of events comprising an ‘extreme’ wet season. It is therefore important to assess what the impact of extreme rainfall events on a rehabilitated landform will be and to assess the possibility of buried contaminants becoming exposed.

CAESAR uses individual rainfall and runoff data enabling the impact of individual large rainfall events to be assessed, a critical attribute given the long times required to contain mine tailings, and the probability that one or more very extreme rainfall events will occur over this time frame. Testing of the proposed design parameters for the constructed landform for extreme events has assumed greater importance given the probability of an increase in the frequency of intense rainfall events as a consequence of climate change.

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Progress and results to date

The progress and results of the collaborative research conducted with Professor Coulthard on the proposed Ranger landform were described in the report presented to ARRTC in October 2007 and are in Evans et al (2008). The study found there is a high level of sediment loss and fluctuation of erosion rates in the initial years as the new landform finds equilibrium. This is the phase of catchment conditioning simulated by CAESAR as fine sediment is removed from the catchment, drainage lines are incised, particle size distribution of the surface material is adjusted (self–armouring)and vegetation grows, leaving coarser material in the thalweg[3] of drainage lines.

Simulations, using the Jabiru airport 22-year rainfall record, showed that the Ranger landform catchment used in this study for erosion simulations takes about 5 years to undergo this conditioning. Similar observations of catchment conditioning or surface armouring have been observed in field erosion studies at mine sites and for natural terrains in the ARR (Moliere et al 2002).

Previous studies in the area give a range of denudation rates for waste rock of -2 mm y^{-1} to 7 mm y^{-1} with a median of 0.04 mm y^{-1}. Denudation rates of -1.3 mm y^{-1} to 1.3 mm y^{-1} with a median of 0.02 mm y^{-1} have been measured for natural land surfaces in the region (Erskine & Saynor 2000). In 2007, a >> 100 y return interval rainfall event was recorded in Jabiru. For model simulations where the ‘extreme’ rainfall year 2006–07 was applied for the first year after rehabilitation, a 9 mm lowering of the land surface was predicted. A lowering of 6 mm was predicted when the 2006–07 rainfall record was applied in year 10 of the 22-year rainfall record simulations. After this initial phase, simulated denudation rates remain at an average of 0.2 mm y^{-1}, within a range of -0.05 mm y^{-1} to 1 mm y^{-1}.

A range of rainfall scenarios over a 1000-year period were simulated for the TCC catchment by Dr Hancock using the Jabiru airport 22-year rainfall record. The other key input parameters used in the simulations included particle size data collected from two sites in the catchment of TCC (QT1 and QT3), and a pit-filled, hydrologically-corrected digital elevation model with a resolution of 10 metres for the TCC catchment. Four different rainfall scenarios were generated, representing:

(a) 22 years of complete rainfall (1972–2006). This data set was termed the ‘Standard’ rainfall data set for the simulations. This data is typical of rainfall up until the 2006–07 wet season and represents average rainfall conditions for the region.

(b) 22 years of rainfall added end to end to produce a 44 year record with the 2006–07 data added at 44 years to produce a total 45 year synthetic record. This was termed the Enhanced Rainfall 45 year (ER45) data set and represents a return interval of approximately 1:50 years for the 2006–07 season.

(c) The complete 23 year rainfall record including the 2006–07 rainfall data. This data set was termed the Enhanced Rainfall (ER23) data set and represents all the currently available rainfall data. The ER23 data set represents a return interval of approximately 1:20–25 years for the 2006–07 season.

(d) The 22 year rainfall record with the 2006–07 wet season data included at both 11 years and 22 years producing a rainfall record 24 years long. This was termed the ER11 data set and represents a return interval of approximately 1:10 years for the 2006–07 season.

[3] The continuous line connecting the lowest points along a stream bed.

The above four rainfall data sets were added end to end to produce a continuous 1000 year record for input into the CAESAR model.

As observed with the simulations of the Ranger landform, the CAESAR model produces high sediment transport rates in the first 10–20 years as surface roughness in the digital elevation is removed and smoothed as well as the soil particle size distribution is sorted across the catchment according the topography and hydrology. There was no variation in surface condition across the catchment The model was initially run using the standard rainfall data and the particle size distributions from the stwo sites (QT1 & QT3) for 44 years as this allowed 2 cycles of the ‘Standard’ rainfall data. The resultant digital elevation model and grainsize distribution for the QT1 and QT3 particle size data was then used as the initial input to all simulations.

Overall the QT1 simulations had similar overall sediment output with the QT3 and ER11 rainfall producing the most (Table 1). For both soil data sets total sediment output increased as the inclusion of the 2006–07 rainfall data became more frequent. In general, the simulations using QT3 parameters eroded approximately 1.5 times more material than the QT1 sequence for the same rainfall input. For all simulations there were years when there was no sediment output from the catchment represented by a negative depth of erosion (Table 1). The variability in annual erosion (standard deviation) also increased with the inclusion of the 2006–07 rainfall year sequence.

Table 1 Minimum, maximum and average depth of erosion (metres) within the catchment over 1000 years. Negative values represent deposition.

	standard	QT1 soil		
		ER45	ER23	ER11
minimum	-0.28	-0.30	-0.30	-0.30
maximum	1.65	1.58	1.80	1.79
mean	0.02	0.02	0.02	0.02
SD	0.10	0.10	0.11	0.12
		QT3 soil		
	standard	ER45	ER23	ER11
minimum	-0.31	-0.42	-0.30	-0.30
maximum	1.72	1.73	1.75	1.81
mean	0.03	0.03	0.03	0.04
SD	0.13	0.14	0.14	0.17

For all simulations annual sediment output starts higher and then declines. Examination of annual sediment output for each simulation demonstrates that each has a unique pattern as there is considerable variability in sediment output over the 1000 year modelled period. All simulations have periods of both low (near zero) and high sediment output. In general, lower peaks occurred using the Standard rainfall than those runs that included the 2006–07 rainfall data. Of particular interest are the periods of increased output particularly for the QT3 parameters using the ER11 rainfall at around 400 years.

There was little difference between simulations in terms of erosion and deposition depths (Table 1). All areas of the catchment were subjected to both erosion and deposition (Figure 1).

The major differences between simulations occurred along the drainage lines with increased depth of erosion occurring along the length of the channel as well as extending further up the hillslope. In terms of deposition, the majority occurred at the bottom of hillslope for first order stream catchments. Maximum erosion depth occurred along the main drainage line in the central region of the catchment.

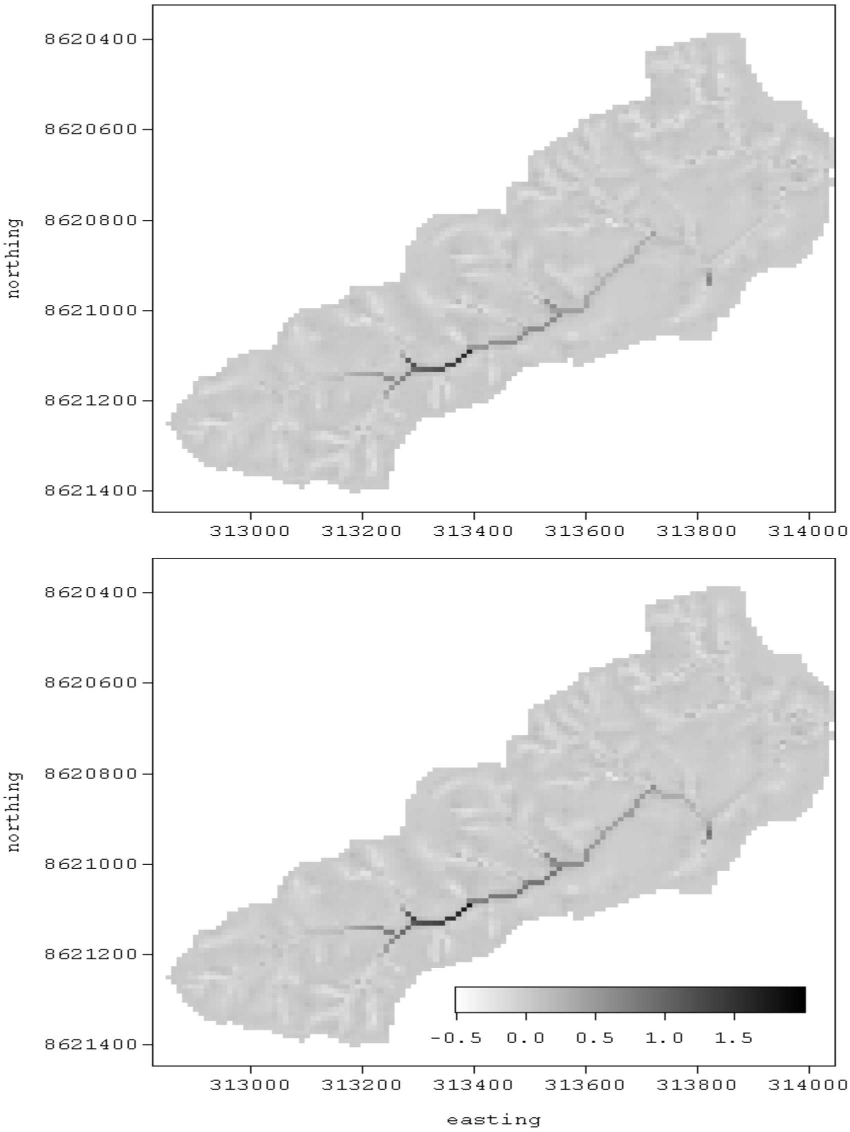
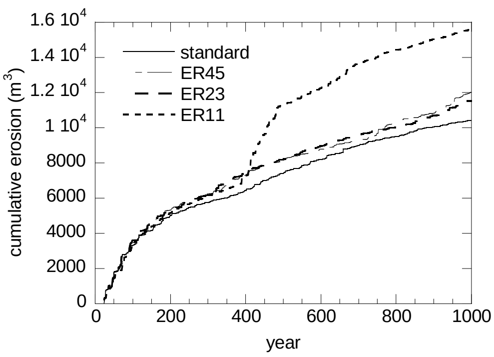


Figure 1 Erosion and deposition patterns for the Tin Camp Creek catchment using enhanced rainfall ER11 (top) and standard rainfall (bottom) and QT1 soil particle size after 1000 years. All dimensions are metres. The erosion key dimensions are metres and negative values represent deposition while positive values equal erosion.

The results demonstrate that there is a unique interaction between both particle size distribution and rainfall. Different particle size distributions produce different sediment transport rates and patterns for the same rainfall. This demonstrates that the relationship between particle size distribution and rainfall is not linear as there appears to be periods where there is enhanced sediment output followed by periods of lower output.

In terms of impact of increased rainfall and storms on sediment output, it is clear that the more frequent returns of the 2006–07 rainfall increases erosion rates. An examination of cumulative sediment output (Figure 2) demonstrates that for both soil data sets erosion follows roughly the same pattern but at approximately 400 years for the QT1 soil and ER23 and ER11 simulations increased erosion while there is a markedly strong increase for the QT3 simulation using ER11 rainfall.



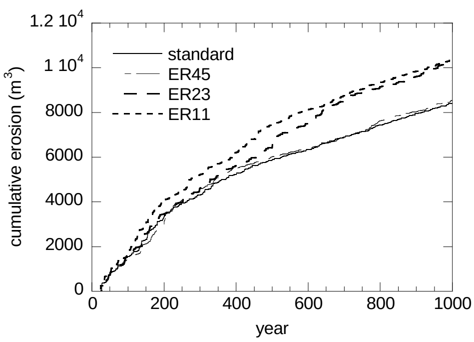


Figure 2 Cumulative erosion for the Tin Camp Creek catchment using the CAESAR erosion model and QT3 (a) and QT1 (b) parameters Standard, ER45, ER23 and ER11 rainfall

Steps for completion

Whilst this work shows great promise to bring new insights into the longevity of mine rehabilitation, there are some steps that need to be taken. These are:

1 Develop model capability to incorporate spatial variability in surface material types and vegetation distribution. Little further simulation work can be done until this step is completed.

2 Test the effects of Digital Elevation Model (DEM) resolution,

3 Test various capping material types such as the laterite mixed proposed for the Ranger vegetation trial plots,

4 Compare long-term erosion rates between CAESAR and Siberia models, and

5 Integrating and evaluating the importance of vegetation on landform stability.

Research programs have been developed co-operatively with both Dr Coulthard and Dr Hancock but competing projects (Ranger landform trial) have taken priority.

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

GR Hancock[1], KG Evans & JBC Lowry

Introduction

Erosion models are being used to evaluate the erosional stability of the ERA Ranger mine final landform. The parameter values used in these models are derived through monitoring sheet flow erosion plots under natural rainfall events. There has been some validation against catchment property field data over the long-term using Tin Camp Creek sites which showed that the model could capture catchment form. However there is a need to ensure that the parameter values currently used approximate natural gully development rate.

This is a joint study between Dr Greg Hancock of the University of Newcastle as principle researcher and initiator of the study and ***eriss***. This field study is on an undisturbed site within the Tin Camp Creek catchment in western Arnhem Land near Nabarlek. Soil erosion rates are being measured by the ^{137}Cs method to determine background erosion rates over the last 40–50 years. A network of erosion pins has been installed and annual erosion rates have been and can continue to be measured. Further, an extensive series of gullies exist which are being monitored for their movement on an annual basis which will be compared with gully development predictions of landform evolution models. Soil carbon has also been measured to assess if a relationship exists between soil carbon content and rate of erosion.

Progress and results to date

The study site (Fig 1) is located in the seasonally wet/dry tropical environment of northern Australia, with an annual average rainfall of approximately 1400 mm, mostly falling in the wet season months from October to April. Short, high intensity storms are common, consequently fluvial erosion is the primary erosion process (Saynor et al 2004). The studied undisturbed drainage basin has been unaffected by European agriculture or pastoral activities, but often experiences fire during the dry season.

Caesium-137 and erosion pins

An assessment of slope erosion was carried out using (1) the fallout environmental radioisotope caesium-137 (^{137}Cs) as an indicator of soil erosion status; (2) two numerical models (Siberia and the Revised Universal Soil Loss Equation – RUSLE); and (3) erosion pins.

Two transects (1 and 1a marked on Fig. 2) were sampled for ^{137}Cs in 2002 and 2004, and two models were used to convert ^{137}Cs measurements into soil loss estimates. The theoretical Profile Distribution Model (Walling & He 1999, Walling et al 2002) used to derive net soil loss rates from the ^{137}Cs data gave net soil losses between 50 and 60 t ha^{-1} y^{-1}, while an Australian empirical model (Elliott et al 1990) for uncultivated soils produced estimates between 7 and 8 t ha^{-1} y^{-1}. RUSLE gave estimated soil losses for the two transects of

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approximately 10 t ha^{-1} y^{-1}, while the Siberia model produced values between 0.5 and 2 t ha^{-1} y^{-1} for the transects and values between 4 and 11 t ha^{-1} y^{-1} for the total catchment. Average net soil losses of 14 and 15 t ha^{-1} y^{-1} for the total catchment and slopes, respectively, were measured by erosion pins. These results indicate that Siberia predicts similar erosion rates (same order of magnitude) to those determined by other methods.

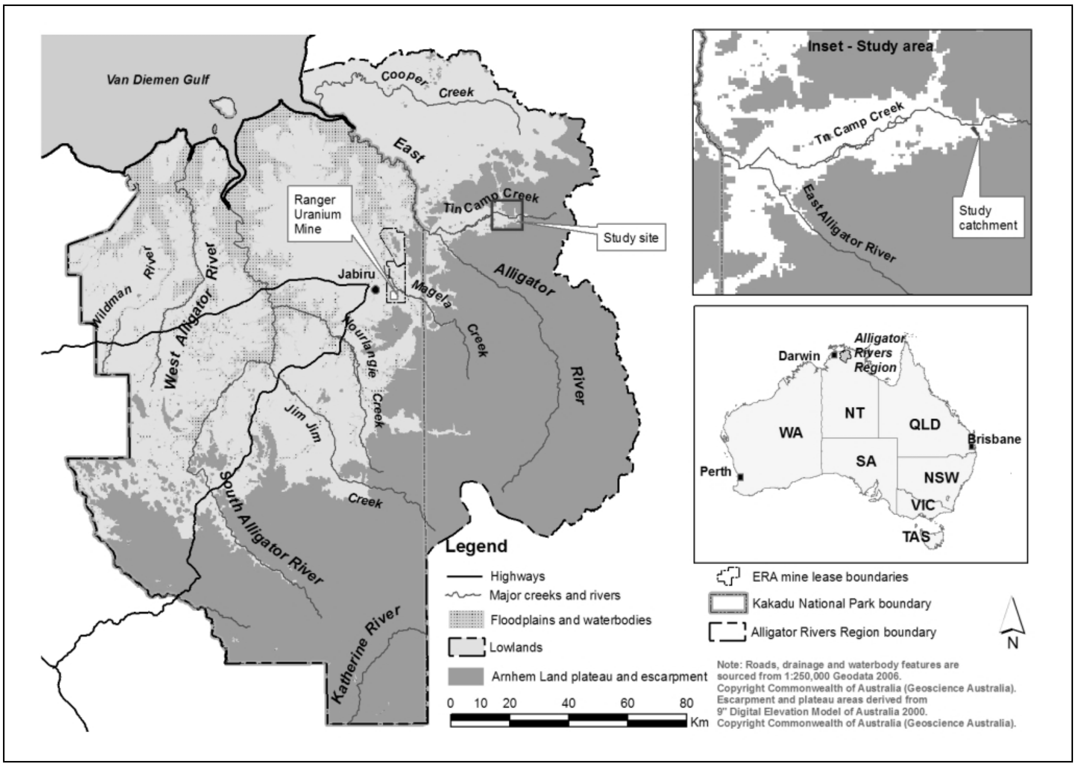

Figure 1 Location of study the Tin Camp Creek (TCC) site and Ranger mine

Excluding the soil loss rate derived from the Profile Distribution Model, the soil losses in the catchment were greater than for other transects in the Northern Territory and similar to rates in the Kimberley region, (measured by the ^{137}Cs Australian empirical model), even though this latter area is affected by pastoral activities. This may be at least partly explained by erosion in Tin Camp Creek catchment during high intensity rainstorms at the commencement of the wet season, especially if the slopes have been affected by fire during the previous dry season (Hancock et al 2008).

Gully mapping

Understanding landscape features such as gullying is an important issue in the long-term dynamics and evolution of both natural, agricultural and rehabilitated (ie post-mining) landscapes. A series of gully heads and other erosion features such as scour holes located in channels have been measured over a five year period (2002–07) in the study catchment (Fig 3). During this period the erosion features were monitored for their headward advance/retreat, enlargement or in-filling. Hillslope erosion was also monitored.

The catchment was subject to a range of rainfall regimes over the 5 years and was burnt on an almost annual basis, so that all grass cover (but not intermediate and overstory species) was removed. Box plot distibutions (Fig 4) of this monitoring show that the erosion features have changed little during this period. The gullies appear to be consistent landscape features that are in equilibrium with conditions during the monitoring period. There was little difference in erosion between years when a fire had occurred and years when there was no fire.

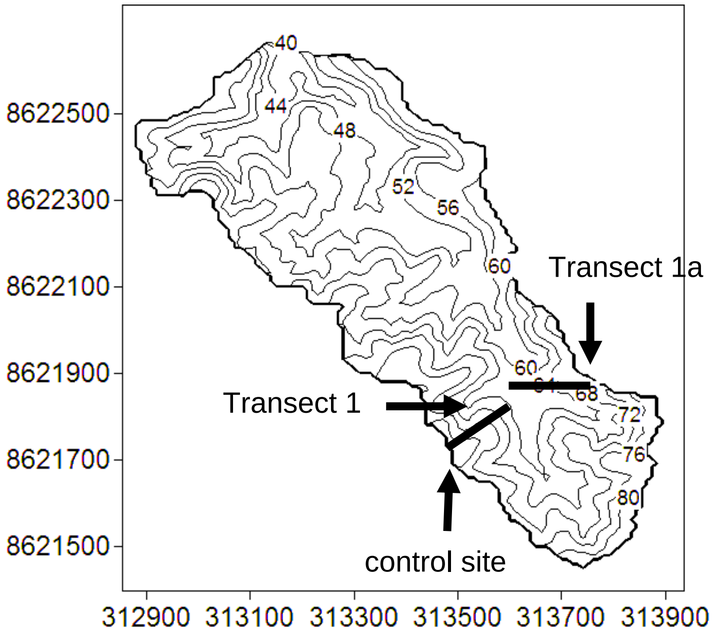

Figure 2 Contour map of the Tin Camp Creek catchment, Northern Territory, Australia**.** Coordinates are UTM WGS84, zone 53 where the x-axis is Eastings and y-axis is Northings.

Depth change of the monitored erosion features appears to be related to hillslope erosion and deposition with strong links found between hillslope and erosion feature aggradation and degradation. There also appears to be a relationship between depth and width of gullies which requires further investigation.

Soil carbon

Soil carbon plays an important role in soil water holding capacity, soil structure and over all soil health. Soil is also a significant store of terrestrial carbon. This part of the study examined soil carbon content at the hillslope and catchment scale in the study area. Results show that soil carbon concentration down hillslope transects is consistent over a number of years and that it is strongly related to hillslope position and topographic factors. These relationships warrant further investigation. An assessment of the relationship between soil carbon and soil erosion using ^{137}Cs and erosion pins suggests that sediment transport and deposition play little role in the distribution of soil carbon. Vegetative biomass appears to be the major contributor to soil carbon concentration with the occurrence of vegetative biomass being strongly controlled by topographic factors.

Steps for completion

The ^{137}Cs work is completed and published (Hancock et al 2008) and the results can be used to further assess how well erosion models simulate natural erosion rates.

Further measurements of erosion pins and gullies will be conducted over the next 1 or 2 years to assess whether there has been a change in erosion rates resulting from Cyclone Monica and/or the extraordinary wet season of 2006–07. The relationships between depth and width of the gullies will also be further investigated.

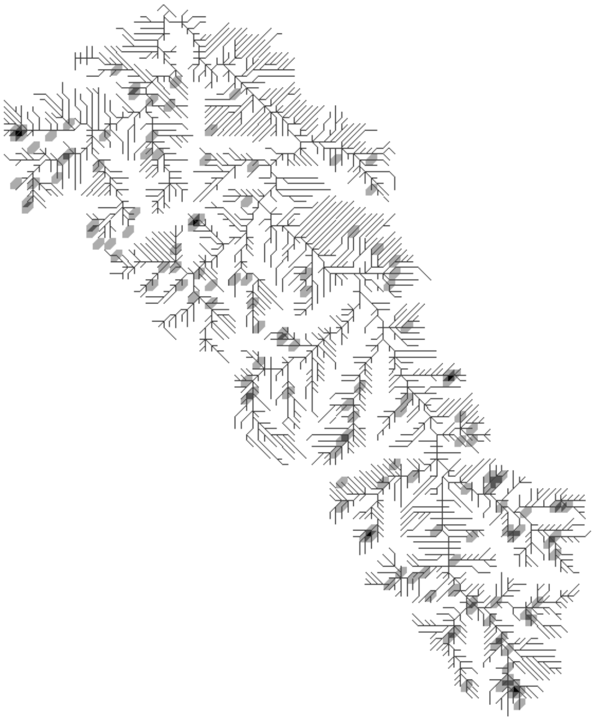

Figure 3 Position of erosion features on the drainage network (represented by grey dots) at Tin Camp Creek. The darker dots indicate two erosion features close to each other.

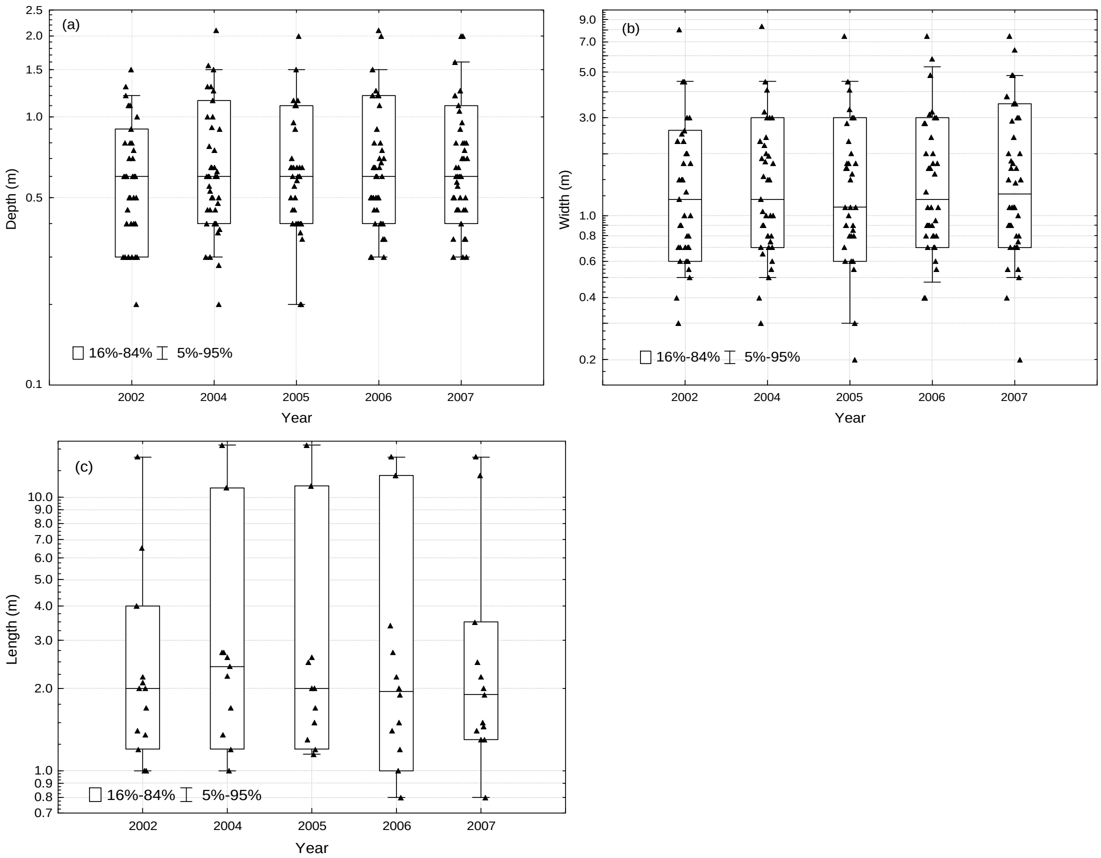

Figure 4 Measured gully features, depth (a), width (b) and length (c). The median line of each data set is shown and the black triangles are the data points.

The soil carbon work is near completion and is being written up for publication. The relationship between hillslope and soil carbon content will be further investigated as part of finalising this study.

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Pre-mining radiological conditions at Ranger mine

A Esparon, K Pfitzner, A Bollhöfer & B Ryan

Introduction

The International Commission on Radiation Protection (1991) recommends that the total annual effective radiation dose to a member of the public from practices such as uranium mining should not exceed 1 milli Sievert (mSv). This dose is on top of the natural pre-mining background dose. In a high natural background area such as the area around Ranger mine, determining an additional dose due to mining activities presents a challenge, and pre-mining conditions need to be assessed accurately so that post-mining changes in effective dose rates, especially in the event of deterioration of radiological conditions compared to the pre-mining situation can be quantified. Historical airborne gamma surveys (AGS) coupled with ground truthing surveys, have the potential to provide a powerful tool for an area wide assessment of the pre-mining conditions.

AGS coupled with ground truthing surveys have been used for area wide assessments of current radiological conditions at rehabilitated and historic mine sites (Martin et al 2006, Bollhöfer et al 2008). The aim of this project is to ground truth historical AGS data at an undisturbed radiological anomaly (an analogue of the unmined Ranger 1 and 3 orebodies) in order to extrapolate to pre-mining radiological conditions at Ranger.

An AGS of the Alligator Rivers Region flown in 1976 has been used to identify undeveloped radiologiocally anomalous areas in the vicinity of the Ranger lease as potential candidates for groundtruthing. A comparison of signal intensity with known uranium occurrences in the MODAT database suggests that Anomaly 2 to the south of the Ranger lease may be a suitable analogue site to determine Ranger pre-mining radiological conditions, as it exhibits a strong airborne gamma signal in the 1976 data. Figure 1 shows the pre-mining airborne signal in the Ranger vicinity by extent and intensity (top 70% of values found in Ranger subset) overlaid over the land surface image acquired by the IKONOS satellite in 2001. It also shows a contour map by Eupene et al (1980) which shows the total counts acquired during an airborne survey of Ranger, which is overlaid on the airborne gamma data from 1976.

Based on the assessment of the historical AGS data it was decided to obtain groundtruthed data in the greater region of Anomaly 2. An extensive fieldwork program to the south of the Ranger lease was commissioned in 2007.

Results and progress to date

The 1976 AGS data were acquired from Rio Tinto by the NT Government, and are available on the public domain (the *Alligator River Geophysical Survey*). Data were re-processed in 2000 by the Northern Territory Geological Survey and then resampled by NTGS at a pixel size of 70 m in 2003.

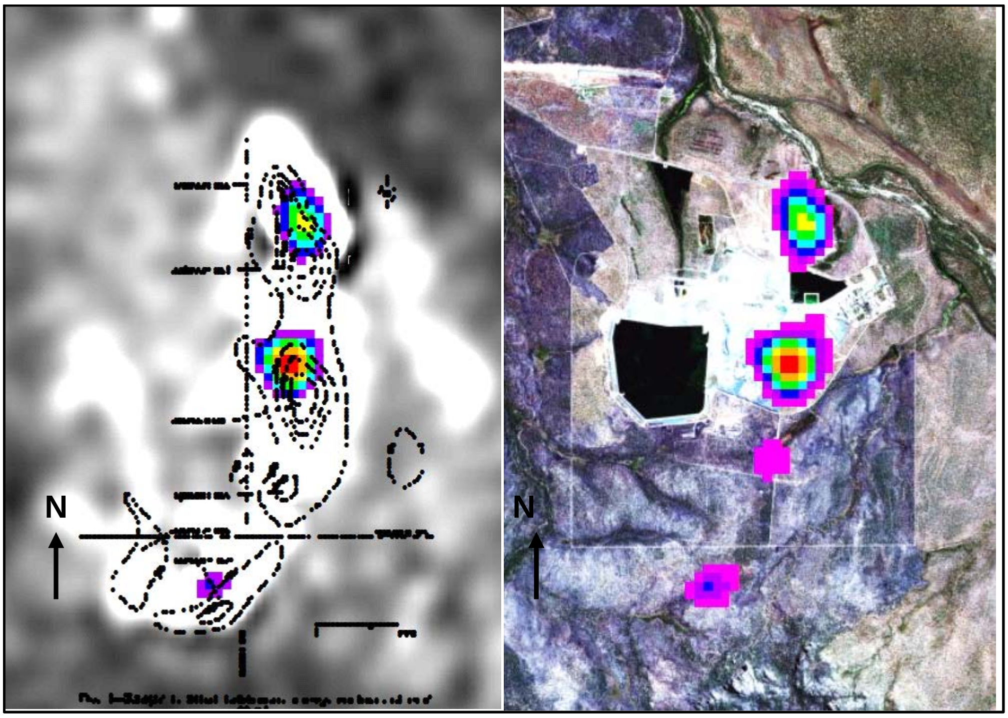

Figure 1 Eupene et al (1980) map of aerial radiometric contours of total count. Overlaid on airborne data (left) and over an IKONOS (2001) optical satellite image (right) with top 70% coloured values.

A radiation survey of the two regions surrounding Anomalies 2A and 2B was conducted in August 2007. It was confirmed that Anomaly 2 actually consists of two anomalies, Anomaly 2A (the southern Anomaly, which has a stronger radiological signal) and Anomaly 2B. Both, dose rate measurements (using environmental dose rate meters) and in situ soil activity concentration measurements (using a portable NaI(Tl) gamma detector) were conducted. The objective of the survey was to:

1 delineate the geographical location of the two Anomalies and compare their actual locations with the locations inferred from the relatively coarse 1976 AGS; and

2 groundtruth four individual 70m x 70m pixels selected from the 1976 AGS date set, spanning a range of 37 to 994 total AGS counts per pixel. Following recent recommendations made in ICRU Report 75 (2006) these measurements on the ground were performed at 40 random locations within each individual pixel.

The location and intensity of the two anomalies were delineated by the 2007 field survey. The survey data showed that there was a spatial offset of approximately 150 m between the 1976 AGS data and the true location of Anomalies 2A and 2B. This was most likely due to the coarse resolution of the 1976 AGS (300 m line spacing) and consequential numerical dispersion of the counts across pixels. Consequently, more fieldwork was required to delineate the true location, extent and intensity of the Anomalies to gain a robust data set, that will allow an extrapolation of the pre-mining AGS data from Anomaly 2 to the now mined Ranger orebodies.

ERA has made available to SSD airborne gamma survey data from an AGS that was flown in 1997. This survey was flown at a higher spatial resolution (200 m line spacing) and was used for refining of further groundtruthing fieldwork in July 2008 to precisely establish the location of Anomaly 2A (southern Anomaly).

Figure 2a shows the ERA data across Anomaly 2A (10% to 100% total count thresholds) and the four areas that were groundtruthed in August 2007. In this figure, a spatial discrepancy can be observed between the true location of the hot spot and the interpolated location from the AGS data . However, this discrepancy is relatively small (compared to that noted for the lower resolution data from 1976) and is not unusual, given the resolution of the AGS that was resampled to 25 m x 25 m.

The rectangle in Figure 2b shows the area across Anomaly 2A which was groundtruthed in July 2008. The purpose of the survey was to determine the area wide dose rate and consequently, recommendations made for the sampling of radionuclides in the environment in ICRU75 (2006) were again followed. Approximately 9.5 hectares were surveyed and measurements were taken randomly along 17 transects, using environmental dose rate meters. A total of 703 measurements were taken in the area.

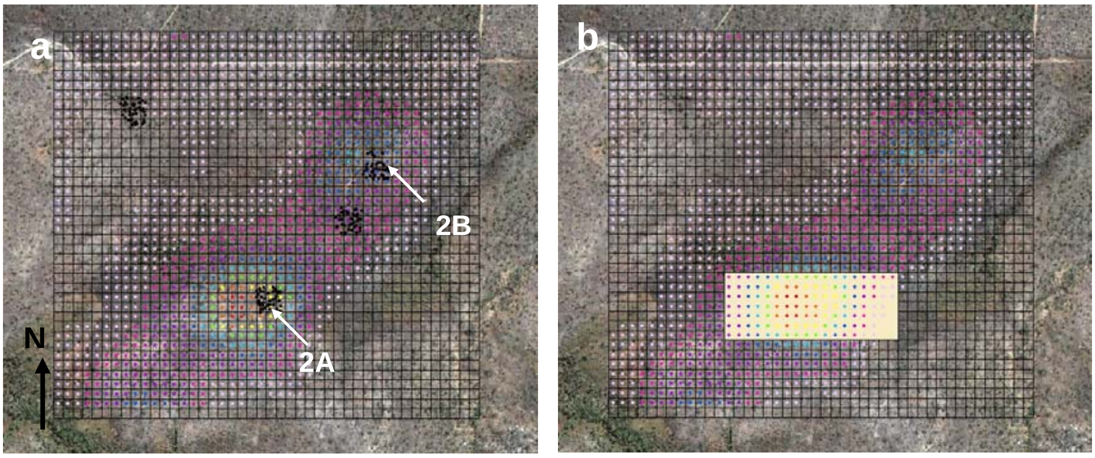

Figure 2 (a) 1997 ERA AGS data, location of the two Anomalies and location of the groundtruthing (black dots) performed in August 2007. The location of the southern hot spot (Anomaly 2A) is ~ 50 m out to the east of the location indicated by the 1997 AGS. There is a similar discrepancy for the northern hotspot (Anomaly 2B). (b) Area surveyed in July 2008. Grid cell size is 25 m x 25 m.

The maximum dose rates measured on ground in July 2008 was 16.3 $\mu Gy \cdot hr^{-1}$, and the minimum was 0.09 $\mu Gy \cdot hr^{-1}$. The area of the hot spot seems to be confined to an area much smaller than indicated by both the 1976 and 1997 AGS data. The field measurements show that the area with dose rates above 10 $\mu Gy \cdot hr^{-1}$ is smaller than ~10 m x 5 m.

Figure 3 shows the results of the on ground dose rate measurements at the area around Anomaly 2A (diamond shaped symbols, data threshold to 5 classes). It also shows the raw line data from the 1997 AGS (total count threshold to 5 classes with red = 10653-17911 and light blue = 559–1774 total counts, respectively), and the interpolated location of the hot spot (red triangles: 25 m pixels of the dose rate 90-100% threshold of the Anomaly 2A region). The five on ground dose rate classes shown are:

0–0.3 $\mu Gy \cdot hr^{-1}$	non-coloured
0.3–0.8 $\mu Gy \cdot hr^{-1}$	green
0.8–2.1 $\mu Gy \cdot hr^{-1}$	yellow
2.1–7.1 $\mu Gy \cdot hr^{-1}$	orange
7.1–16.3 $\mu Gy \cdot hr^{-1}$	red

The red line of the raw AGS data (squares) starts at ~11 000 total counts and increases to around 17 000 total counts around the interpolated 90–100% region (red triangles) of the AGS. The signal then tapers off when approaching the area of the highest readings in the field. This essentially highlights the limits of the relatively coarse line AGS data and the spatial discrepancies that arise from interpolating the raw line data.

It was also found that the dose rate signal on the ground appears to be fanning out to the south due to the movement of surface material through natural erosion processes, which is confirmed by visual inspection. Scrape samples were taken along those erosion lines for further analysis via gamma spectrometry.

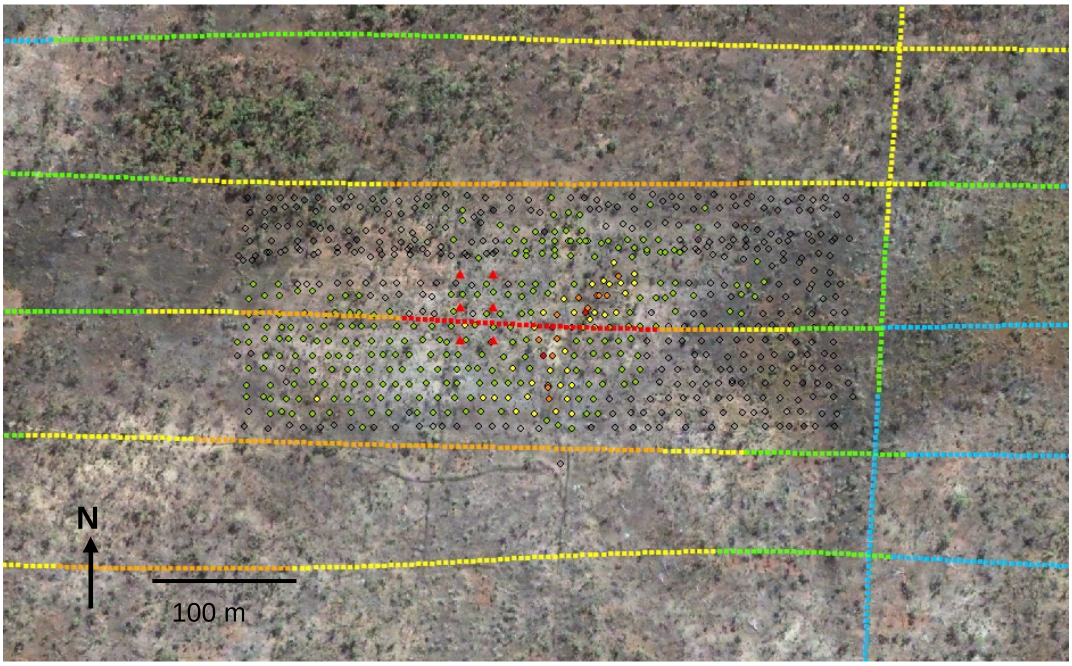

Figure 3 Raw line data for the 1997 AGS, the location of the interpolated hot spot from the AGS data (triangles) and results of the on ground γ dose rate measurements (diamond shapes) performed in July 2008

Steps for completion

Further groundtruthing around the area of Anomaly 2A and Anomaly 2B was performed in July 2008 and September 2008 respectively, and data analysis is underway. It was also realised that futher groundtruthing was required to the southwest of Anomaly 2A and groundtruthing was performed in October 2008. Once data are processed and analysed, an area wide picture of dose rates measured on ground will evolve. As the footprints of the two methods (AGS vs groundtruthing) are very different (Bollhöfer et al 2008), suitable smoothing techniques need to be explored, in order to compare on ground measurements with measurements made from a plane, with a terrain clearing of 50 m for the 1997 and an unknown height for the 1976 data. Modelling the signal measured in the plane using the flight path from the 1976 survey and the dose rates measured on ground, may provide a tool for quality control of the historic AGS data.

Radon exhalation measurements need to be performed at the Anomaly. It is anticipated that a radon exhalation study will be part of the 2009/10 work program.

Summary

Airborne gamma data revealed that Koongarra is not suited as a natural analogue for Ranger pre-mining conditions. Because groundtruthing excercises are costly and need to be carefully planned the most appropriate site needs to be indentified before fieldwork commences. Radiological anomalous areas in the vicinity of the Ranger lease, Anomalies 2A and 2B, may prove valuable analogue sites to determine Ranger pre-mining conditions. Data anlysis of the measurements performed on ground is still underway. Once data analysis is complete, the radiological conditions on ground around Anomalies 2A and 2B will be correlated to the pre-mining airborne signal in an attempt to extrapolate to the area wide radiological conditions at Ranger before mining commenced.

Acknowledgments

The NT Geological Survey, Roger Clifton and Mark Foy are thanked for discussion at an earlier stage of the project, and for data provision for the project. Jared Sellwood, Gary Fox, Robert Thorn and Alan Hughes are thanked for assistance in the field. Thanks in particular to the Mirrar people for allowing access to the sites and thanks to ERA for the provision of 1997 AGS data.

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Radio- and lead isotopes in sediments of the Alligator Rivers Region (PhD project)

A Frostick, A Bollhöfer & D Parry[1]

Introduction

This project aims at developing an innovative, sensitive and cost-effective methodology to assess and monitor impacts of past, present and future uranium mining activities in the wet-dry tropics. Funded through the ARC Linkage Projects scheme, the project is a collaboration with researchers from Charles Darwin University. The objective of the project is to characterise sources and pathways of pollutants in catchments in the Alligator Rivers Region at the decommissioned and rehabilitated Nabarlek minesite, the operating Ranger mine, and at natural analogues in order to develop a joint lead isotope/radionuclide approach for monitoring erosion from a (rehabilitated) uranium minesite to assess post-rehabilitation landform stability.

Due to the source-specific lead isotope signature and the fact that no physical or chemical fractionation of lead isotopes occurs during transport and deposition, stable lead isotopes are ideally suited as a contaminant source tracer. Lead isotopic fingerprinting relies on the fact that three of the four stable lead isotopes, lead-204, lead-206, lead-207 and lead-208, are produced by the decay of uranium-238 ($^{238}U \rightarrow {}^{206}Pb$, $t_{1/2} = 4.5 \cdot 10^9$ yrs), uranium-235 ($^{235}U \rightarrow {}^{207}Pb$, $t_{1/2} = 0.7 \cdot 10^9$ yrs) and thorium-232 ($^{232}Th \rightarrow {}^{208}Pb$, $t_{1/2} = 14 \cdot 10^9$ yrs), respectively, whereas ^{204}Pb is of primordial origin only.

In uranium and thorium rich minerals radiogenic lead is continuously produced over time. For example, monazites with high Th/U exhibit $^{208}Pb/^{207}Pb$ and $^{206}Pb/^{207}Pb$ ratios much higher than the present day average crustal (PDAC) lead (Bosch et al, 2002). On the other hand uranium ore bodies show elevated $^{206}Pb/^{207}Pb$ ratios but are low in $^{208}Pb/^{207}Pb$, as ^{208}Pb is formed by the radioactive decay of thorium. Gulson et al (1992) for example measured $^{206}Pb/^{207}Pb$ ratios in particulates from uranium tailings at Ranger as high as 9.69, whereas $^{208}Pb/^{207}Pb$ are as low as 0.0494, in agreement with results from a study of airborne dispersion of Ranger mine origin dust (Bollhöfer et al 2006).

Coupled with the measurement of radionuclide and trace metal concentrations, this technique enables identification and quantification of the deposition pathways of solids (Frostick et al 2008b, Munksgaard et al 2003). Past and present erosion, and subsequent deposition of contaminants can be identified and quantified.

Results

Nabarlek

Results of the study around the Nabarlek mine lease have been summarised and have been published in Frostick et al (2008b).

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Ranger and analogue

Sample collection in 2007/08 has concentrated on the Ranger lease area and the Magela catchment, the Nourlangie Creek billabongs, and the Ranger Anomaly 2 outside of the mineral lease area. The location of samples that were taken around the Ranger and Koongarra mineral leases are shown in Figures 1 and 2, respectively. Results and data interpretation from inductively coupled plasma mass spectrometry and gamma spectrometry analyses of these samples are due in late 2008.

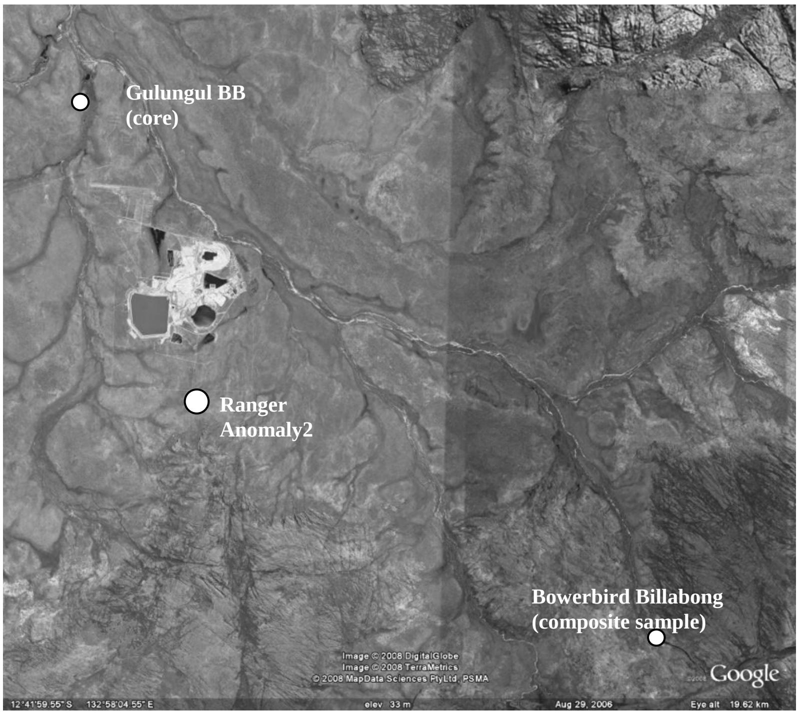

Figure 1 Location of the 2007/08 sampling sites around Ranger Mineral Lease

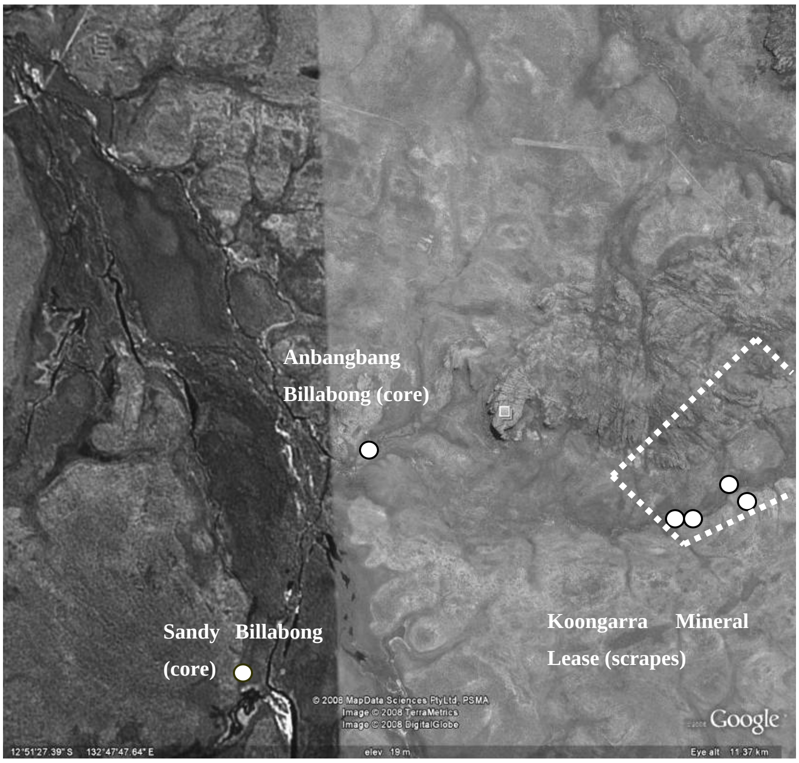

Figure 2 Location of the 2007/08 sampling sites around Koongarra Mineral Lease

In 2006–07 three sediment cores were taken in Georgetown Billabong by EWLS and were kindly supplied to ***eriss*** for analysis of radionuclides and stable lead isotopes. Results from the inlet and outlet cores are shown in Figure 3. The results from the inlet show a random pattern of lead isotope ratios measured with depth, indicative of a high degree of mixing that may have occurred during sample collection and/or through bioturbation in these sediments. However, lead isotope ratios in sediments from the outlet display a marked stratification with uraniferous sediments influencing the lead isotope ratios at less than 20 cm depth.

Highest radiogenic $^{206}Pb/^{207}Pb$ isotope ratios of ~1.90 were measured in the top 10 cm of the core. Assuming two component mixing of highly radiogenic erosion products from Ranger mine with a $^{206}Pb/^{207}Pb$ isotope ratio of 9.69, with sediments exhibiting a background ratio of ~1.51, allows to calculate a ~5 % contribution of radiogenic, mine origin lead in the top 10 cm of the outlet core.

Key trace metal and radionuclide activity concentrations in these cores will also be assessed when results of ICPMS analyses are available later in 2008. The core from the mid-section of Georgetown Billabong has also been analysed and data analysis is underway.

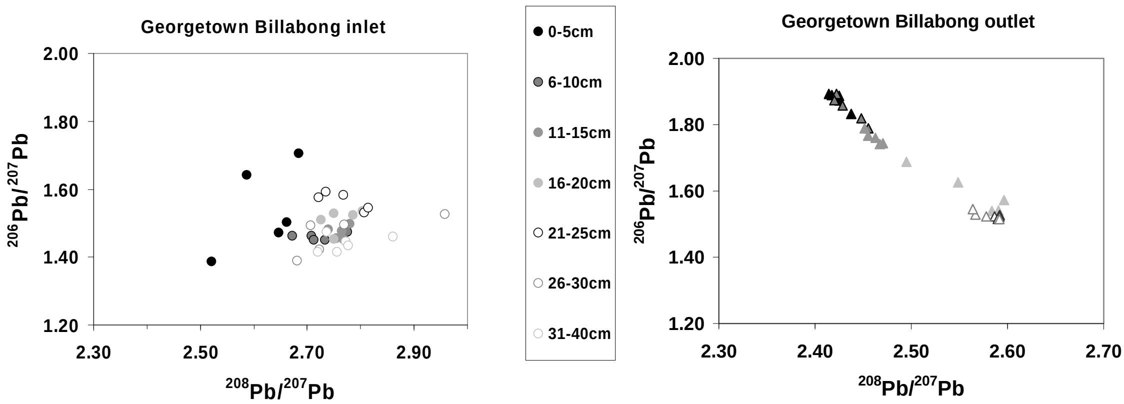

Figure 3 Lead isotope ratios measured in sediment cores from Georgetown Billabong at the inlet (left) and outlet (right)

Scrape samples were collected in 2006 on and in the vicinity of the Ranger lease. Locations are given in Frostick et al (2008a) and shown in Figure 4.

Figure 5 summarises the stable lead isotope results measured in these scrape samples and from some sediment cores from the Gulungul catchment. It can be seen that samples from the Gulungul catchment taken close to the tailings dam southern road culvert (TDSRC in Figures 4 and 5) exhibit more radiogenic lead isotope ratios indicating a contribution of uraniferous material to the total sediment composition. Lead isotope ratios are most radiogenic in the black soils downstream of the southern tailings dam wall (TDSRC Flow 1 and 2). It has previously been shown that the black soils in the Gulungul catchment attenuate high concentrations of uranium and trace metals (Mellor, 2006). In contrast, surface scrapes from Jabiluka Billabong on the Magela Creek floodplain downstream of the Ranger mine exhibit common lead isotope ratios and no siginificant contribution of radiogenic lead. More samples from the Magela catchment are currently being analysed and a full interpretation and discussion of the data is expected late in 2008.

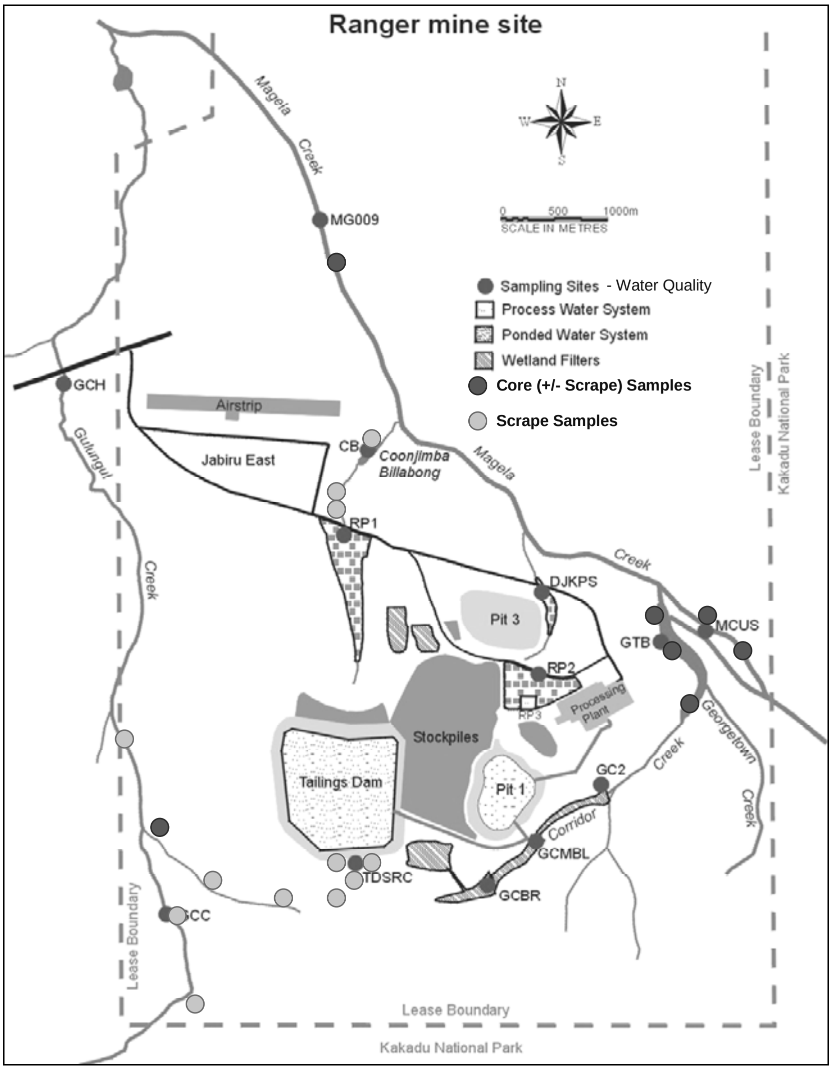

Figure 4 Location of the 2006 sampling sites around Ranger uranium mine

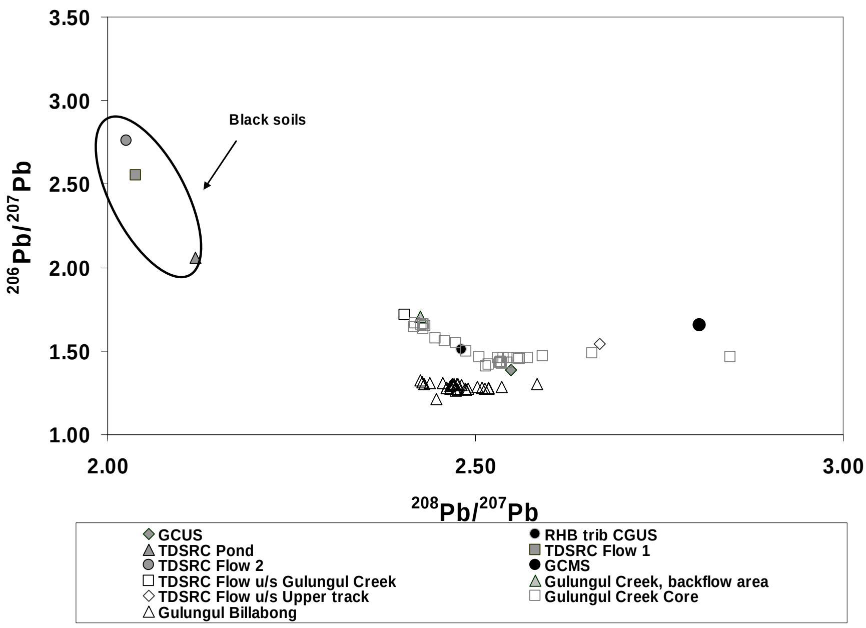

Figure 5 Stable lead isotope ratios measured in surface scrapes from the Ranger lease and sediment cores from the Gulungul catchment

Steps for completion

It was envisaged that the Koongarra Mineral Lease and the Nourlangie Creek catchment could be used as a natural undisturbed radiogenic analogue for this study. However, due to current access restrictions onto the Koongarra lease area , only 4 scrape samples were able to be collected from within the mineral lease. An additional/alternative analogue site was therefore required. Samples have been taken from the vicinity of the Ranger Anomaly 2 and are currently being analysed. Further samples from drainage channels around the Anomaly have been collected during the late dry season in 2008.

Acknowledgments

EWLS is acknowledged for providing three sediment cores from Georgetown Billabong.

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Radon exhalation from a rehabilitated landform

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

Introduction

Radon (^{222}Rn) exhalation depends on the soil (or rock) radium content, soil porosity and moisture, and vegetation cover and hence the measured radon exhalation flux density can vary greatly over short distances. For the rehabilitated landform, radon exhalation will also depend on the depth of the layer containing uranium rich material, and the characteristics of the capping material. A project conducted in 2002–05 measured radon exhalation in the vicinity of and on the operational Ranger mine site, and identified key controlling factors. Annual and diurnal variations of radon exhalation were described in this study. The outcomes have been published and factors determined, which allow prediction of radon flux densities from the soil radium content of various geomorphological units (Lawrence et al 2009). This work is important as knowledge of the source term of radon exhalation from the ground and the factors controlling its variation are needed in order to tie in results of regional radon and radon progeny measurements with predictions of radon dispersion models.

Temporal and geographical variations of radon exhalation have also been observed at the rehabilitated Nabarlek mine (Bollhöfer et al 2005). Radon flux densities reported immediately after the rehabilitation work at Nabarlek were on average 4–5 times higher (Kvasnicka 1996) than the results from the study 10 years after rehabilitation. While the use of different methodologies may have contributed to these differences in measured exhalation rates, it is possible that time-dependent landscape processes might also be responsible. In particular, it has been suggested that the porosity of the soil might have decreased over the years due to gradual weathering, infilling of voids by fines and compaction of the material, hence reducing the overall radon flux.

Specific soil closure criteria for the rehabilitation of Ranger Uranium mine need to incorporate radiological aspects to ensure that exposure of the public to radiation after rehabilitation of the mine is as low as reasonable achievable. As the inhalation of radon is likely to be a main contributor to radiological dose, radon exhalation and its temporal variability need to be estimated for the rehabilitated landform. Radon exhalation may change during the geomorphic evolution of the landform and as vegetation develops through time. The proposed trial landform will provide a unique opportunity to determine factors controlling the evolution of radon exhalation, over a period of many years. Radon exhalation for various cover thicknesses and vegetation types, taking into account weathering, erosion and compaction effects, and the effect of developing vegetation on the landform, will be determined. The project will enable ***eriss*** and ERA to predict a long-term radon exhalation flux from the rehabilitated landform and contribute to the development of radiological closure criteria and effective design of the landform.

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[2] SafeRadiation Pty Ltd, Brisbane, Qld, Australia

Methods

Conventional charcoal canisters have been used to determine the radon exhalation of the natural soil profile underlying the planned trial landform, before the landform is constructed. The charcoal cups were deployed in the 2008 dry season and exposed for three days. The charcoal canisters used were a standard brass cylindrical design with an internal diameter of 0.070 m, depth 0.058 m and wall thickness 0.004 m.

If the 'open face' of a brass charcoal canister is sealed against a surface, then all the radon emanating from the surface will diffuse into the canister and adsorb to the charcoal. With this configuration radon flux densities can be calculated assuming that the radon exhalation rate from the ground is constant over the exposure period. The existing data for natural land surfaces indicate that diurnal variations in ^{222}Rn exhalation rates in the Alligator Rivers Region are small, probably less than 20% of the mean exhalation rate (Todd et al 1998, Martin et al 2002), and consequently an assumption of a constant radon flux density is reasonable.

Progress to date

Radon (^{222}Rn) can diffuse from depths of several meters with lower layers making a decreasing contribution. The diffusion of ^{222}Rn can generally be described by Fick's law and ^{222}Rn diffusion length for dry soils has been reported to be in the 1–2 m range (van der Graaf et al 1992) and 2–5 m for sandy type materials (Holdsworth & Akber 2004). However, Lawrence (2004) determined that diffusion lengths of ^{222}Rn in stockpile structures differed substantially from normal ground and may be much larger, in the order of tens of metres. Consequently, as ^{222}Rn may reach the top surface from the base of the land form structure it was deemed necessary to conduct a survey of the substrate, before the landform is constructed. In addition, these data will contribute to the acquisition of baseline data for radon exhalation on the Ranger lease area. Figure 1 shows the location of the radon exhalation survey and the individual sampling sites.

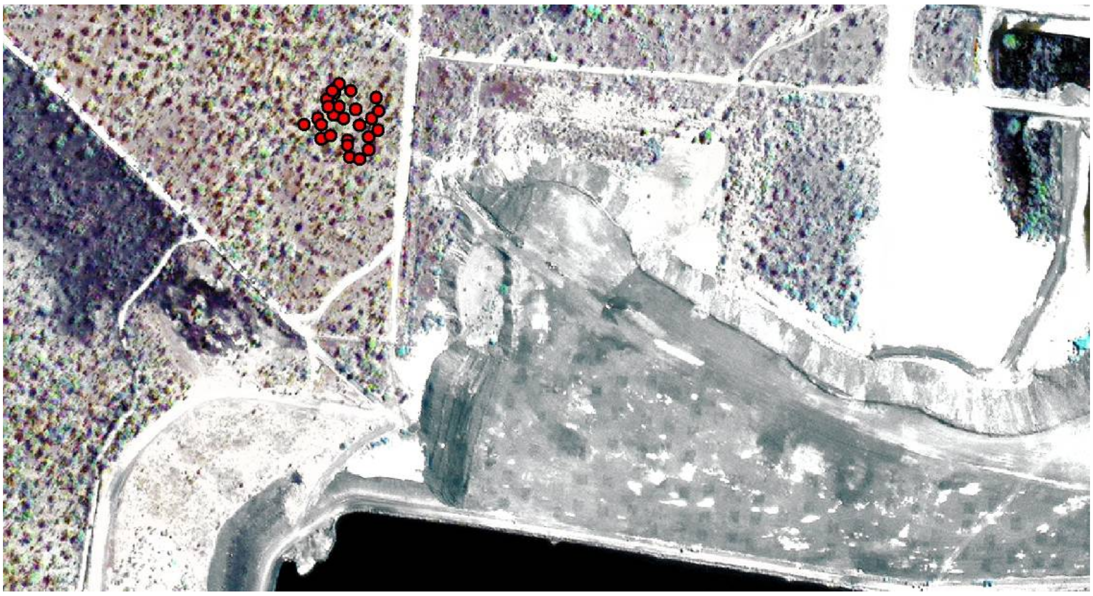

Figure 1 Location of the radon exhalation survey and individual charcoal cup locations. The north western corner of the tailings dam can be seen at the bottom of the picture.

Results of the study on radon flux densities from the substrate are shown in Figure 2. A goodness of fit test shows that the distribution is better described by a log-normal rather than a normal distribution. which is typical for radon exhalation measurements reported elsewhere (Lawrence et al 2009, Bollhöfer et al 2005). Ott (1990) states that a concentration undergoing a series of independent random dilutions in the environment tends to be log-normally distributed and this theory is especially appropriate for representing inert substances and gases released at high concentrations, such as soil radon, into carrier media, undergoing physical movement and agitation before they are measured (Ott 1995).

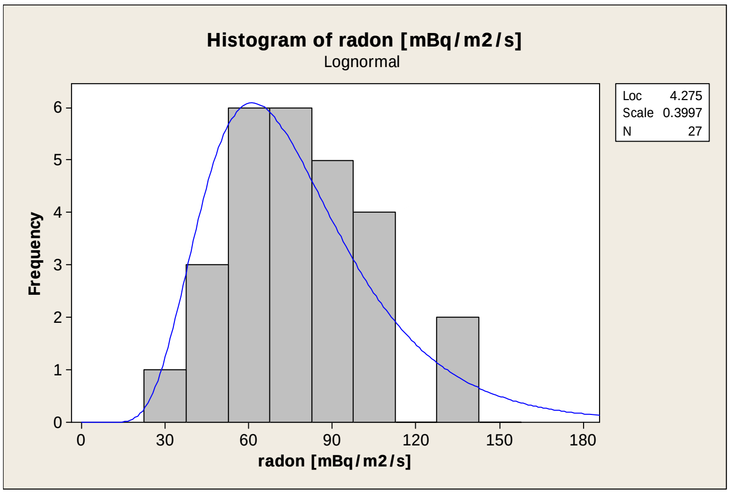

Figure 2 Histogram of radon flux densities measured on top of the substrate of the planned trial landform, dry season 2008

Radon flux densities range from 24 to 144 mBq·m^{-2}·s^{-1} and the geometric mean and median amounts to 73 mBq·m^{-2}·s^{-1}. This is similar to late dry season environmental radon flux densities previously determined at Jabiru East of 64 mBq·m^{-2}·s^{-1} (Lawrence et al 2009) and the average for the region of 64±25 mBq·m^{-2}·s^{-1} (Todd et al 1998).

Radon flux densities and soil activity concentrations will also be measured immediately after 20 cm of the top soil at the planned trial plot has been stripped, and before the landform is constructed. This will provide information on the contribution to the radon flux densities from the top 20 cm of the soil and also on radon flux densities from the subsoil profile. In addition, the collection of samples of waste rock and laterite that will be used for construction will provide a relationship between radon flux densities and soil activity concentrations for the different construction layers in the landform.

Future work

Once the landform is constructed, radon exhalation measurements will be performed every four months to determine annual variability in radon flux densities. The measurements will be conducted on the three different capping treatment areas on the trial landform, to determine whether there is a difference in radon exhalation depending on cover thickness, vegetation cover and developing vegetation on the landform. The project will continue for several years to determine the influence of weathering, erosion and compaction effects, on radon flux densities from the rehabilitated landform.

Acknowledgments

Graeme Passmore and Chloe Bradley from EWLS are greatfully acknowledged for supporting the project in the field.

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Erosion studies of the Ranger revegetation trial plot area

MJ Saynor, KG Evans & P Lu[1]

Introduction

The start of the project, scheduled for the 2007–08 work period was postponed and rescheduled to start in the 2008–09 work period.

The trial landform area is located immediately adjacent to the northwest corner of the Ranger tailings dam and will represent an extension of the topography extending out from the wall in a northwest direction (see cleared footprint in Figure 1). The constructed trial landform will cover an area of 8 hectares (400 m x 200 m).

Figure 1 Trial landform prior under construction adjacent to the northwest wall of the tailings dam (20-Oct-08)

The total area will be divided into three 2.7 ha sub-plots with different growth medium treatments; and separated by drainage lines:

1 30% lateritic material mixed with waste rock to a depth of 2 m (planted with tube stock in one half and by direct seeding in the other);

2 30% lateritic material mixed with waste rock to a depth of 5 m (planted with tube stock in one half and by direct seeding in the other); and

3 Waste rock material, divided down the middle of the plot length wise, planted one side with tube stock and by direct seeding on the other.

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Erosion plot requirements on demonstration plots at Ranger

Considering the 2.7 ha size of each treatments, it is proposed to construct four erosion plots with dimensions of 30 m x 30 m (900 m^2) on each of the treatment areas as shown in Figure 2. That is, one on each of the lateritic mix sites with different vegetation treatments and two on the wasterock material (different vegetation trials). It was decided that two plots on the laterite treatment (one on each vegetation type) would be appropriate as it is unlikely that there would be much difference between erosion on the 2 m wasterock/laterite mix and the 5 m wasterock/laterite mix. ***eriss*** has previously worked with plots of 20 m x 30 m (600 m^2) at several locations on the Ranger wasterock dump area (Evans & Riley 1993, Saynor et al 1995, Evans et al 1998, Saynor & Evans 2001).

The small sub-plots within the 2.7 ha areas are appropriate for detailed erosion studies which include topographic survey, water and suspended sediment discharge measurement and bedload measurement. Small plots have been used in previous studies in the ARR to obtain data to derive erosion model input parameter values and in numerous studies worldwide with peer acceptance. It should also be noted that a plot size of 900 m^2 is compatible with the resolution provided by landsat imagery and is similar to plot sizes used in the cyclone Monica tree fall study (Staben & Evans 2008).

Surface runoff from the plots will be measured with a flume, located at the plot outlet. The removal of the water from the plot surface down over the batters and into Retention Pond One (RP1) needs to be properly engineered/constructed so that erosion of the batter slopes is not an issue with respect to the integrity of the trial and the structure. There will also need to be some form of structure to reduce the impact of suspended sediment and perhaps bedload (sedimentation basin) into RP1 and further downstream into Magela Creek.

Erosion plot location on demonstration landform

It is proposed that the erosion plots be located along one side of the larger treatment areas near the up-slope end with the flumes off to the side and at the down-slope end of the erosion plot (Figure 2). To reduce the risk of pooling in the erosion plots and also the larger areas, the ripping *must* be level along the contour. It is likely. that water will pool in all rip lines as the surface will have a very low longitudinal slope.

Plot construction

To isolate the 30 m x 30 m erosion plots from runoff from the rest of the demonstration surface area, borders will be constructed around the plot boundary using 150 mm wide damp course and mortar. Concrete is then laid along the outer edge of the damp course covering the 40 mm long leg and the nails.

Half-section 250 mm diameter PVC stormwater pipes (Figure 3) will be placed at the down slope ends of the plots to catch runoff and channel it through rectangular broad-crested (RBC) flumes (Bos et al 1984, Clemens et al 2001) where discharge will be measured. A reservoir will be constructed at the end of the plots to trap bedload sediment in the runoff. A RBC flume with a trapezoidal broad-crested control section will be placed at the downstream end of the reservoir and equipped with a level sensor mounted in a stilling well to measure discharge (Figure 4).

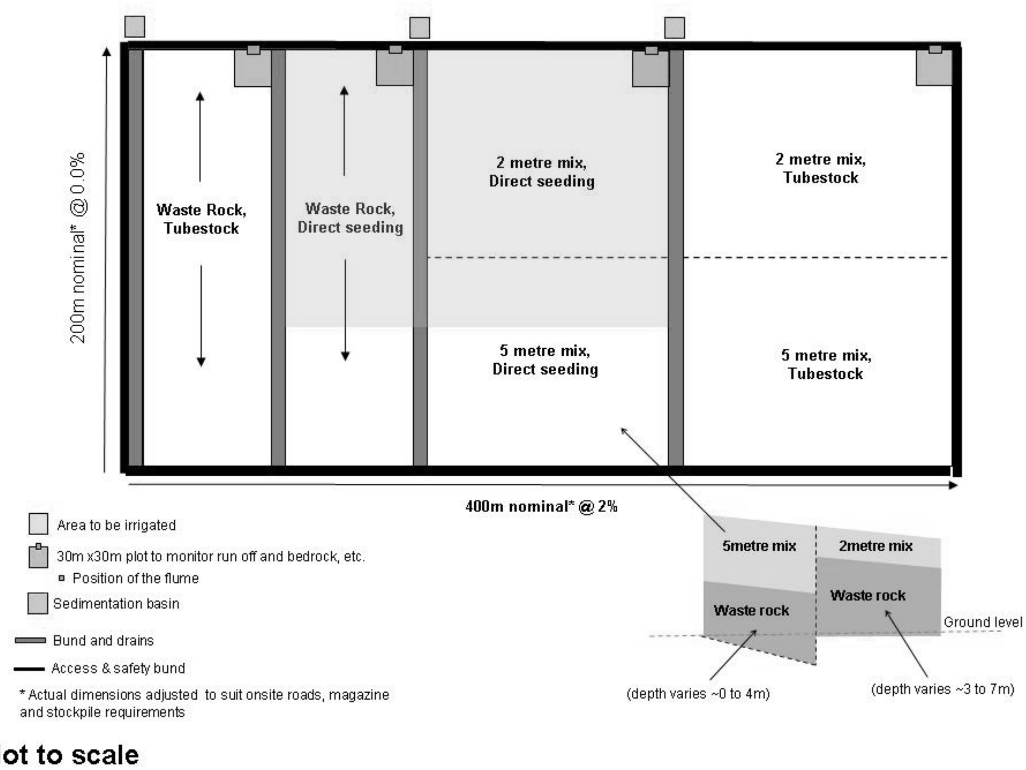

Figure 2 Layout of the plots on the demonstration landform. Note that the flumes will be at the bottom end of the plots and not in the middle as is shown (source: EWL Sciences 2008).

Figure 3 Damp course surrounding a previous erosion plot at Ranger. Note the large boulders present.

Erosion pins will be installed on a 10 m grid spacing across the entire landform surface. All pins will be measured on installation but in subsequent years only those areas with visible erosion or deposition will be measured and an assumption will be made that the surface of other areas has remained constant. The ground surface of these plots including the erosion pin locations will be surveyed immediately following installation to provide the reference land surface.

Flume size

RBC flumes with a 150 mm throat/restriction were used for earlier erosion trials on the wasterock dumps at Ranger. Overtopping did occur on one occasion and therefore it is recommended, based on peak discharge estimates, that RBC flumes with either a 200 mm or 250 mm throat/restriction are used for the trial landform.

Figure 4 RBC flume on an earlier erosion plot at Ranger. In the plots on the demonstration land form the flume will be located at the left end of the plot near the edge of the landform.

Instrumentation

The erosion plots will be constructed as soon as practicable after the demonstration trial landform has been built and prepared. The flumes will be installed at the outlet of the plots and instrumented with the following equipment and sensors: pressure transducer, shaft encoder, turbidity probe, data logger, automatic water sampler, conductivity probe. A raingauge will be installed near the flume to record the rainfall at each of the plots. The data will be accessed automatically by telemetry, downloaded once a day and then stored in the hydrological database Hystra. Decisions on how often the plots will be visited to clear bedload and collect water samples will be made after the plots are in place and there has been an opportunity to observe bedload erosion rates and discharge relative to rainfall event size.

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

C Humphrey, K Turner & D Jones

Background

The approach to deriving water quality criteria from local biological response data outlined in the Australian and New Zealand Water Quality Guidelines (ANZECC & ARMCANZ 2000) is being applied to the derivation of water quality closure criteria for waterbodies such as Georgetown Billabong, located immediately adjacent to the minesite (see maps 2 & 3). Specifically, if the post-closure condition in Georgetown Billabong is to be consistent with similar undisturbed (reference) billabong environments of Kakadu, then the range of water quality data from the billabong over time that supports such an ecological condition in Georgetown Billabong (as measured by suitable surrogate biological indicators) may be used for this purpose.

For shallow lowland billabongs such as Georgetown Billabong, distinctive wet season and dry season water quality regimes can be recognised. This is a consequence of flushing of the billabongs during the wet season, followed by contraction in surface area and substantial evaporative concentration of solutes during the six months of the subsequent dry season. If water quality closure criteria were derived from the annual-average water quality record, then the resultant values would be too conservative for the dry season and too lenient for the wet season. For this reason, two sets of water quality criteria are required – one for the wet season and one for the dry season.

Macroinvertebrates have enhanced sensitivity to water quality generally so can provide a good basis for the setting of water quality criteria for the protection of aquatic ecosystems. Hence monitoring of macroinvertebrate communities through time is being used to provide the data needed to develop closure criteria for relevant water quality indicators in the local Ranger billabongs.

Macroinvertebrates collected from aquatic plants (ie macrophytes) that are usually abundant along the waterbody edges, predominantly represent animals exposed to the water column while those collected from the benthic habitat represent those organisms exposed to sediment. It is possible that while mine-derived contaminants in the water column may be transient, reflecting short-term (wet season) changes to water quality, contaminants may accumulate in the sediment over much longer time periods, and thus present a legacy well after water quality has improved. Thus, sediment-dwelling organisms could potentially have a longer duration contact and perhaps higher effective concentration exposure to contaminants than those inhabiting the water column.

Before closure criteria for water quality indicators can be derived for Georgetown Billabong, it is important to establish whether or not the macroinvertebrate communities from each of benthic and macrophyte habitat do indeed resemble those of reference waterbodies. For example, if this were not the case for benthic macroinvertebrates and these assemblages resembled those from mine-disturbed waterbodies, it could suggest that the ecological health of this billabong was already impaired and a different approach to setting of water quality objectives would be required.

Detailed sampling for macroinvertebrates in most of the Ranger and relevant reference waterbodies was conducted previously in 1995 and 1996 and provides the starting point for time series comparison. For the 1995 and 1996 surveys (O'Connor et al 1996, 1997 respectively), the macroinvertebrate communities of Georgetown Billabong resembled those of reference waterbodies in the Alligator Rivers Region (ARR). However, for these surveys the macrophyte and benthic samples at each location were combined prior to compiling sample statistics. Thus the data arising from the composited samples represents a habitat 'averaged' condition for the macroinvertebrate communities in these billabongs.

Given the changes that have occurred on the minesite since 1996 – in particular the increased wet season loads of solutes entering Georgetown Billabong – a contemporary survey was needed to determine if the macroinvertebrate communities in the billabong were still comparable to reference waterbodies in the region. This survey was conducted in May 2006.

Previous reporting

The 2006 billabong survey results were last reported by Humphrey et al (2008) when Georgetown Billabong macroinvertebrate communities from macrophyte and benthic habitats were compared with corresponding communities collected from other ARR waterbodies, both mine-exposed and reference. For the first time, the samples from the macrophyte and benthic zones were not combined, and were processed separately prior to analysis of the data. The key findings from Humphrey et al (2008) are summarised below:

Combined habitats

- Simulating the approach adopted in previous years (1995 and 1996) where benthic and macrophyte samples were composited before sample processing, the separate datasets for the two habitats sampled in 2006 were combined for analysis. The resulting multivariate ordination showed a separation of macroinvertebrate communities from waterbodies most influenced by minesite inputs (RP1 and Coonjimba Billabong) from communities of reference waterbodies and those with minimal mine site influence (including Georgetown Billabong). This result was similar to that observed in 1995 and 1996, ie the composite habitat dataset for Georgetown Billabong indicated that the billabong had not been significantly impacted by mine inputs.
- The mean total abundance and mean taxa number for the combined habitat did not vary markedly amongst the waterbodies.

Macrophyte habitat

- When the macrophyte habitat data were analysed separately, the same ordination pattern arose as was observed for the combined habitat ordination described above (ie no evidence of mine-related effects upon Georgetown Billabong).
- The mean total abundance and mean taxa number for the macrophyte habitat also did not vary markedly amongst the waterbodies.

Benthic habitat

- When the benthic habitat data were analysed separately, the communities from Georgetown Billabong clustered among those sampled from the benthic habitat from more impacted waterbodies, most notably Coonjimba Billabong.
- The mean total abundance and mean taxa number for Georgetown benthic communities were also among the lowest recorded from the waterbodies.

In summary, the data indicated that macroinvertebrate communities from macrophyte (or combined) habitat in Georgetown Billabong are unaffected by inputs of mine-derived solutes (by nature of their similarity to those from the same habitat in reference waterbodies). However, the benthic communities from this billabong are relatively impoverished and resemble those from waterbodies receiving higher concentrations of mine water solutes.

Results to date

In August 2007, an extensive sampling program was conducted in which sediments were collected from the same waterbodies and littoral zones that macroinvertebrates were sampled from in 2006. Metals, including uranium, were analysed for in the fine (< 63 µm) sediment fraction and the results used to assess whether or not poor sediment quality was a possible cause of the lower diversity of benthic fauna in Georgetown Billabong compared to reference billabongs.

Uranium concentrations in the sediments of the waterbodies sampled in 2007 are shown in Figure 1 with a comparison, where available, to data from samples collected prior to mining in 1978 (Noller & Hart 1993).

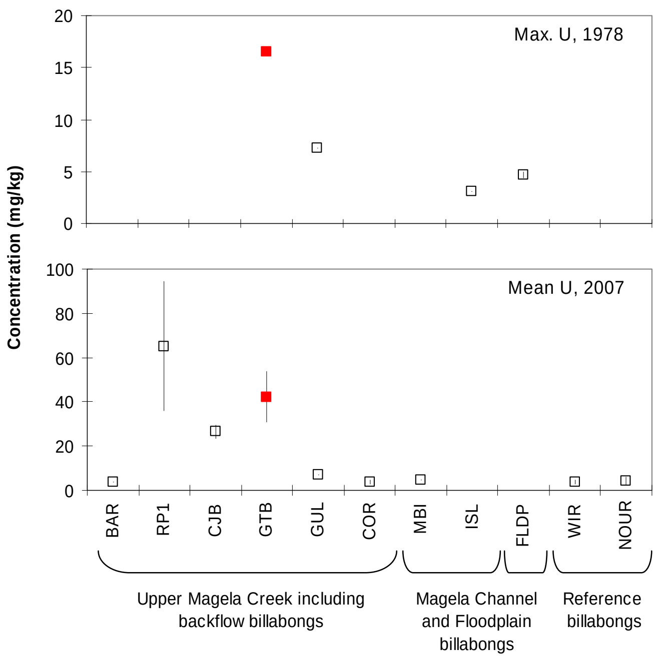

Figure 1 Maximum or mean (± SD) uranium concentration measured in sediments of a number of ARR waterbodies collected in 1978 and 2007. Uranium extracted using a nitric/perchloric digest of the fine (< 63 µm) sediment fraction. Site codes are Ranger Retention Pond 1 (RP1), Coonjimba (CJB), Georgetown (GTB), Gulungul (GUL), Baralil (BAR), Corndorl (COR), Wirnmuyurr (WIR), Mudginberri (MBI), Island (ISL), Magela floodplain (Leichhardt, Jabiluka: FLDP) and Nourlangie (Malabanjbanjdju, Anbangbang, Buba and Sandy: NOUR) Billabongs.

It is evident that U concentrations in the sediments of Georgetown Billabong have been systematically higher than those of other natural billabongs of the region since before the start of mining. This same pattern is observed for uranium concentrations found in freshwater mussels in or just downstream of Georgetown Billabong. The higher pre-mining uranium concentrations in both sediment and mussels from the billabong are attributed to natural erosional contributions from the surface expression of ore body number 1 located in the Georgetown Creek catchment (see 'A longitudinal study of radionuclide and metal uptake in mussels from Magela Creek and Mudginberri Billabong', this volume, 91–97).

The available time series of U data for Georgetown Billabong were reviewed carefully since methods of sample preparation (eg using size fractionated or total sediment) and chemical digestion (different acids and mixtures of acids) can confound the interpretation of such historical datasets. The most internally consistent set of data were selected and these have been plotted in Figure 2.

The most important observation from Figure 2 is that there appears to have been little change in sediment U concentration from before the start of mining until about 2002. Unfortunately, no directly comparable data are available between 2002 and 2007. Further and as noted above, the 2007 data were obtained from the edge of the billabong (matching sampling locations for macroinvertebrate collection) rather than from its centre. Since most of the data points in Figure 1 are for sediment from closer to the centre of the billabong, then this raises the question of whether the more organic-rich sediment from the edges contains higher U concentrations than the billabong centre. This aspect needs to be specifically addressed by obtaining contemporary samples from the centre.

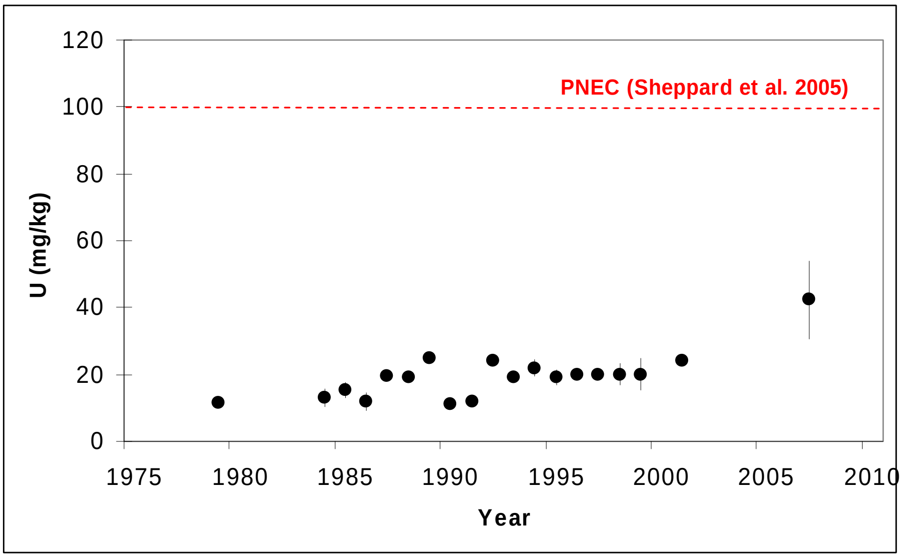

Figure 2 Mean (± SD) uranium concentration measured in sediments of Georgetown Billabong over time. Uranium extracted using a nitric/perchloric digest of the fine (< 63 µm) sediment fraction.

A similar trend for concentrations of U in sediment is evident in Coonjimba Billabong (data not shown here) with the increases in both billabongs likely due to an overall rise in U loads passing through both the Corridor Creek (Georgetown) and RP1 (Coonjimba) systems.

Despite the increase in U concentrations in Georgetown Billabong, the concentrations may not yet be sufficiently high to be toxic to benthic biota. Sheppard et al (2005), for example, derived a 'predicted no-effect concentration' (PNEC) for U in sediment based upon field-effects observed for freshwater benthic invertebrates. The PNEC (100 mg/kg, marked as a reference line in Figure 2) is well above the highest concentration observed in Georgetown sediments. This PNEC may be a reasonable ‘no effects’ threshold for ARR waterbodies given that sediments from locations within RP1 approached the PNEC (Figure 1) yet benthic communities from this waterbody were adjudged as similar-to-reference condition in macroinvertebrate diversity (Humphrey et al 2008). Nevertheless, possible confounding by effects of other metals present in the sediments from the field studies reported by Sheppard et al (2005), together with wide ranging toxicities reported in the limited number of laboratory studies completed and published to date (varying by at least three orders of magnitude, references not provided here) highlight the significant knowledge gap in sediment toxicity information for both the operational and closure phases of the Ranger mine.

Apart from lack of relevant sediment toxicity data and as previously reported (Humphrey et al 2008), a number of confounding effects diminish the ability to infer mining-related change to the benthic communities of waterbodies sampled in this study, including lack of comparable historical macroinvertebrate data for this habitat and the nature of this habitat itself. In particular, the size distribution of sediments can strongly influence macroinvertebrate communities; fine-grained, cracking-clay sediments that characterise the littoral benthos of Georgetown and Coonjimba Billabongs in particular, provide less habitat and less suitable dry season refugia for organisms compared with coarser-grained, sandy sediments such as those from Ranger RP1 and Jabiru Lake where higher abundances and taxa number were observed (Humphrey et al 2008).

At the time of sampling, field staff also made note of the particularly high amounts of leaf litter present in the Georgetown littoral substrate, arising from Melaleuca trees that closely abut the water’s edge in this billabong (ie more so than for the other waterbodies). Whether production of tannins or other compounds from recent leaf-fall had an inhibitory effect upon the benthos is an aspect that requires further study.

Future investigations will focus on:

1 Better quantifying and describing the physical nature of sediments from the various waterbodies by way of particle size distribution and possibly mineralogy (to confirm the fine-grained nature of sediments in Georgetown Billabong in particular).

2 Collecting a limited number of littoral and corresponding deeper-water sediment samples from Georgetown for chemical analysis. (The littoral samples collected in 2007 may be unrepresentative of the more central billabong samples collected by other agencies in the past.).

3 Examining the extent of metal extraction from sediments using different digest techniques on different size fractions. The results would be used assess the degree to which historical sediment quality data, often derived using different digest methods and size fractions, may be validly compared.

4 Using data from (i) and (ii), re-analyse and model environmental and biological data to better assess the degree and extent, if any, of possible mine-related change to benthic communities of Georgetown Billabong.

The outcome from this more detailed assessment will indicate if billabong closure criteria may be needed for sediments as well as water. The issue of the derivation of a site-specific sediment quality criterion for uranium at least, will be progressed with the development by ***eriss*** and collaborators of a manipulative, field toxicity experiment proposed for 2010–11. This will redress the general issue referred to above of the lack of relevant (site-specific) sediment toxicity data for the region.

Should the collective sediment quality studies proposed above show that sediment U concentrations have been increasing in Georgetown Billabong in recent years, and implicate sediment U toxicity as a factor in the low diversity of benthos, this could suggest that water quality criteria in Georgetown have already exceeded limits that protect the resident biota if it assumed that such recent sediment quality trends are a consequence of the declining overlying water quality. However, should the collective studies implicate other non-mining related factors as responsible for the observations of low benthic diversity in Georgetown, then the water quality record in the billabong associated with the periods of biological sampling (1995, 1996 and 2006) may be used to derive closure criteria. Jones et al (2008) derived interim criteria for Georgetown based on the assumption of the latter (non-mining-related) interpretation of results for benthic diversity in the billabong. Should this assumption be confirmed, then macroinvertebrate sampling may be repeated several more times between now and projected mine closure, and the guideline values adjusted, if required, to incorporate this new information. Whichever of these outcomes is confirmed, knowledge of the dynamic interaction between sediment and water quality, and interdependence of sediment quality on overlying water quality, will be required to be certain that final criteria derived for water quality in Georgetown Billabong are such as to be also sufficently protective of sediment quality.

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

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Background

Characterisation of plant communities from appropriate natural analogue sites is being used to assist in selection of species for revegetation of the Ranger mine landform following rehabilitation of the site. The characteristics of these communities will also assist in developing performance measure targets (closure criteria) against which the success of revegetation can be tracked by the post-rehabilitation programme. For the range of key vegetation community types that represent the spectrum of environments likely to be found across the rehabilitated footprint, relationships with key geomorphic features (parent material, slope, effective soil depth etc) also need to be identified. By identifying the key environmental features that are associated with particular vegetation community types, either (i) the conditions required to support these communities or, alternatively, (ii) the community types that best suit particular environmental conditions, may then be specified for the rehabilitated landform at Ranger. Additional background and rationale for this study can be found in Humphrey et al (2006, 2007).

EWLS and ***eriss*** have collaborated on this project, combining and analysing vegetation and environmental data that both groups have been collecting in the Alligator Rivers Region since the early 1990s.

All EWLS vegetation analogue sites are located on the ancient weathered Koolpinyah landscape surface, with differences in vegetation assemblages determined by topography, depth of soil profile and availability of water (Hollingsworth et al 2003). In particular, EWLS focused its vegetation surveys on the so-called ‘Georgetown analogue area’ – a relatively confined area located a short distance to the south-east of the Ranger mine. A deliberately broad range of vegetation environments was covered by the EWLS work, encompassing rocky outcrops, slopes and crests, stream alluvium and poorly-drained flats. The philosophy behind this approach was that all of these types of environments, in greater or smaller measure, would be present across the rehabilitated footprint of the area disturbed by mining.

The ***eriss*** program also included areas adjacent to the Ranger mine (on the Koolpinyah surface). However, the study sites were deliberately focused on low, broad ridge environments, perceived at the time (early 1990s) to be more similar to conditions expected to prevail across the bulk of the rehabilitated Ranger waste rock dumps (Brennan 2005). The ***eriss*** work also covered a range of hill sites elsewhere in the ARR where, again, the topography and/or substrates were considered to resemble likely final landform conditions (based on the landform design concept at that time) at Ranger (Needham et al 1973, Uren 1992). The underlying geology of these hill sites encompassed quartzite, sandstone and schist mineralogies (Brennan 2005).

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Progress to date

At ARRTC 20 (October 2007), progress was reported on (1) classification of analogue vegetation communities (trees and shrubs) to seek pattern and groupings in the plant community data, (2) collection or collation of additional plant and environmental data for analysis, and (3) data analysis to seek relationships between plant, environmental and fire history data.

From the October 2007 meeting, a key information need that ARRTC sought from future analogue work was defining the characteristics of waste rock mixed with laterite material in the context of comparison with similar data from the analogue sites. Given that some combination mix of the waste rock and laterite will constitute the substrate for both the trial and final landforms, it was important to determine whether the vegetation selected for these landforms would thrive and be self-sustaining in these conditions. Thus, the analogue work would inform species selection for the landforms. Since ARRTC 20, staff changes at EWLS and within ***eriss***, together with delays by ERA in permitting access to the Ranger site to collect waste rock/laterite substrates, have led to the following issues with the project:

- Acquiring the full dataset of environmental variables for the analogue sites. While most of the data are now available, reconstructing the EWLS environmental dataset that was used in Hollingsworth et al's (2003, 2007) modelling has been a protracted exercise.
- Soil chemistry and physical characterisation data are not available for a small number of remote hill sites surveyed by Brennan (2005). For two other hill sites surveyed (see below), the surface substrate was so rocky that soil water retention properties could not be acquired. However, these missing data are not likely to affect future modelling proposed in this study. (Soil chemistry data were not included in the modelling conducted by Hollingsworth et al (2003, 2007).)
- Physical and chemical characterisation has not yet been conducted of the components that will comprise the proposed landform capping layer. Two types of substrate need to be characterised: (i) potential substrates or substrate mixes that will be used for the new trial landform, and (ii) existing, constructed (mine-derived) substrates that have provided a medium for growth of vegetation on various trial rehabilitation sites across the Ranger mine site over the past two decades. For (i), access to suitable sampling sites for the laterite material was only gained in early September (2008), after nearly 6 months of exchanges with ERA. Arrangements for physical and chemical analysis of substrates from (i) and (ii) above are now underway.

Notwithstanding the delays that have led to the slower than originally anticipated acquisition of data described above, some initial analyses have been conducted using the (mostly) original vegetation and recently-acquired soil description and chemistry data for 30 analogue sites. These sites include:

- 18 original sites from the Georgetown analogue area surveyed by Hollingsworth et al (2003)
- 2 additional Melaleuca-dominated sites from the Georgetown analogue area surveyed for vegetation and soils more recently (June 2007), and
- 10 of the original sites surveyed by Brennan (2005), including 4 lowland Koolpinyah sites adjacent to Ranger and the Georgetown analogue area, 4 sites from the hills at Tin Camp Creek, and 2 rocky hill sites (quartz and sandstone substrates respectively).

In previous meetings, ARRTC has sought physical and chemical data for components of the mine waste rock/laterite mix so that an assessment of the suitability of mine-derived substrates for plant sustenance on rehabilitated landforms could be made. While these data are not yet available (for the reasons given above), an indication of the importance of soil properties per se in determining the local vegetation classification groups may be gained by classifying and ordinating the soil data corresponding to the natural analogue vegetation sites. If the soil characteristics produce classifications similar to the vegetation groups, this might suggest that soil factors have some influence on the vegetation patterns. If a different pattern is observed, it would suggest that other environmental variables have greater influence over the vegetation patterns, including soil variables that have not been measured.

The classification and ordination of natural soil properties provides a useful tool for assessing suitability of mine-derived substrates as a medium for plant growth. Thus, once the latter substrates are characterised, the associated soils data may be combined with those from the natural sites and re-analysed. If the mine substrate data fall outside the space of the natural soils ordination (or ‘envelope’), further investigation would be required. However, should the data fall within the ordination space of natural soils’ properties, it would suggest the medium per se is not inhibitory to plant growth.

Group average cluster analysis and multi-dimensional scaling (MDS) ordination were conducted on tree and shrub data from 28 analogue sites for which there were complete, corresponding soil quality data, using the PRIMER (v6) multivariate software package (Clarke & Gorley 2006). Of the 28 sites, 26 of these had been used in the original 38-site classification derived by Humphrey et al (2006, 2007) and used in Hollingsworth et al’s (2007) modelling. The new (re-)classification is shown in Figure 1A where site labels indicate the original vegetation classification class (C1-C5, described in Table 1). As shown in Figure 1A, all of the sites reclassify according to their original vegetation classes. Further, the two additional Melaleuca-dominated sites from the Georgetown analogue area surveyed more recently (June 2007, sites 33 and 41), classify together with the existing Melaleuca woodland sites (within classification class, C1, Table 1).

Table 1 Descriptions of the analogue communities identified in this study and, where available, the matching vegetation units according to Schodde et al (1987)

Broad vegetation community	Dominant and/or distinguishing tree or shrub species	Classification unit from this study (Fig 1A)	Vegetation units used by Schodde et al (1987)
Melaleuca woodland	*Melaleuca viridiflora*	C1	Myrtle–Pandanus savannah
Mixed Eucalypt woodland	*Eucalyptus miniata* *Eucalyptus tetrodonta* *Corymbia porrecta* *Xanthostemon paradoxus* *Acacia mimula*	C2	Open forest
Dry mixed Eucalypt woodland	*Corymbia foelscheana* *Xanthostemon paradoxus* *Erythrophleum chlorostachys* *Eucalyptus tectifica*	C3	Woodland
Low diversity schist hill	*Eucalyptus pruinosa* *Corymbia foelscheana* *Calytrix achaeta*	C4	(Not described)
Low diversity, *Corymbia foelscheana* dominated woodland	*Corymbia foelscheana* *Planchonia careya* *Syzygium suborbiculare*	C5	(Not described)

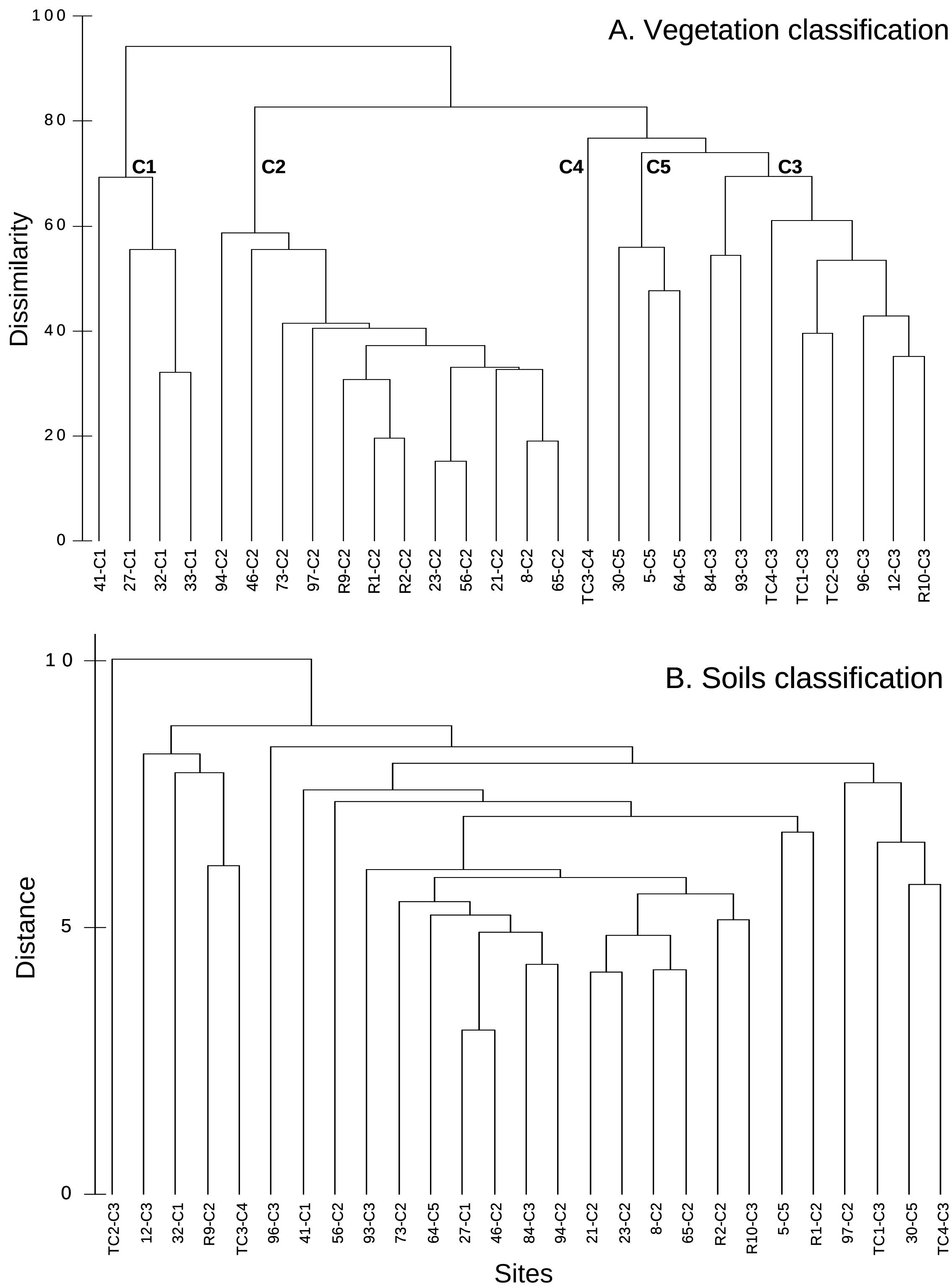

Figure 1 Cluster analysis (group average linkage) of A. vegetation (trees and shrubs) and B. corresponding soil description data for Alligator Rivers Region (ARR) vegetation analogue sites. (Vegetation data log transformed density/hectare units.) Key to site codes:
Site label suffix (C1-C5) = original classification class for the site (Humphrey et al 2006, 2007; see Table 1).
Numbered-only sites = EWLS Georgetown analogue area,
'R' sites = lowland Koolpinyah sites around Ranger,
'TC' sites = schist hill sites of Tin Camp Creek.

A total of 39 soil variables were available for the analogue sites. These variables represented soil chemistry (major ions and nutrients, 18 variables), particle size distribution (4 classes), soil water retention properties (10 variables) and soil morphology and surface drainage classes from published classifications representing horizon thickness, gravel and texture, and soil permeability (total of 7 classes). Group average cluster analysis, together with Principal Components Analysis (PCA) and Multi-dimensional scaling (MDS) ordination, were conducted on normalised soil description data using Euclidean distance (using the PRIMER software). Data were available for different soil depths and for this study, analyses were conducted for both the 0-0.05 m depth interval and for data averaged over the two depth intervals, 0-0.05 and 0.05-0.2 m. For both depth intervals (0-0.05 m and averaged), data were analysed in their entirety, or for different subsets including soil chemistry only and/or with highly correlated variables ($r > 0.80$) removed. The primary soils analyses reported below and depicted in the figures is for the entire dataset, using data for the 0-0.05 m depth interval.

The soils cluster analysis (based on Euclidean distance) is shown in Figure 1B, together with the corresponding vegetation cluster analysis (based on Bray-Curtis dissimilarity) described above (Figure 1A). Corresponding MDS ordinations for vegetation and soils data are also shown in Figure 2. Apart from some separation of the soils from *Melaleuca* woodland (C1 sites) and soils from the Tin Camp Creek schist hills, both the soils cluster analysis and corresponding ordination display generally weak groupings, unlike the well-defined major groups represented in the vegetation analysis (statistically verified using ANOSIM analysis from PRIMER). The soils PCA ordination (not shown here) showed a very similar pattern to that of the MDS ordination. Comparatively small amounts of the variability present in the original 39 soil variables were captured in the PCA, namely, 31, 16 and 9% for PC axes 1 to 3 respectively. Thus, soils from sites representing the three woodland classification groups (Table 1) are generally interspersed (Figure 2) indicating similar, average soil characteristics amongst the vegetation communities.

For cluster and ordination (MDS and PCA) analyses, the various permutations of data analysis (soil depth, data subsets, correlated variables removed) showed similar patterns to those shown for the entire dataset for the 0-0.05 m depth interval (results not shown).

The BIOENV routine in PRIMER was used to calculate the smallest subset of soil variables explaining the greatest percentage of variation in the vegetation ordination patterns. (The BIOENV procedure takes combinations of the environmental variables, k at a time, and derives the best matches of biological and environmental similarity matrices for each k, as measured by (in this case) Spearman rank correlation.) For the soils data in which highly co-correlated variables were removed, quite low Spearman rank correlations (rho) were observed amongst the best 10 solutions. Results may be summarised for the two depth intervals as follows:

- Data from the 0–0.05 m depth interval:
  - Rho values between 0.356 and 0.377. Influential variables included % sand and sulfur (all 10 solutions), potassium (8), soil permeability (7), cation exchange capacity (6), copper (3), zinc (2) and total organic carbon (1).
- Data from the 0–0.2 m depth interval:
  - Rho values between 0.337 and 0.356. Influential variables included % sand and sodium (all 10 solutions), soil permeability (9), pH (7), potassium (5), zinc and soil depth (2) and borehole infiltration, soil density and cation exchange capacity (1).

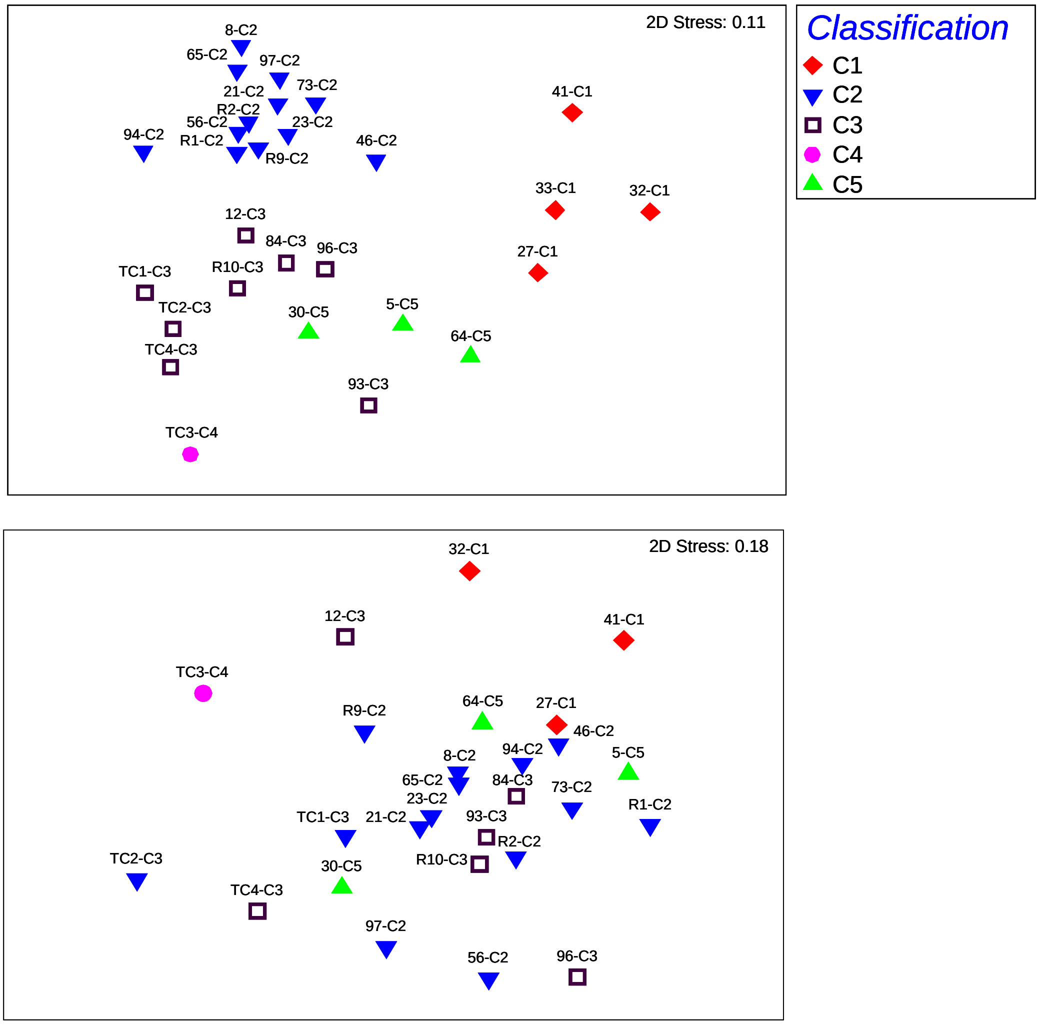

Figure 2 MDS ordinations of vegetation and corresponding soil description data for 28 ARR analogue sites. Site codes described in caption to Figure 1 while classification codes (C1–C5) are described in Table 1.

While several soil chemistry variables are correlated with the vegetation ordination space, cause and effect may be difficult to separate as similar correlations were also observed for key landscape parameters (results of the latter analysis not reported here). For example and from the BIOENV results reported above, soil pH averaged over the 0–0.05 and 0.05–0.2 m depth intervals was correlated with the vegetation ordination space, with lower pH soils associated with sites of poorer drainage and slower runoff – such as Melaleuca woodland sites (C1) sites) and low diversity, *Corymbia foelscheana* dominated woodland sites (C5), ie sites on the right hand side of Axis 1 of the vegetation ordination (Figure 2A). (Axis 1 of the vegetation ordination represents a gradient in slope from left to right, with hills sites located on the left hand side of the axis, merging to sites on flats on the right hand side.) When the numeric values of pH and runoff are superimposed on the vegetation ordination as symbols of differing size, reflecting variable magnitude ('bubble plots'), the close correspondence between the variables can be seen (Figure 3).

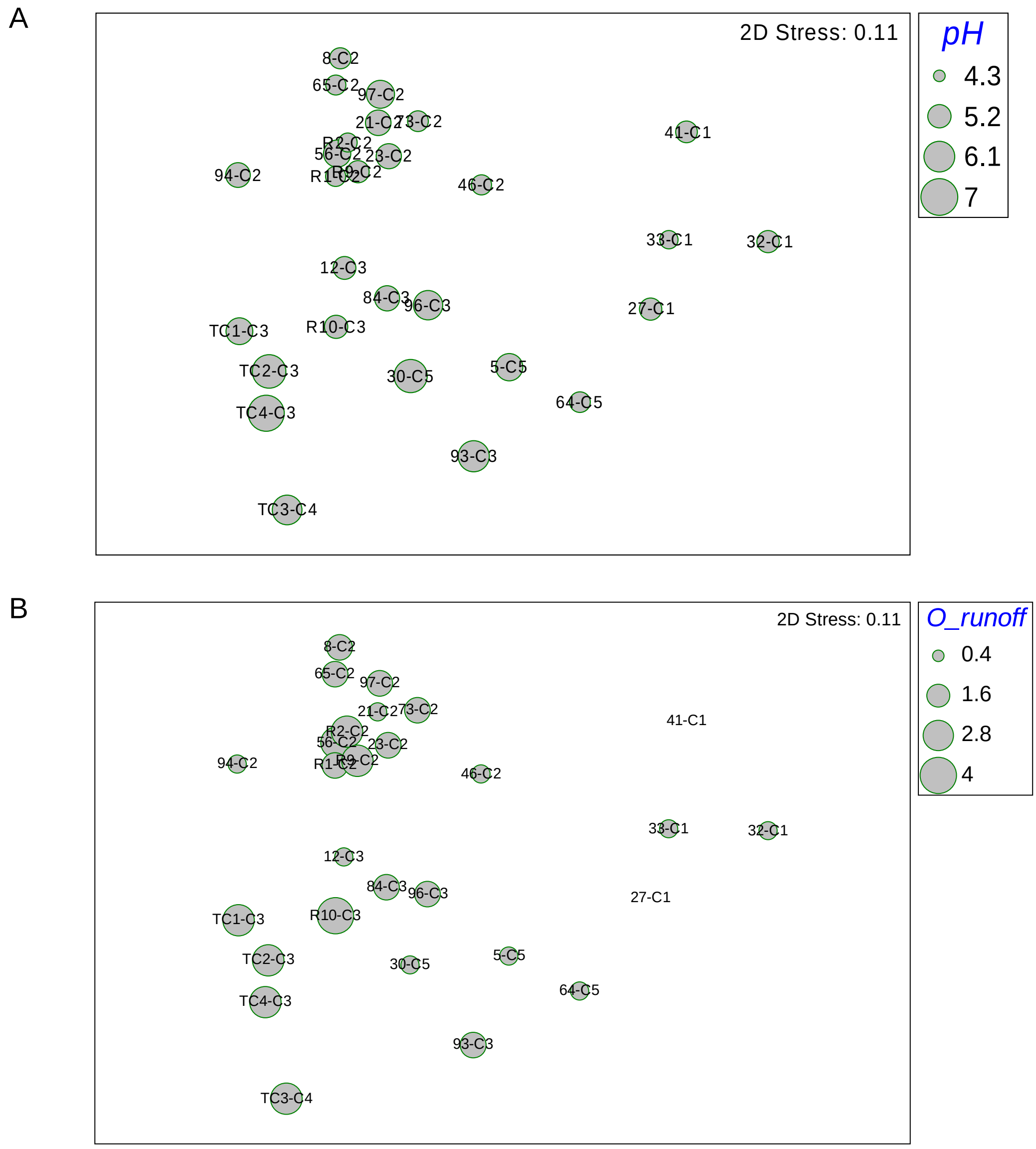

Figure 3 MDS ordination of shrub and trees data for 28 ARR analogue sites with superimposed (A) soil pH and (B) site runoff attributes overlain as bubble plots. Site codes described in caption to Figure 1 while Classification codes (C1–C5) are described in Table 1.

In general, the classification, ordination, ANOSIM and BIOENV results indicate a degree of independence of vegetation community composition and structure from the underlying soil properties that were measured and used in this analysis. These results and the demonstration by ERA and its consultants over the years of successful plant growth on harsh and stony mine-derived substrates (including neat waste rock), suggest that most of the soil descriptor variables used in the present analysis may not be fundamental to successful minesite revegetation at Ranger.

Only a limited number of soil descriptor variables (broad classes of parent material and soil morphology) were included in original plant-environment modelling conducted by Hollingsworth et al (2007), hence the need to undertake a more comprehensive assessment of the relationship between vegetation assesmblage type and soil physico-chemical parameters.

Other variables included in the analysis by Hollingsworth et al (2007) were relevant to climate and water balance, local topography, as well as fire disturbance. This work concluded that landform relief, slope and curvature were the highest correlates with important framework tree species. The results of the present analysis also strongly implicate landscape variables as being the most important determinants of vegetation communities, as drainage and runoff (correlates of topographic descriptors) were consistently correlated with the vegetation ordination space (eg Figure 3B).

The primary environmental factor to be tested for in the trial landform being established by ERA is the soil/substrate medium (three treatments – waste rock and two thicknesses of laterite/waste rock overlying waste rock). Landform features such as those identified by Hollingsworth et al (2007) (eg relief, slope, curvature) have been deliberately removed from the design in order to control just for substrate. In this conext the results presented here show that the relationship between soil characteristics and vegetation class are weak and suggest broad tolerances of plant communities to soil type. Thus, it is unlikely that potential plant-soil description modelling would greatly inform ERA of the species composition to plant out on the trial landform.

The only caveat to apply to the suggestion that soil quality may not be an important factor to revegetation at Ranger is the observations of Ashwath et al (1994) who noted elevated leachable $MgSO_4$ concentrations in some waste rock types at Ranger. Soils associated with this waste rock have the potential to become saline and therefore may be potentially toxic to some plant species (Ashwath et al 1994). Vegetation potting trials are required to elucidate these risks further.

For the trial landform, the key species representing the dominant vegetation communities that characterise the broader ARR analogue sites will be planted out and their progress monitored ('learning by doing'). In tandem with this approach, modelling of plant and environmental data using a more expanded data set than that used by Hollingsworth et al (2007) is proposed such that both approaches will inform the broader revegetation requirements for the final landform at Ranger. The expanded modelling will include the current soil description data as well as those from the newly-constructed trial landform and from existing, constructed (mine-derived) substrates that have provided a medium for successful vegetation growth on various trial rehabilitation sites on the Ranger minesite. A full suite of landscape variables will also be included.

At the time of writing this report and as reported above, the datasets for only a few of the landscape variables originally used by Hollingsworth et al (2007) had been located for re-analysis. In the documentation prepared for ARRTC 20 (October 2007), additional modelling requirements were also noted. In particular, future analyses were also needed to incorporate better (finer) resolution Landsat-derived fire history information to ensure that the long-term ecological effect of fire as a driving variable is accounted for at the required scale (see 'Undertake an ecological risk assessment of Magela floodplain to differentiate mining and non-mining impacts', this volume, 196–198).

It is worth noting, finally, that the trial landform is not likely to yield useful information on the role and importance of landform features such as relief, slope and curvature in determining vegetation outcomes (because these factors have been 'designed out' of the landform). These factors were identified from the analogue study of Hollingsworth et al (2007) as important in determining the occurrence of plant community types. Thus future analogue work will continue to be important in gaining an appreciation of how the final landform will need to be designed to facilitate establishment of the required vegetation types.

Tree root penetration study

For final landform design, the depth of soil/substrate cover that retains sufficient water-holding capacity for plant maintenance and growth, particularly during the dry season, is a key knowledge requirement. To this end, ERA/EWLS have incorporated two depth treatments (2 and 5 m) for the mixtures of laterite/waste rock overlying waste rock on the trial landform. The basis for the advice and decisions on these cover depths is from the published reports of Kelley et al (2002, 2007), as well as reports cited within these papers (not yet sighted). Tree root depth is either (i) inferred from the soil water store required to account for 100% of dry season transpiration of trees (eg Kelley et al 2002), or (ii) from cited observations of the maximum depth tree roots have been observed in the NT Top End's soils, which purportedly may reach maxima of 8 m (Kelley et al 2007). The water balance budget of Kelley et al (2002) indicated that 100% of transpiration of trees can be accounted for if rooting depth is set to 5 m.

Nevertheless, these findings appear to contradict the findings of Werner and Murphy (2001) who examined the roots of 47 trees (*Eucalyptus miniata*, *E. tetrodonta* and *E. papuana*) excavated from a site on Kapalga, on the lowlands of western Kakadu National Park. A root from only one of the 47 trees had penetrated the ferricrete layer located, on average, 0.5–0.6 m (or full range of 0.3–1.4 m) below the surface. Thus all the root mass was confined to the overlying sandy clay loam. Werner and Murphy (2001) concluded (along with other papers they cited) that trees in the NT's Top End could obtain all their water requirements during the dry season from water reserves present in the upper metre of soil.

To seek further information on this matter and thereby perhaps usefully inform decision making on this aspect with respect to minesite rehabilitation, local studies were initiated to examine root penetration of trees planted on trial rehabilitation sites on the Ranger mine site.

Some trial rehabilitation sites dating back to over a decade have been recently subsumed as a result of the recent lift of the tailings dam wall. This provided an opportunity to carefully excavate a selection of trees in the areas to be cleared. EWLS and ***eriss*** have used these occasions to opportunistically examine the depth of penetration of roots of excavated trees, growing in media that includes waste rock, waste rock and fines, or various laterite/waste rock mixes. In addition to the historic trial rehabilitation areas, the footprint area for the new trial landform has recently been cleared. This provided the opportunity to excavate trees from a woodland growing in a natural soil profile.

While the measurements made during the present investigations have not yet been analysed fully, some early findings include the observation that, while some roots can penetrate to depths of 2.1 and 2.5 m in mine-derived and natural soils, respectively, the main rootball of trees, comprising an estimated >95% of the root biomass, is invariably contained in the top 0.7 m of the soil profile (see Figure 4). What is unknown, however, is the extent to which the small percentage of roots which do penetrate to greater depths (in moist soil zones) play a critical role in meeting the tree's water requirements in the dry season. To this end, EWLS has recently completed installation of monitoring systems to measure soil moisture down to 6 m in four vegetation types in the Georgetown analogue site. Over the next two years, seasonal soil water extraction patterns, combined with whole-tree water use patterns, will be measured to assist in resolving this issue (Ping Lu, EWLS, pers obs). Such knowledge on the extent to which trees in the local region depend upon soil moisture at depth during the dry season is important. For example, if local trees, on natural or on trial rehabilitation sites are obtaining the vast majority of their dry season water requirements from the top metre or so of soil/substrate, then a 5 m thickness cap treatment for trial or final landforms would represent a substantial and costly overdesign.

Figure 4 Rootball of 7.7 m high *Eucalyptus glomericassis* excavated from a trial rehabilitation site on the eastern edge of the Ranger tailings dam (from the so-called, 'Heritage' site). While shallow lateral roots have been broken off, the main primary (tap) root is intact.

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Charles Darwin University seed biology research

S Bellairs[1]

Introduction

Charles Darwin University staff are undertaking seed biology research to optimise germination of local native species to support the rehabilitation of the Ranger mine site. The project involves collaboration between the CDU researchers and personnel from ***eriss***, ERA, Kakadu Native Plant Suppliers (KNPS), EWLS, Greening Australia and Top End Seeds.

Energy Resources of Australia are intending to establish a range of local native species on rehabilitation areas at the Ranger mine site when revegetation of the site commences. To rehabilitate these areas large numbers of plants of a range of species will be required. Therefore effective techniques will be required to germinate the seeds, whether for direct seeding or for propagation of tube stock. KNPS are producing tube stock for current rehabilitation areas using nursery facilities in Jabiru. They are collecting seeds from the local area to produce native plants that are adapted to local conditions.

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

This project is investigating seed collection, viability, germination, dormancy and storage for 50 species that occur on the Ranger mine lease that have been identified as potentially important for rehabilitation of the Ranger mine site. The project aims to develop protocols for effective seed storage and germination.

A PhD student research project supervised by Dr Sean Bellairs commenced in May 2008 and is investigating seed germination of fleshy and woody fruited species on the priority species list. The student, Mr Jagmohan Singh Sidhu, gained a CDU research scholarships and operational costs are being supported by ERA.

Approach

Seed lots are being supplied by KNPS, or from Top End Seeds and Greening Australia when KNPS is unable to provide supplies of seed. While most seed lots have been obtained from KNPS, factors such as unusual rainfall patterns and cyclones have prevented KNPS from

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supplying seeds during some seasons and alternate supplies have been obtained. It also appears that seed collection of many of the species is more difficult that originally anticipated.

Testing of seeds is being carried out under standard conditions in laboratory incubators to control light and temperature conditions. Seed responses are highly susceptible to variation in temperature and moisture conditions, thus controlled conditions are necessary to standardise the many factors impacting on seed germination. Otherwise variability can make it difficult to determine which treatments are most effective, without using large quantities of seeds.

Testing is based on the International Seed Testing Association guidelines and methodologies, with modifications to enable fewer seeds to be utilised. The methodologies used by the Australian Millennium Seed Bank Projects and other published studies are also being used as a guide so that results can be compared with those groups and so that the results obtained by MSB projects can be used as a guide when choosing treatments.

Factors being tested include seed viability following collection, the types of dormancy mechanism(s) present, effective treatments to overcome dormancy and seed longevity under various storage conditions.

The frugivory project involved feeding captive figbirds (*Sphecotheres viridis*) and pied imperial pigeons (*Ducula bicolour*) with seeds of two *Ficus* species (*F. virens* Aiton and *F. benjamina* L) to determine if seed passage through frugivore digestive tracts results in higher seed germination success than untreated (control) seeds. Seeds were also treated with acid to ascertain if artificial procedures could mimic the action of gut passage on seeds. Seed retention time was recorded to determine the likely dispersal distance of consumed seeds.

Progress to date

The seed biology project commenced on 3 July 2006. The research associate position was advertised and strong local and interstate candidates were interviewed. Ms Julie Crawford accepted the position and commenced employment on 3 July 2006. Ms Crawford obtained another position and resigned in November 2007. The position was advertised and Ms Melina McDowell commenced full time employment on 3 March 2008. Previously Ms McDowell worked part time on this and other CDU seed biology projects.

A meeting was held at Jabiru with ERA, ***eriss***, EWLS, KNPS and project staff in 2006. Fifty priority species were chosen based on their abundance in the analogue sites (data provided by EWLS) and their difficulty in propagation (information provided by Peter Christophersen). Species were chosen that tended to be perennial and were likely not to create a fire risk when established on the rehabilitation areas.

Seed lots of twenty-five of the fifty priority species have been received at CDU (Table 1). KNPS supplied five seed lots in August 2006, two in October 2006, four in January 2007 and 14 in July 2007, including some additional seed lots of previously supplied species. Thirty two seed lots have been supplied by Top End Seeds or Greening Australia NT between November 2006 and September 2008. As well as the priority species some testing has occurred for 17 other species (Table 2). Where we have not been able to source seed lots of the priority species but have been able to obtain local seeds of other species in the same genus we have obtained seed lots and tested them. In some cases other species that also occur on the Ranger mine lease have been tested for student projects or to provide a more detailed assessment of germination and dormancy trends.

An annual report on the viability, germination and dormancy present in seventeen species was provided to project sponsors August 2007 (Bellairs & Crawford 2007). The species in the report included: *Alloteropsis semialata, Aristida inaequiglumis, Brachychiton diversifolius, Brachychiton megaphyllus, Buchanania obovata, Chrysopogon fallax, Denhamia obscura, Eriachne burkittii, Eriachne obtusa, Livistona humilis, Owenia vernicosa, Persoonia falcata, Setaria apiculata, Tephrosia rosea, Terminalia carpentariae, Terminalia ferdinandiana* and *Verticordia cunninghamii*. Test procedures have been developed for these species and initial viability, germination and dormancy testing has been conducted.

In 2008 summary reports in excel have been provided to sponsors and to Kakadu Native Plant Suppliers. The research work will continue to test new species and new seed lots of existing species. Considerable literature has been obtained on the taxa (although often on southern species of the genera in the priority list).

Student Ms Bela Shah has completed carrying out experimental work for the *Ficus* frugivory project and successfully completed her Masters of Tropical Environmental Management research thesis. Removing the flesh from the seeds of *Ficus benjamina* or *F. virens* resulted in substantial germination and similar germination to that achieved by passing through the gut of either bird species. Thus acid treatment was not necessary to simulate levels of germination achieved by having the fruit eaten by the birds.

Table 1 Summary of progress investigating seed biology of the 50 priority species

Species	***# Lots received***	***Weight / Number***	***Viability***	***Imbibition***	***Germination***
Alloteropsis semialata	2	C	C,N	C,N	C,N
Aristida inaequiglumis	2	C	C,N	C,N	C,N
Brachychiton diversifolius	1	C	C	C	C
Brachychiton megaphyllus	1	C	C	C	C
Buchanania obovata	2	C	C	I,C	C
Chrysopogon fallax	2	C,P	C,N	C,N	C,N
Denhamia obscura	2	C	C,N	C	C,N
Eragrostis sp TBI	1	C	C	C	P
Eriachne burkittii	1	C	C	C	C
Eriachne glauca	1	C	C	C	P
Eriachne obtuse	2	C	C,N	C,N	I,N
Gomphrena spp TBI	2	C	C	N	P
Haemodorum coccineum	1	C	C	I	C
Livistona humilis	2	C	C	C	I
Livistona inermis	2	C	C	C	I
Owenia vernicosa	3	C	I,N,N	I,N,N	I,N,N
Persoonia falcate	2	C	C	C	C
Schizachyrium fragile	1	C	C	N	P
Setaria apiculata	1	C	C	C	C
Setaria sp TBI	1	C	C	N	P
Spermacoce sp TBI	1	C	C	N	P
Terminalia carpentariae	3	C,C,N	C,C,N	C,I,N	C,C,N
Terminalia ferdinandiana	4	C	C,N,N,N	C,N,C,N	C,P,P,N
Terminalia pterocarya	1	N	N	N	N
Verticordia cunninghamii	1	C	C	C	C

C Completed; P In progress; N Not started, I insufficient seeds.

Table 2 Investigations of species that are related to the 50 priority species or other species that occur on the Ranger area, including student projects

Species	***# Lots received***	***Weight / Number***	***Viability***	***Imbibition***	***Germination***
Chrysopogon latifolius	1	C	N	N	N
Cymbopogon bombycinus	1	C	N	N	N
Cymbopogon sp TBI	2	C	C	C	C
Dichanthium sericeum	1	C	N	N	P
Ectrosia leporine	1	N	N	N	N
Ectrosia sp TBI	1	C	C	N	P
Eriachne schultziana	1	C	C	N	P
Eriachne triseta	1	C	C	N	P
Eulalia aurea	1	N	N	N	N
Eulalia sp TBI	1	C	C	C	C
Ficus benjamina	1	I	C	C	C
Ficus virens	1	I	C	C	C
Fimbristylis sp TBI	1	C	C	N	P
Heteropogon contortus	1	N	N	N	N
Heteropogon triticeus	2	C,N	C,N	C,N	C,N
Sarga intrans	1	N	N	N	N
Sarga plumosum	1	C	N	N	N
Tephrosia rosea	1	C	C	C	P
Themeda triandra	1	N	N	N	N
Triodia bitextura	1	N	N	N	N

C Completed; P In progress; N Not started, I insufficient seeds, TBI – identification to be confirmed.

In June 2008 Casuarina Secondary College student Ms Pritika Desai carried out germination trials on fresh Gomphrena seeds under the supervision of Sean Bellairs and Melina McDowell as part of the CSIRO Student Research Scheme. The results suggested that strong physiological dormancy was present in freshly collected seed batches.

PhD student Mr Jagmohan Singh Sidhu will focus on the biology of the more difficult species with woody and fleshy fruits. Trials investigating germination of *Persoonia falcata* following various fruit treatments are currently underway.

Figure 1 Treatments applied to *Persoonia* seeds (Photo: Julie Crawford)

Acknowledgments

Ms Pritika Desai from Casuarina Secondary College established experiments investigating germination of *Gomphrena* as part of the CSIRO Student Research Scheme.

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Bioaccumulation of radionuclides in terrestrial plants on rehabilitated landforms

B Ryan, A Bollhöfer & P Medley

Introduction

The principal conduits identified for radiological exposure from current mining and milling operations at Ranger are the atmospheric and aquatic pathways. Martin et al (1998) when estimating the radiation dose received from the consumption of aquatic foodstuffs following a hypothetical release of Ranger Retention Pond 2 waters to Magela Creek, found that the mine related dose would be dominated by the intake of ^{226}Ra in freshwater mussels. This is due to the high radionuclide concentration factor for mussels and their strong representation in the local Aboriginal diet. The permanent Aboriginal settlement Mudginberri is located approximately 12 km NNW of Ranger and downstream and adjacent to Mudginberri Billabong on the Magela Creek. As of May 2008 there were approximately 30 residents of this community accommodated in permanent housing.

Customary harvesting of terrestrial bush foods may become more prevalent following the rehabilitation of the Ranger Mine with the land use expectations of local Aboriginal people changing. If this is the case the ingestion pathway has been identified as a major potential contributor to the post mining related radiological dose to humans of the area, and needs to be taken into account for the post rehabilitation dose assessment. During the development of the dose assessment models it has become apparent that these models need to be site specific and must include local dietary habits, land use and the land use expectations of the region and make use of concentration factors specific to the wet-dry tropics.

Results

Estimate of the dose to local Aboriginal people: concentration factors

Little work has been done on the uptake of radionuclides into traditional bushfoods across Australia. However, a relatively comprehensive knowledge base exists for aquatic bushfood items (Johnston 1987, Martin et al 1998) and some terrestrial foodstuffs (Martin & Ryan 2004, Ryan et al 2005) in the Alligator Rivers Region.

The aim of the current study is to bring together the radiological data collected from earlier studies which focussed more on the aquatic pathway, conducted by the Supervising Scientist Division over the last 25 years, and combine this information with the data that have been gathered more recently, particularly the terrestrial plant and animal data. This information will be used to create an up-to-date ingestion pathway model using locally derived values, replacing the IAEA default values previously used, for the group most at risk – the Aboriginal inhabitants living downstream of the Ranger mine at Mudginberri Billabong.

Dose estimate

A hypothetical diet has been developed by gleaning information from several sources which include:

- a questionnaire developed by ***eriss*** and distributed to local Aboriginal people in 2006,
- information supplied by a local supplier of meats to Aboriginal outstations and
- data gained from the ***eriss*** Kakadu bushfood project over the last 11 years.

The current status of the model diet has previously been reported (Ryan et al 2008). It must be noted that this is a work in progress and as more data become available the tables are updated with the latest results, and doses estimates are recalculated.

The model diet reported in Jones et al (2008) is somewhat different to the diet reported for Aboriginal people at Nabarlek in western Arnhem Land (see section 4). This is due mainly to the availability of shop bought food, as it is easier to access for people living in this region because of the proximity of the outstations to the mining town of Jabiru. ICRP Publication 23 states that the per capita estimate of food supplies for Reference Man from Oceania (Table 122, page 349) is 677 kg/yr. Through our research we have estimated that shop bought food makes an approximately 55% contribution to the total food intake. This would then give an annual bushfood intake of approximately 318 kg/yr. For a 10 year old child the intake is halved.

Activity concentrations for the various foodstuffs used in the dose calculations are presented in Table 1. Table 2 gives the concentration factors that have been determined for Aboriginal dietary items in northern Kakadu, and their sources are listed below.

ICRP Publication 23 gives a reference value for water intake at 1900ml per day for Reference Man (page 359). It is also stated in this publication that 'at temperatures greater than 25°C, there is a sharp rise in water intake, largely to meet demands of an increased sweat rate'. To take into account the higher temperatures an consequent water consumption in Kakadu it has been assumed that an adult drinks four litres of water (two litres for a 10 year old) water per day. This gives a total of 1400 litres for the year.

Measured radionuclide activity concentrations in food items were used for the dose assessment calculations where possible. However, to determine radionuclide activity concentrations downstream of Ranger for some food items such as fish, turtle, crocodile and freshwater shrimp, concentration factors shown in Table 2 were used in conjunction with radionuclide activity concentrations measured in Mudginberri Billabong water.

Concentration factors will also be used to help estimate the pre mining contribution to terrestrial ingestion dose as the current Ranger Anomaly 2 project progresses (see Anomaly 2 paper in section 2) and results become available. Radioactive equilibrium of all progeny in the dietary items was assumed for dose assessment purposes, unless direct measurements of progeny were available.

To assess the terrestrial pathway it was assumed that all buffalo meat consumed by the Aboriginal inhabitants of north Kakadu was supplied from the buffalo farm situated in Kakadu. Since the BTEC buffalo eradication program, Parks Australia North have kept buffalo numbers down in the north of the park and this is especially true for the Magela wetland area, making it difficult to hunt wild buffalo.

Table 1 Activity concentrations [mBq·kg^{-1}] in food stuffs used in North Kakadu dose calculations

Food item	^{226}Ra	sd	^{210}Pb	sd	^{210}Po	sd	^{238}U	sd	^{234}U	sd	^{230}Th	sd	^{228}Ra
Buffalo – flesh	18	15	16	5	230	20	8	6	13	12	2	6	-
heart/tongue	20	16	97	31	525	42	15	6	11	11	13	7	-
kidney	188	16	140	20	19000	600	5	5	11	8	7	6	-
liver	43	15	320	20	3165	320	4	8	2	17	5	5	-
Mussels	705687	-	169829	-	534751	-	2027		2027		2027		233678
Pig	29	17	21	8	4967	267	14	4	12	5	12	4	-
Magpie goose	57	25	50	8	1199	121	26	2	8	5	7	4	-
Fish group 2	216	-	43	-	1280	-	5		10	-	8	-	56
Fish group 1	3500	-	198	-	882	-	85		160	-	15	-	910
Wallaby flesh	1889	64	700	72	700	72	25		25	-	25	-	491
liver	1565	103	4300	165	4300	165	420		420	-	420	-	407
kidney	4700	201	24033	933	24033	933	281		281	-	281	-	1222
heart	635	49	981	95	981	95	29		29	-	29	-	-
Yams	6	91	9	16	0	1	3	6	0.06	.01	0.08	0.	-
Turtle flesh	160	57	98	-	1210	-	7	7	8	8	7	4	-
liver	990	1146	890	-	45000	-	95	50	130	71	45	22	-
Water lily	5090	-	4310	-	4310	-	960		1440		1440	-	-
Fruit	7.4	8.5	0.7	0.4	2.5	1.4	0.3	0.3	0.3	0.3	0.6	0.5	-
Filesnake	31	10	37	16	1177	456	21	9	21	9	53	13	-
Crocodile	120	-	34	-	1200	-	8		1		0	-	-
F/W shrimp	530	-	40	-	800	-	30		40		2	-	-
Water	3	-	6	-	3	-	1		2		1	-	-

Table 2 Concentration factors used in North Kakadu

Food item	^{226}Ra	sd	^{210}Pb	sd	^{210}Po	sd	^{238}U	sd	^{234}U	sd	^{230}Th	sd
[a]Buffalo flesh (x10^3)	0.2	-	0.2	-	2.3	-	16	-	0.20	-	0.04	-
heart/tongue (x10^3)	0.3	-	0.3	-	3.4	-	0.2	-	0.3	-	0.2	-
kidney (x10^3)	2.6	-	1.4	-	78	-	0.1	-	0.1	-	0.1	-
liver (x10^3)	0.3	-	3.1	-	8.3	-	0.1	-	0.01	-	0.1	-
[a]Pig (x10^3)	0.28	-	0.14	-	45	-	0.15	-	0.10	-	0.20	-
Magpie goose	80	-	30	-	400	-	4.0	-	8.0	-	7.0	-
Fish group 2	190	-	35	-	180	-	15	-	-	-	22	-
Fish group 1	1200	-	160	-	1400	-	250	-	-	-	40	-
[a]Wallaby flesh (x10^3)	0.08	0.03	-	-	13	0.64	0.04	0.01	-	-	-	-
liver (x10^3)	0.20	0.05	-	-	79	20	0.76	1.1	-	-	-	-
kidney (x10^3)	1.4	0.18	-	-	504	146	0.39	0.37	-	-	-	-
heart (x10^3)	0.23	0.10	-	-	20	2.0	0.04	0.01	-	-	-	-
[a]Yams (x10^3)	93	62	19	16	19	17	2.9	1.6	2.9	1.6	8.2	9.6
Turtle flesh	250	71	120	-	1000	-	25	21	18	15	32	19
liver	1200	1131	1100	-	38000	-	420	325	380	311	120	28
[a]Water lily (x10^3)	27	22	16	4.0	28	6.0	11	8.0	11	7.0	17	11
[a]Fruit (x10^3)	18	1.2	4.5	0.7	11	4.4	1.6	0.6	-	-	5.0	0.7
Filesnake	93.0	-	20.0	-	490	-	38.0	-	-	-	150.0	-
Crocodile	200	-	23	-	2400	-	40	-	3.0	-	40	-
F/W shrimp	14	-	0.5	-	20	-	0.9	-	1.1	-	0.1	-

[a] concentration factors have been multiplied by 10^3, eg the buffalo flesh concentration factor for ^{226}Ra is $0.2{\cdot}10^{-3}$

Below is a list of the dietary items and the source of the information used for the dose calculations summarised in Table 3.

Buffalo and pig: Activity concentrations for buffalo and pig are from the Magela floodplain (Martin et al 1995, 1998).

Wallaby: Activity concentrations used in the dose calculations have been determined from samples taken from Ranger Mine and Maningrida in western Arnhem Land (Ryan & Bollhöfer 2008).

Fish: Concentration factors from Martin et al (1995, 1998) were used to calculate activity concentrations for most progeny. However, measured flesh activity concentrations for ^{226}Ra in both groups of fish and ^{210}Po in Group 2 fish were used from fish caught at Mudginberri Billabong in 2002 and 2003 (Ryan et al 2005) .

Magpie go ose + waterfowl : Magpie Goose flesh activity concentrations from animals collected at Red Lily Billabong in the South Alligator district and from Maningrida in Western Arnhem Land (Ryan & Bollhöfer 2008) were used in the dose calculations.

Turtle: Concentration factors for turtle have been determined for animals collected at Bowerbird Billabong (Martin et al 1998). These have been used in conjunction with activity concentrations in water from Mudginberri Billabong to calculate doses.

File Snake: Concentration factors for file snake flesh are from Martin et al (1998) and flesh activity concentrations have been calculated using radionuclide activity concentrations in Mudginberri Billabong water .

Crocodile: Concentration factors for crocodile flesh are from Martin et al (1998) and activity concentrations have been calculated using radionuclide activity concentrations in Mudginberri Billabong water.

Shrimp: Freshwater shrimp activity concentrations were calculated from concentration factors in Martin et al (1998) and Mudginberri Billabong water activity concentrations.

Mussels: Average mussel flesh activity concentrations from mussels collected from Mudginberri Billabong from 2000–2007 were used in the dose calculations.

Yams/Fruit: Activity concentrations for specimen collected in the vicinity of Ranger, Magela Creek and from the South Alligator floodplain were used from Ryan et al (2005).

Table 3 Becquerels ingested per year from various bush foods and total annual ingestion dose in μSv for a person from North Kakadu using the model diet reported previously

Food item	^{226}Ra	^{210}Pb	^{210}Po	^{238}U	^{234}U	^{230}Th	^{228}Ra	Dose μSv/yr
Buffalo flesh	0.59	0.52	7.42	0.26	0.41	0.06	0.15	10
Buffalo heart/tongue	0.06	0.11	1.49	0.05	0.04	0.04	0.02	2
Buffalo kidney	0.42	0.31	42.6	0.01	0.03	0.02	0.11	52
Buffalo liver	0.15	1.08	10.7	0.01	0.01	0.02	0.04	14
Mussels	356	85.7	270	1.02	1.02	1.02	118	564
Pig	0.20	0.14	34.7	0.10	0.08	0.09	0.05	42
Magpie goose	0.32	0.28	6.78	0.15	0.05	0.04	0.08	9
Fish group 2	0.83	0.17	4.95	0.02	0.04	0.03	0.22	6
Fish group 1	10.5	0.60	2.65	0.26	0.48	0.04	2.74	8
Wallaby flesh	9.25	9.25	3.43	0.12	0.12	0.12	2.40	15
Wallaby liver	0.53	0.53	1.45	0.14	0.14	0.14	0.14	2
Wallaby kidney	1.35	1.35	6.89	0.08	0.08	0.08	0.35	10
Wallaby heart	0.18	0.18	0.27	0.01	0.01	0.01	0.05	1
Yams	45.6	67.1	2.78	20.2	0.46	0.58	11.9	72
Turtle flesh	0.14	0.09	1.05	0.01	0.01	0.01	0.04	1
Turtle liver	0.18	0.16	8.17	0.02	0.02	0.01	0.05	10
Water lily	1.27	1.07	1.07	0.13	0.17	0.17	0.33	3
Fruit	7.00	0.66	2.31	0.25	0.25	0.53	1.82	7
Filesnake	0.02	0.02	0.58	0.01	0.01	0.01	0.00	1
Crocodile flesh	0.40	0.06	3.02	0.03	0.00	0.03	0.10	4
F/W shrimp	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0
Water	3.88	8.32	4.10	1.96	2.48	1.78	1.36	13
Total dose [μSv/yr]								844

Using ingestion dose coefficients given in ICRP72 (1996) and the estimated annual consumption, total annual doses received via the ingestion of various bush foods for adults have been calculated and are shown in Table 3. It can be seen that in terms of terrestrial food items, buffalo, pig and yams are the biggest contributors to ingestion doses. As previously discussed an above background dose will be calculated and incorporated into the dose assessment when further data are collected and analysed for Anomaly 2.

Radium uptake in terrestrial plants

Concentration factors for passionfruit (*passiflora foetida*) have been determined relative to total soil and various soil leach fractions, respectively. These leach fractions are meant to represent a range of bioavailability from the easily available water leachable fraction (which can be mobilised by rainfall) through to the least available fraction (mainly $RaSO_4$) that can only be mobilised through the use of complexing agents .

The results of the study are shown in Table 4. A wide range of concentration factors for radium isotopes from varying environments has been reported in the literature (Simon & Ibrahim 1990, Ryan et al 2005, IAEA 1994). Fernandes et al (2006) suggest that a range of 10^{-3} to 10^{-1} should accommodate most currently known concentration factor values for soil/plant systems across different cultures, soils and natural radionuclides. This proposed range of concentration factors is similar to the range observed for radium uptake in passiflora at our study sites, relative to total soil activity concentrations of radium.

Table 4 Concentration factors for radium uptake in passionfruit (*passiflora foetida)* relative to different fractions produced by a sequential extraction process for soils from the Rockhole Residues (RR) site in the South Alligator Valley, Nabarlek and the Ranger mine

Sample ID	Total Soil	Water	$CaCl_2/MgCl_2$	HCl	EDTA	$CaCl_2/MgCl_2$ + Water
RR	0.030	213	31	0.20	0.70	27
Nabarlek	0.086	64	1.4	0.32	1.6	1.4
Magela LAA	0.018	29	0.44	0.08	0.16	0.43
Gulungul 1	0.005	8.5	0.35	0.01	0.04	0.33
Gulungul 2	0.007	11	0.32	0.007	0.03	0.31
Magela DS	0.238	4.9	2.8	1.1	1.4	1.8

The hypothesis developed from previous $^{226}Ra/^{228}Ra$ activity ratio measurements on passiflora samples and associated leaches (Medley 2007) was that the variability of concentration factors should reduce significantly when calculated relative to the water and $CaCl_2/MgCl_2$ fractions, respectively. However, a range of concentration factors still spanning almost two orders of magnitude for the water extractable fraction was observed, and the variation across all samples was similar to that when using the soil based concentration factor.

This variation is, however, biased by the concentration factor values determined for passiflora growing on an area that is contaminated by ^{226}Ra rich tailings from historic mining activities at the Rockhole Residues site in the South Alligator River Valley. In contrast, when considering natural soil profiles at Nabarlek and Ranger Uranium Mine only, concentration factors relative to the water extractable and the $CaCl_2/MgCl_2$ fractions, respectively, exhibit a much higher degree of uniformity. In particular, concentration factors relative to the water and $CaCl_2/MgCl_2$ fractions across four sites on the Ranger lease, covering 2 substantially different soil types (Chartres et al 1988), vary by a factor of only six compared with a variation spanning two orders of magnitude when using the total soil concentration (Figure 1).

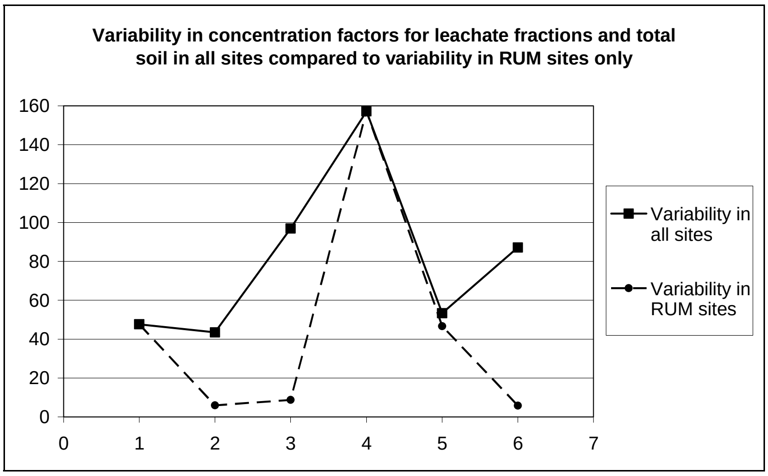

Figure 1 Variability in concentration factors (maximum:minimum) for each leach fraction at each site. Variability is shown considering all sites studied, compared to just RUM sites (from Medley 2007).

Future work

Final data analysis, in particular analysis of ^{226}Ra/^{228}Ra activity ratios in all samples and leachate fractions is underway to determine the fraction most suitable to use to (a) determine concentration factors for uptake of radium and (b) predict radionuclide activity concentrations in plants at the rehabilitated Ranger mine.

Acknowledgments

We would like to acknowledge Anthony Sullivan and Sally Atkins from SSD, the Mudginberri residents of Kakadu, and David Lindner for providing advice and for their help with sample collection .

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Development of a spectral library for minesite rehabilitation assessment

K Pfitzner, A Bollhöfer & A Esparon

Introduction

The aim of this project is to develop a spectral library of land cover components measured in situ in order to make recommendations for appropriate acquisition of remotely sensed data for land cover condition assessment and monitoring of the mine environment and surrounding country. The hypothesis is that with a well designed approach to collecting field spectral measurements and metadata, extraneous factors can be accounted for, accurate processing of spectra can be performed and the first database of Top End spectra relevant to the mine environment can be developed.

To populate the spectral library, reflectance characteristics of weed and native ground covers have been sampled fortnightly from permanent plots at Crocodylus Park, CSIRO and Berrimah Farm near Darwin. Mineral assemblages and soils that represent both mining surfaces and the 'reference' surrounding country will be measured, along with spectra representing the operating mining environment, such as stockpile material.

A field spectrometer that measures reflectance continuously across 350–2500 nm at full-width-half-maximum (FWHM) resolution of 3 nm for the region 350–1000 nm and 10 nm for the region 1000–2500 nm is used to collect the spectra. The spectral data are supported by metadata describing the viewing and illumination geometries, environmental conditions and state of the target measured. SSD has developed standards for collecting field reflectance spectra and these have been reported previously. The Spectral Database will be used to reference, categorise and manage the spectral data and metadata, with the aim to query and analyse suitable spectra only, that are not influenced by extraneous factors that may influence data quality.

Results – vegetation sampling

As part of the spectral database development, dense and homogenous plots of vegetation, particularly those priority species of weeds and native ground covers that are of concern to the revegetation success at minesites, have been established and measured seasonally between 2006 and 2007. Selected examples of these vegetation plots are illustrated in Figure 1.

In 2006 and 2007, around 260 and 200 different sets of spectral measurements of native and weedy vegetation covers were made, respectively. The Research Scientist of this project was on maternity leave March–October 2007, but the collection of spectra and metadata continued with the Remote Sensing Technician. The status of the vegetation plots for the spectral library project was reported in an internal report (IR546), and the collection of spectra ceased in early 2008. The plots at Berrimah Farm and CSIRO were dismantled. Plots remain at Crocodylus Park, but these are not being maintained for homogeneity or density of coverage by the target species.

Digitaria milanjiana (Jarra Grass) _2008_01-23

Brachiaria humidicola (Tully Grass)_2008_01-23

Digitaria eriantha (Pangola Grass)_2008_01-23

Digitaria swynnertonii (Arnhem Grass)_2008_01-23

Stylosanthes humilis _2008_03_19

*Sorghum stipodeum*_2008_03_19

Hyptis suaveolens_2008_03_04

Melinis repens (red natal grass)_2008_03_19

Aeschynomene americana_2008_03_04

Pennisetum pedicellatum_2008_03_04

Figure 1 Examples of vegetation plots used to record the spectral reflectance of selected species over time

To maximise the usefulness of the spectra measured, and account for any extraneous variation in spectral measurement, a system was developed to organise and retrieve spectral and metadata records in SSD's Spectral Database. Metadata entries include SSD's standardised site and target descriptions, environmental and illumination conditions, measurement information and photographic records. SSD required a system to account for and link the spectral and metadata standards and hence, the database structure has been custom designed to maximise cross referencing between spectra, photos and metadata. An SQL server is used as a data warehouse to store all information.

The spectra and photos are stored as binary files within the database. The metadata table contains information about the conditions at the time spectra and photos were taken. Metadata include a unique code (site and date), date of spectral measurement, atmospheric conditions (smoke, haze, temperature, humidity, air pressure, wind direction, wind speed and description), cloud level and cover, probe height (from ground), plant height (from ground level) and ground description (by cover and phenology). Searches can be performed on the fields and individual records displayed. Figure 2 illustrates an example spectrum metadata page with associated photographs. Figure 3 displays the spectrum list for the same metadata and photo page illustrated in Figure 2.

The structure allows the user to easily query information. Selected spectra can be viewed and overlaid to give a visual comparison. Figure 3 provides an example of a solar irradiance, target and white reference spectra. Each spectrum has certain characteristics that can be used to categorise them through an iterative process. As part of the quality control procedure all spectra are processed through an algorithm that categorises the spectrum into a specified group depending on the defined boundary conditions. Spectra that are not found to fit into a known category are marked as undefined ('not defined') by the 'Classify' algorithm. These spectra require further examination and indicate problematic conditions.

Highlighting a detector array issue or atmospheric influence in a white reference spectrum is crucial for data analysis and these anomalies would be very difficult to detect visually with the volume of data that are being stored, processed and analysed. Accurate metadata is required during the data analysis stage to ensure that environmental conditions (such as solar azimuth) are not influencing the spectral response, particularly for time series of spectral measurements. Photographic records help to interpret and determine the quality of time series data by supporting quantitative and qualitative measurements of the hemispheric component.

Future work

Once all 'suspect' spectra have been filtered out, analysis can commence on the high integrity data. There are a number of spectral analysis management systems available online, including SAMS (Rueda & Wrona 2003), SPECCHIO (Hüni 2007, Hüni & Kneubühler 2007), SPECtrum Processing Routines (SPECPR) (Clark 1993, Kokaly 2005) and SpectraProc (Hüni & Tuohy 2006). SSD also has expertise in computing language and interactive environments for algorithm development, data visualisation, data analysis, and numeric computation.

There are a number of toolboxes available including project specific signal processing techniques. These may be tested in parallel to benchmark the performance of any custom methods produced by this project. The data will be analysed for both between and within species similarity and dissimilarity. Feasibility studies will be performed to determine the spectral separability of ground covers for a variety of remotely sensed platforms and recommendations made on the most appropriate datasets required for minesite rehabilitation assessment.

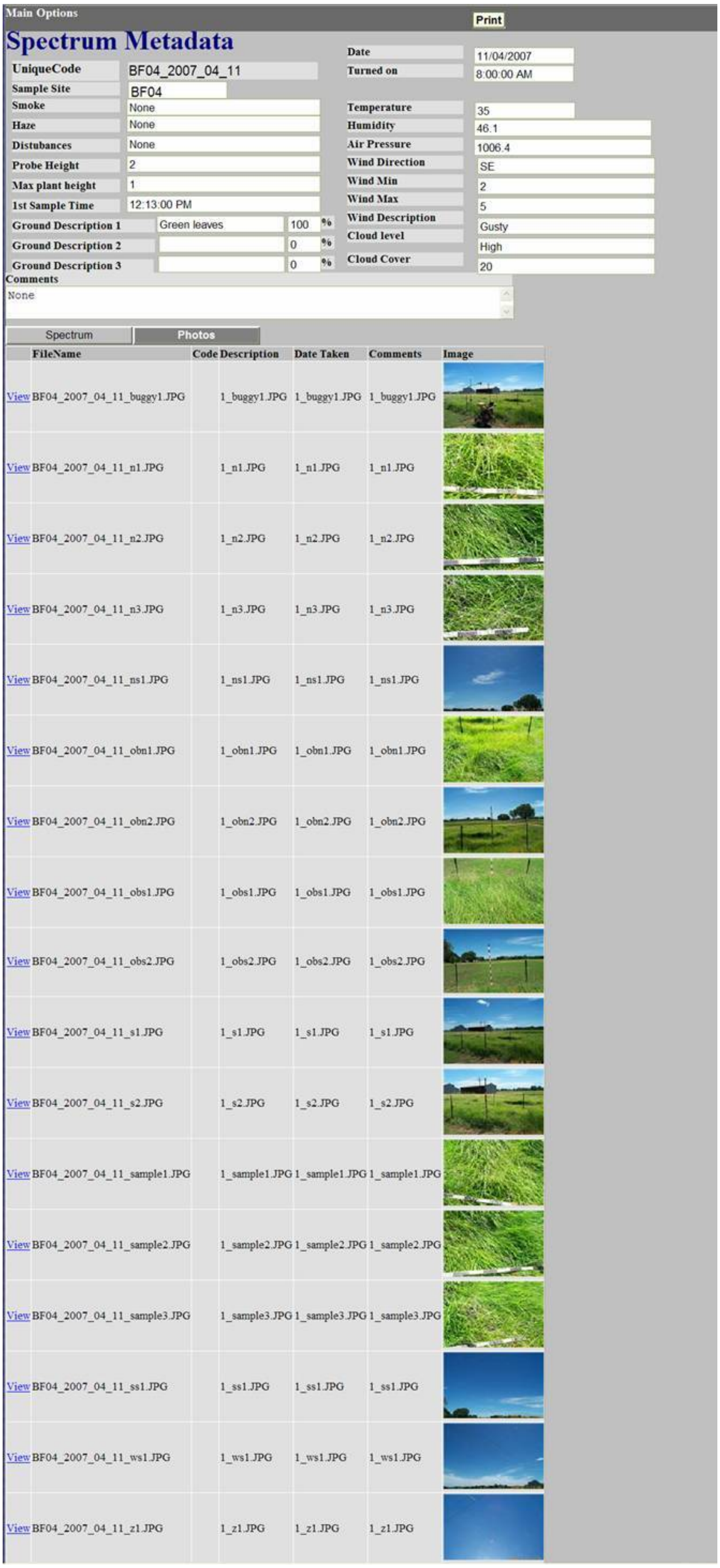

Figure 2 An example metadata page with associated photographs

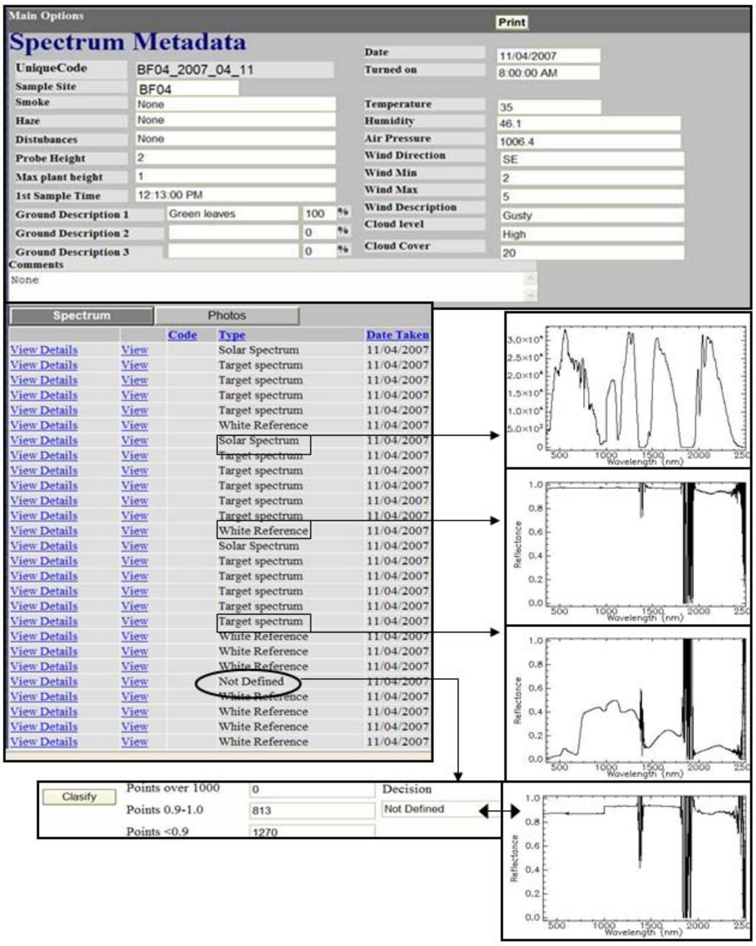

Figure 3 The spectral data associated with Figure 2

Alongside the vegetation spectral analysis will be the extension of the database to incorporate field measurements of mineral and mineral assemblages at minesites. In addition, soil samples taken around the Ranger lease will be prepared and measured in the laboratory, and the field and laboratory measurements of standard panels will be included in the database.

Acknowledgments

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Development of catchment geomorphic characteristics of Gulungul Creek – monitoring results

DR Moliere, MJ Saynor & KG Evans

Background

The aim of this project is three-fold: (1) to develop reliable impact assessment methods for quantifying the mud loads transported during rainfall/runoff events; (2) to characterise the channel stability of the creek; and (3) to characterise the bedload movement upstream and downstream of Ranger.

In regards to the first aim, event mud load data collected since 2003 from upstream and downstream (GCUS and GCDS) of the Ranger operations footprint along Gulungul Creek have been used to quantify the magnitude of mud loads and assess whether the source is natural or potentially mining related. The locations of the monitoring stations are shown in Figure 1. Trigger levels (which can be used for future impact assessment) for event mud loads have been derived for current pre-rehabilitation conditions using two complementary methods (Moliere & Evans 2008) – BACIP analysis and a relationship between mud load and discharge characteristics.

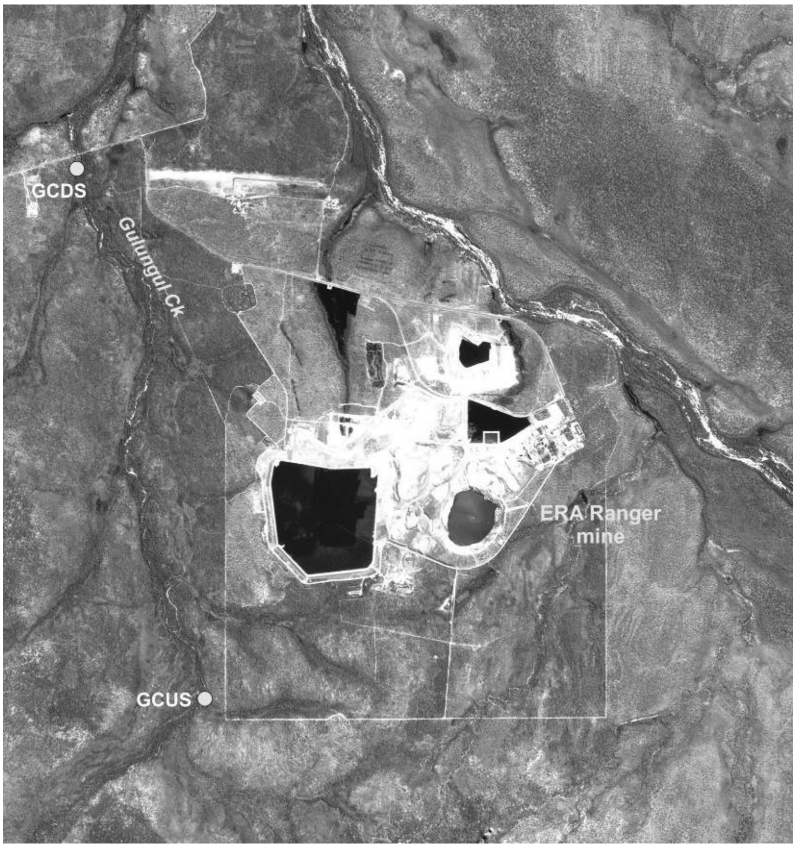

Figure 1 Location of the monitoring stations along Gulungul Creek

Bed material sediment sampling was initiated during the 2007–2008 wet season using a pressure difference Helley-Smith bed load sampler to determine baseline sediment loads for the derivation of a complete sediment budget. The sediment budget will be used to (1) provide data inputs into and validation of the predictions of, landform evolution models, and (2) baseline for monitoring the performance of the rehabilitated landform.

Cross sections, scour and fill and bed particle size are measured at numerous locations along the main channel on an annual basis to determine the stability of the stream channel along Gulungul Creek. These data will be used to develop knowledge of the geomorphic behaviour of the creek under current conditions (which can be used for future impact assessment). At this time the collected bedload samples have yet to be processed and analysed.

Results

Impact assessment methods for quantifying the mud loads (Aim 1)

Event mud load data collected since 2003 at GCUS and GCDS have been used to establish preliminary trigger values for both the event-based BACIP analysis and the relationship between mud load and discharge approaches to analysing event loads. Details of the methods used for deriving trigger values for both assessment techniques are described in detail in the companion paper 'Turbidity and Suspended Sediment Management Guidelines and Trigger Values for Magela Creek' under this KKN and in Moliere and Evans (2008).

Using the BACIP approach, events that lie above the 95^{th} percentile ('action' trigger) are considered to have an elevated mud load measured downstream relative to the load upstream. Using the regression model approach, a potentially impacted event is identified if the mud load measured downstream is significantly elevated compared to the corresponding event discharge characteristics (ie lies above the + 2 SD line) for that location. An impact is confirmed if the corresponding event mud load measured upstream of Ranger is not significantly elevated compared to the flow discharge characteristics (ie lies within + 2 SD of the fitted relationship) for that location.

Events where mud loads measured downstream have exceeded the 'action' trigger levels associated with both assessment techniques are highlighted by specific date labels in Figures 2 and 3.

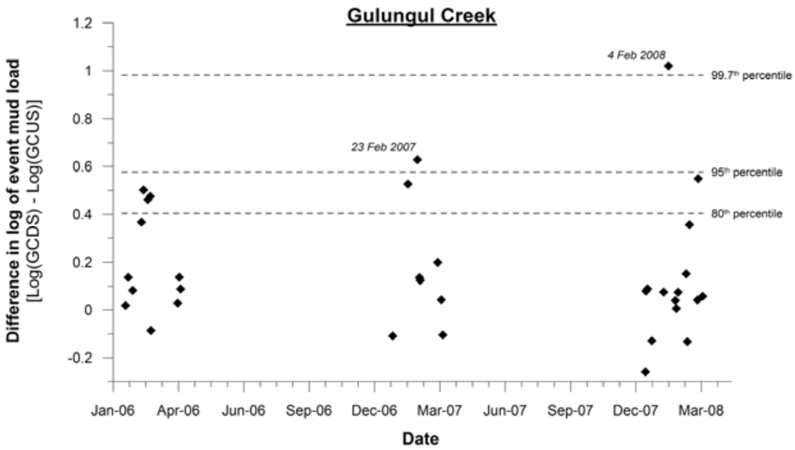

Figure 2 Temporal variation of the difference in the logarithms of the discrete event mud loads (indicated as ♦) measured in Gulungul Creek during the 2005–06, 2006–07 and 2007–08 wet seasons. The 80th, 95th and 99.7th percentiles of the difference in the logarithms of the event mud loads are marked so that potentially impacted events (indicated by date) can be identified.

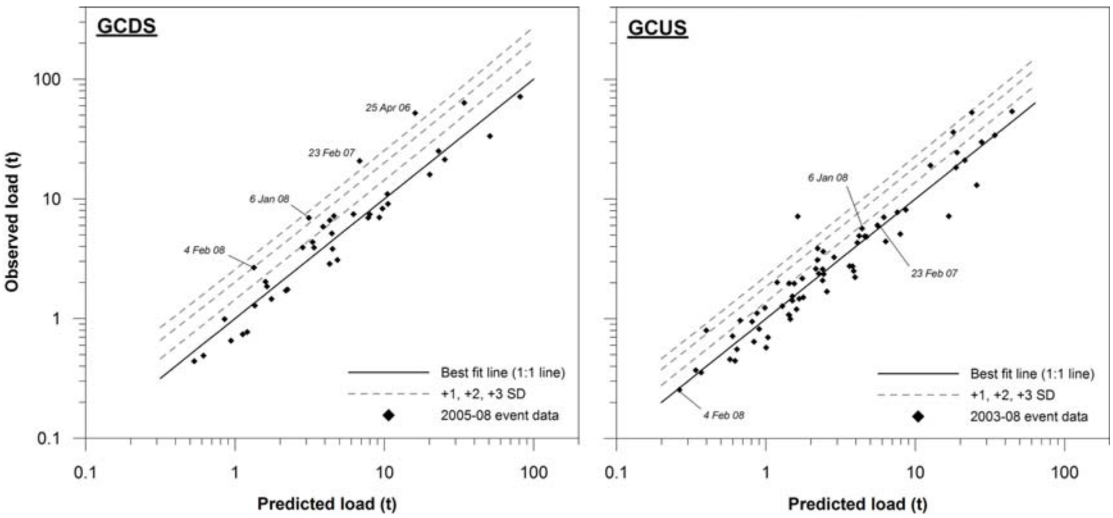

Figure 3 Event-based mud load relationships and associated +1, +2, and +3 standard deviation lines for GCDS (Left) and GCUS (Right). Discrete event data collected during the monitoring period are marked as dots, with potentially impacted events shown with event description and date. (Note: no mud load data were collected at GCUS during the event on 25 April 06.)

As discussed in the companion paper 'Turbidity and suspended sediment management guidelines and trigger values for Magela Creek', it is considered that an impacted event is one with a significantly elevated mud load compared to both the mud load measured upstream and the corresponding event discharge characteristics observed at the downstream site. Data collected within the Gulungul Creek catchment since 2003 show that there have been two events that would be classified as 'impacted' (23 February 2007 and 4 February 2008). These events are identified as elevated by both BACIP and the mud load-discharge relationship impact assessment techniques (Figs 2 & 3). Prior to the 2006–07 wet season, construction works commenced to elevate the tailings dam wall for increased storage capacity (Jacobsen 2007). It is possible that mud was washed into the main Gulungul Creek channel downstream of GCUS as a result of relatively intense rainfall over the exposed soil associated with earthworks around the perimeter of the tailings dam (Moliere & Evans 2008).

Channel stability (Aim 2)

To investigate the channel stability on Gulungul Creek, 12 cross-sections were installed in 2002 between GCUS and GCDS. These cross-sections have been surveyed annually and generally exhibit scour of the bed sediments over time indicating naturally driven net export of bed sediments along the creek (Saynor et al 2005, Saynor & Smith 2006). Figure 4 shows the cross-section where the most scour has occurred . This trend has been more pronounced during the last two survey's (2006 and 2007), which were conducted after Cyclone Monica passed through the region late in April 2006 and after the largest recorded flood occurred early March 2007, respectively. Subsequent surveys of the channel cross-sections will show if these events had any long-term impacts on the channel stability.

To further investigate scour and fill of the bed sediments, scour chains were first installed in 2002 at 6 of the 12 cross sections. Late in each dry season (when the watertable has dropped to its lowest level) the scour chains are located, measured and re-installed to current bed level (resetting the datum). In some cases, the scour chains could not be located as the water table had not dropped sufficiently to enable their recovery. Bed material sediments were also collected from the cross sections during each survey. These sediments have been characterised by particle size analysis to allow comparisons between the years. The samples collected late in 2007 are currently being analysed in the laboratory with the results to be incorporated into an internal report.

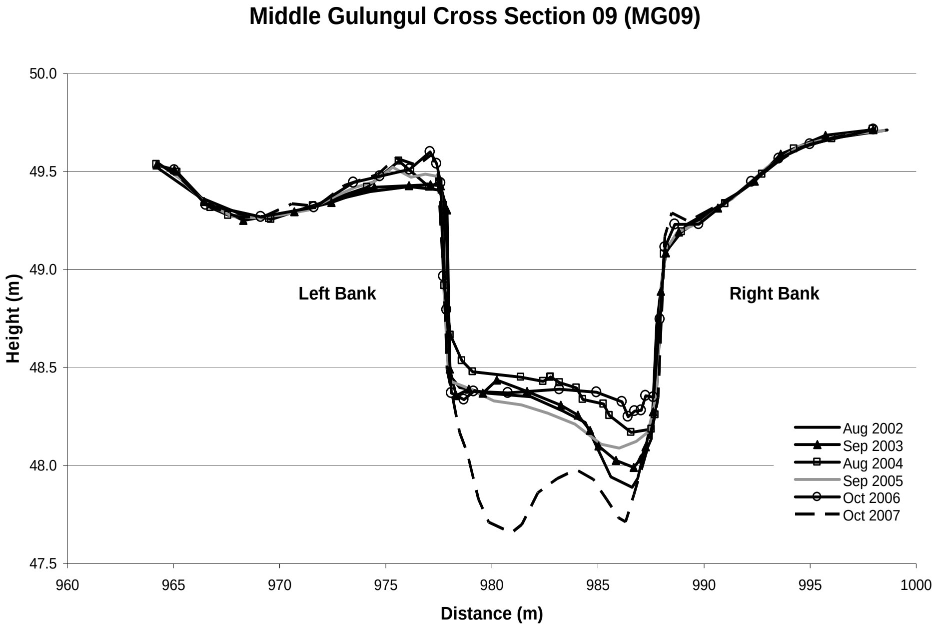

Figure 4 Cross section survey data collected since 2002 at middle Gulungul cross-section 09 (MG09)

Conclusions and future work

The combination of BACIP and regression model techniques has identified two possible mine-related impacts on event mud loads measured downstream of Ranger in Gulungul Creek since 2003. Future measured event load data which plot above the 'action' trigger levels derived from these methods of sediment load analysis should prompt further investigation and management action, if required.

Scour of the bed sediments was generally more pronounced in 2006 and 2007 after Cyclone Monica and an extraordinary flood event occurred, respectively. Subsequent surveys of the channel cross-sections will show if these events had any long-term impacts on the channel stability. Particle size analysis of the bed materials still needs to be completed.

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Development of catchment geomorphic characteristics of Gulungul Creek – gauging station upgrades

D Moliere, G Staben, M Saynor & R Houghton

Background

During the 2006–07 wet season, an extraordinary rainfall event occurred across the region over a 3-day period between 27 February and 2 March 2007, resulting in the highest flood levels in the Gulungul Creek catchment since recording began in 1971 (Moliere et al 2007). Temporary gauging stations located in the catchment upstream (GCUS) and downstream (GCDS) of the Ranger mine (see previous paper 'Development of catchment geomorphic characteristics of Gulungul Creek – monitoring results' for locations) were submerged by floodwaters (Fig 1), resulting in damage to equipment and data loss during the peak and subsequent recession of the flood. Given the substantial extension to the life of the Ranger mine, and the potential for increased impact in the Gulungul catchment, a strategic decision was made to replace these stations with structurally engineered, permanent stations.

Figure 1 Gauging station at GCDS submerged by floodwaters during the March 2007 flood. This photo was taken several hours after the flood peak when waters were at least 0.5 m higher than that shown here.

Station upgrade

The new stations, designed in-house with external engineering review and approval, were built late in the 2007 Dry season (Fig 2). The instrument platforms were elevated at least 1 m above the peak water level height recorded during the March 2007 flood. Both the design and the construction of these stations constituted a large part of the HGP 2007–08 work effort to have the stations in place and the new instrumentation operational for the 2007–08 wet season. In addition to this, the gauging station in Ngarradj Creek downstream of the Jabiluka Mineral Lease also required rebuilding as a result of flood damage.

The stations were equipped with a new generation of dataloggers (selected after considerable market research) to provide the capability of being able to remotely access the outputs of all connected sensors and also being able to trigger the automatic water samplers installed at the stations by predetermined output levels from the water quality or flow sensors. Prior to the upgrade, not all of the flow and water quality sensors could be connected to the original dataloggers, which meant some of the data had to be manually downloaded in the field. Therefore, the upgrade of both the stations and the logging equipment represents a significant improvement to the reliability and efficiency of data collection from our remote sites. The new generation loggers were installed at all SSD stations along Gulungul Creek, Magela Creek and Ngarradj to provide uniformity across the sites.

Figure 2 Original station at GCUS (left) and the new permanent gauging station constructed late in the 2007 dry season (right)

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Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on Gulungul Creek catchment, Ranger minesite and Nabarlek area

G Staben, MJ Saynor, DR Moliere, GR Hancock[1] & KG Evans

Introduction

The title of this project in the 2007-2008 workplan is 'Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on stream sediment loads resulting from tree fall in the Gulungul Creek catchment'. However in addition to the Gulungul creek catchment this paper is also reporting on data collected from the Ranger mine site and Nabarlek area.

During April 2006, severe tropical Cyclone Monica impacted the coast of northern Australia, including the Alligator Rivers Region (ARR). The very destructive core of Monica (category 5) crossed the Northern Territory coastline approximately 35 km west of the township of Maningrida with maximum gust speeds estimated to be 360 km h^{-1} (Australian Bureau of Meteorology 2008). It continued to move inland in a south-westerly direction, rapidly weakening in intensity. Based on the track map produced by the Bureau of Meteorology, and satellite imagery, the eye of the cyclone passed over the rehabilitated Nabarlek minesite, with estimated maximum wind gusts of 180 km h^{-1}. It then passed directly over the Ngarradj sub-catchment (maximum wind gusts of 140 km h^{-1}) in Kakadu National Park (KNP), continuing through the Gulungul Creek sub-catchment and the town of Jabiru (Figure 1). Wind speed estimates for Nabarlek and Ngarradj were derived from a logistic decay curve equation in Cook and Goyens (2008).

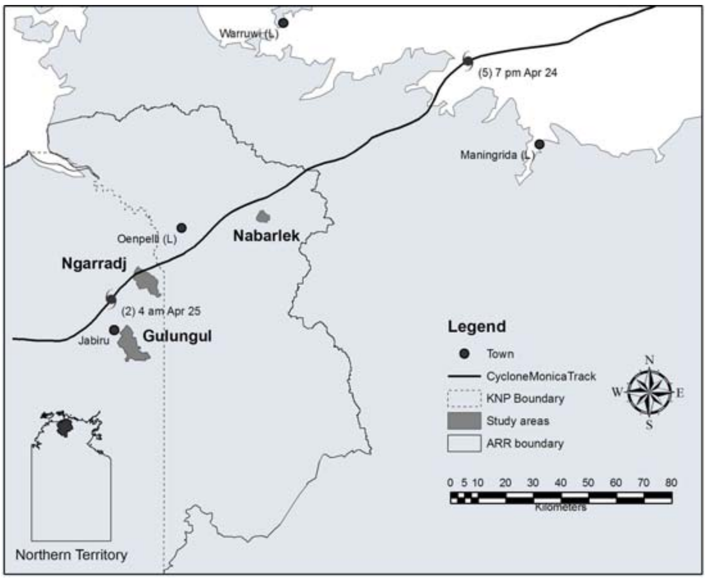

Figure 1 Estimated track of Cyclone Monica across the Alligator Rivers Region

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By the time winds gusts affected the Gulungul catchment, wind speed had reduced to a category 2 level with maximum destructive wind gusts of 135 km h^{-1}. Monica then continued to track westerly, weakening to below cyclone intensity 12 h after first making landfall.

Remote sensing and fieldwork were undertaken to assess the impact the cyclone had on the catchments within the Ranger and Jabiluka leases. The findings from the remote sensing study were reported in the 2006–07 SSD annual report and recently in Staben and Evans (2008) The initial results from the extensive on-ground field data collected in the months after cyclone Monica in the Nabarlek lease area, Gulungul Creek catchment, and rehabilitated sites located at the Ranger minesite are provided here.

Field data

A total of fifty-five 30 x 30 m plots were sampled, 31 in the Gulungul Creek Catchment, 15 at Nabarlek, and 9 on the Ranger minesite. Selection of the sites in the Gulungul catchment was undertaken using a stratified random sampling approach. Five broad vegetation communities were derived from a Landsat TM5 satellite image (acquired on 15 April 2005) using an unsupervised Isodata classification. Six survey sites were then randomly selected within each of the vegetation classes using the Hawths Tools extension in ArcMap v.9. The escarpment region within the catchment was excluded from fieldwork due to sacred site access restrictions.

Slightly different methods were used to select sites on both Nabarlek and Ranger. The selection of the 15 sites on Nabarlek mine lease were based on seven land units taken from the Land Units of the Nabarlek Mine Area, Northern Territory 1:5 k dataset created by the Northern Territory Government (Day & Czachorowski 1982). Three plots were located in rehabilitated areas and the remainder in natural vegetation communities. Plot sites at Ranger were located on experimental rehabilitation areas with three different soil treatments. The 31 plots in the Gulungul catchment were located within natural vegetation communities while the 9 plots on the Ranger minesite were located on revegetated areas of waste rock.

A number of parameters were measured for trees within each plot including: identification to species level of all trees ≥ 2 m in height (fallen and standing), diameter at breast height (DBH), tree orientation, living or dead and amount of damage. In addition, each tree was assigned one of eleven status codes (Table 1) describing the level of physical impact of the cyclone.

Table 1 Eleven status codes used to describe the level of impact of cyclone Monica on each tree ≥ 2 m in height

Status code	Status explanation	Status code	Status explanation
AS	Alive standing undamaged	DU	Dead uprooted
ASS	Alive standing snapped trunk	DL	Dead leaning
ASB	Alive standing broken limbs	DSN	Dead standing snapped
AU	Alive uprooted	DSC	Main trunk dead standing coppicing at base
AL	Alive leaning	DUC	Main trunk dead uprooted coppicing at base
DS	Dead standing		

The dimensions of the crater and the volume of material uplifted, termed pit and mound in the literature (Putz 1983, Norman et al 1995), caused by tree throw and potentially available for future erosion, were also measured. This was done to assess whether it was likely that the displaced material would be washed back into the depression by rainfall or be moved to the surrounding surface and be transported by overland flow. Particle size analysis (PSA) of soil samples collected from uplifted soil and nearby undisturbed surface soil was done to provide estimates of erosion potential. The potential for movement of soil will be influenced by the surface grade (slope) in the plot area where the trees have fallen. Slope angle of the plots sampled was usually less than 5%.

Results

A total area of 4.95 ha containing a combined total of 3049 trees was surveyed in the three study areas. For the initial analysis, the 11 status codes describing the impact of Cyclone Monica on each of the trees measured were pooled into three broad categories;

- undamaged (AS)
- damaged (AL, ASB, ASS, AU)
- dead (DS, DU, DL, DSN, DSC, DUC)

The results show that 13.8% of the trees in the 31 plots located in the Gulungul catchment had died, with a further 25.5% suffering some form of damage. The remaining 60.7% were undamaged. Of the 765 trees impacted by Cyclone Monica at Nabarlek, 36.5% died, 21.8% were recorded as suffering some form of damage, and 41.7% were undamaged (Table 2).

Table 2 Summary of the impact of Cyclone Monica on trees for each study area

Status post Cyclone Monica	Gulungul (Cat 2)*	Ranger minesite (Cat 2)*	Nabarlek (Cat 3)*
N trees	1579	705	765
Alive undamaged **	958 (60.7%)	321 (45.5%)	319 (41.7%)
Alive damaged	403 (25.5%)	10 (1.4%)	167 (21.8%)
Dead	218 (13.8%)	374 (53 %)	279 (36.5%)

* Cyclone category. ** percentage of total in parenthesis

On the rehabilitated Ranger sites 53% of trees were recorded as dead, 1.4% were alive with damage and 45.5% were not impacted by the cyclone (Table 2). Results of the three rehabilitated plots located at Nabarlek appear to show a similar trend to the Ranger sites. It appears that trees, on rehabilitated areas, which have been significantly impacted by a cyclone are more likely to suffer mortality (Table 3). It must be noted that it is possible that the occurance of fire at both the Ranger and Nabarlek plots may be contributing to the results observed, this requires further research.

Table 3 Summary of the impact of Cyclone Monica on trees in rehabilitated mine sites

Status post Cyclone Monica	Ranger minesite (Cat 2)* 9 sites	Nabarlek (Cat 3)* 3 sites
Number of trees	705 (27 species)	151 (15 Species)
Alive undamaged	321 (45.5%)	80(53%)
Alive damaged	10 (1.4%)	10 (6.6%)
Dead	374 (53%)	61 (40.4%)

* Cyclone category

The eleven status codes were then pooled into six broad categories to provide capacity for more detailed assessment of the data:

- undamaged (status code AS)
- broken limbs (status code ASB)
- leaning (status code AL, DL)
- trunk snapped (status code ASS, DSN)
- uprooted (Status codes, AU, DU, DUC)
- dead standing (status codes DS, DSC)

Of the total number of trees affected by Cyclone Monica in the 31 plots in Gulungul catchment 15.6% (246) had some form of limb damage, 3.2% (41) were leaning as a result of the cyclonic winds, 2.6% (51) were snapped at the main trunk, 10.9% (173) were uprooted and 6.9% (110) were dead standing. While the number of trees unaffected by the cyclone represented just over 60% of the total trees, the percentage of tree basal area[2] impacted was far greater with. Just over 65% of the total stand basal area (8.15 m^2 ha^{-1}) in the 31 plots occurring in the 5 categories represented cyclone damage. Trees with broken limbs represent 34% of the total basal area, uprooted 21.5%, snapped 4.4%, dead standing 3.1% and leaning 2% (Figure 2 (a) & (b)).

Just over 58% (446) of the total trees surveyed at the Nabarlek rehabilitated mine site were impacted by the cyclone. A total of 26.2% (201) of the trees were uprooted, 15.3% (117) had snapped main trunks, 13.1% (100) were dead standing, 2% (15) had broken limbs and 1.7% (13) were leaning. The total basal area for the 15 plots in Nabarlek was 6.2 m^2 ha^{-1} with trees in the uprooted status representing 37.4% of the total basal area. Undamaged trees represented 26.2%, trees with snapped trunks represented 19.7%, broken limbs 9.9%, dead standing 5.5% and trees leaning 1.3% (Figure 2 (a) & (b)).

Assessment of the impact of the cyclonic winds on trees within the 9 plots on the Ranger mine site found that of the majority of the 705 individuals were either recorded as undamaged 45.5% (321) or dead standing 41.4% (292). Of the remaining trees 7.5% (53) were uprooted, 4.3% (30) had snapped trunks and 1.3% were leaning. Again trees in the undamaged and dead standing categories represented the majority of the total basal area (3.5 m^2 ha^{-1}) with 45.5% and 36.6% respectively. Uprooted trees represented 11% of the total basal area, snapped trunks 7.1% and trees leaning 1.2% (Figure 2 (a) & (b)).

This initial analysis of the field data collected at the three study areas shows that the level of impact of cyclone Monica on trees was greatest at the Nabarlek site. The maximum distance between any one of the survey plots located in the Gulungul creek catchment and the Ranger mine is ~ 10 km. It could be expected due to the geographic location of the survey plots in Gulungul creek catchment and Ranger mine that the level of impact may have been similar. However, the results clearly show that the trees on the Ranger mine site suffered similar levels of damage to Nabarlek, which experienced greater wind speeds than both Gulungul and Ranger. The results observed in the analysis to date may be due to the differences in soil types/structure, morphological differences between tree species or the occurance of fire. Further analysis of these data is being undertaken to investigate these possibilities.

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[2] Tree basal area is the cross-sectional area (over the bark) measured at breast height (1.3 metres above the ground) measured in metres squared (m). Tree basal area is a measure of tree volumes and stand competition.

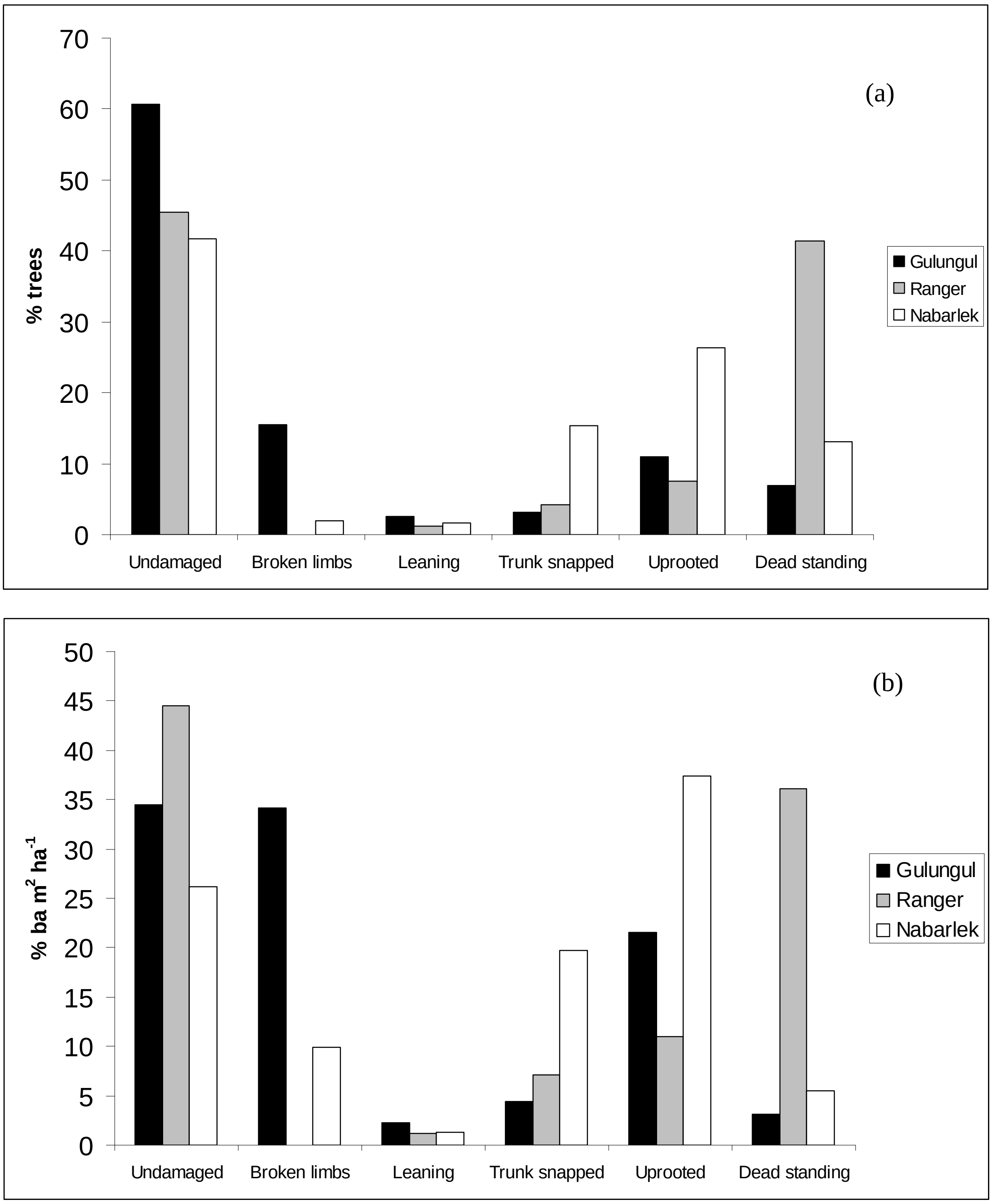

Figure 2 Number of trees (a) and total basal area (b) represented as a percentage for each of the pooled status codes describing cyclone impact in the three study areas

Sediment transport

The total volume of sediment that was contained within root balls in the uprooted trees in the three catchments is shown in Table 4. It was expected that the sediment contained within the rootballs would fall mainly back into the crater or onto the ground surrounding the crater hole. Observations at some of the sites that were visited during the 2007 dry season suggest that the little sediment that had been displaced from the rootball had fallen back into the craters although this is not quantifiable.

The gently sloping land surface (<5% grade) of the plots suggested a priori that displaced soil was unlikely to be transported any great distance from the area of immediate disturbance. Suspended sediment studies in Gulungul Creek catchment have found that despite a large rainfall and runoff event during the cyclone giving elevated stream mud loads at the time, the sediment transport characteristics within the catchment during 2006–07 and 2007–08 were not significantly different to previous years (Moliere & Evans 2008). This finding implies that the disturbance of the soil profile by tree throw has not had a significant impact on the net export of fine suspended sediment in Gulungul Creek. Thus the effect of a cyclone (at least to Category 3) on export of fine soil material appears to be relatively minor.

Table 4 Volume of sediment in the rootballs of fallen trees in each study area

Study site	Fallen trees with rootball	Total volume of rootball (m^3)	Total hectares sampled (ha)	Total volume per hectare ($m^3 ha^{-1}$)
Gulungul catchment	121	23.60	2.79	8.46
Nabarlek minesite	154	10.03	1.35	7.43
Ranger minesite	45	0.75	0.68	1.10

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Turbidity and suspended sediment management guidelines and trigger values for Magela Creek

D Moliere & K Evans

Background

Fine suspended sediment (the mud or < 63 µm fraction) moves through stream systems in pulses or waves generated by rainfall/runoff events. Reliable impact assessment requires a method for quantifying the mud loads transported by these pulses, such that inputs coming from the subcatchments adjacent to the mine can be clearly distinguished from inputs from the natural catchment upstream of the mine.

Mud load event data, using turbidity as a surrogate, collected during the past three wet seasons from upstream and downstream of Ranger along Magela Creek (MCUS and MCDS) have been used to quantify the magnitude of mud loads and assess whether the source is natural or potentially mining related. The locations of the monitoring stations are shown in Figure 1.

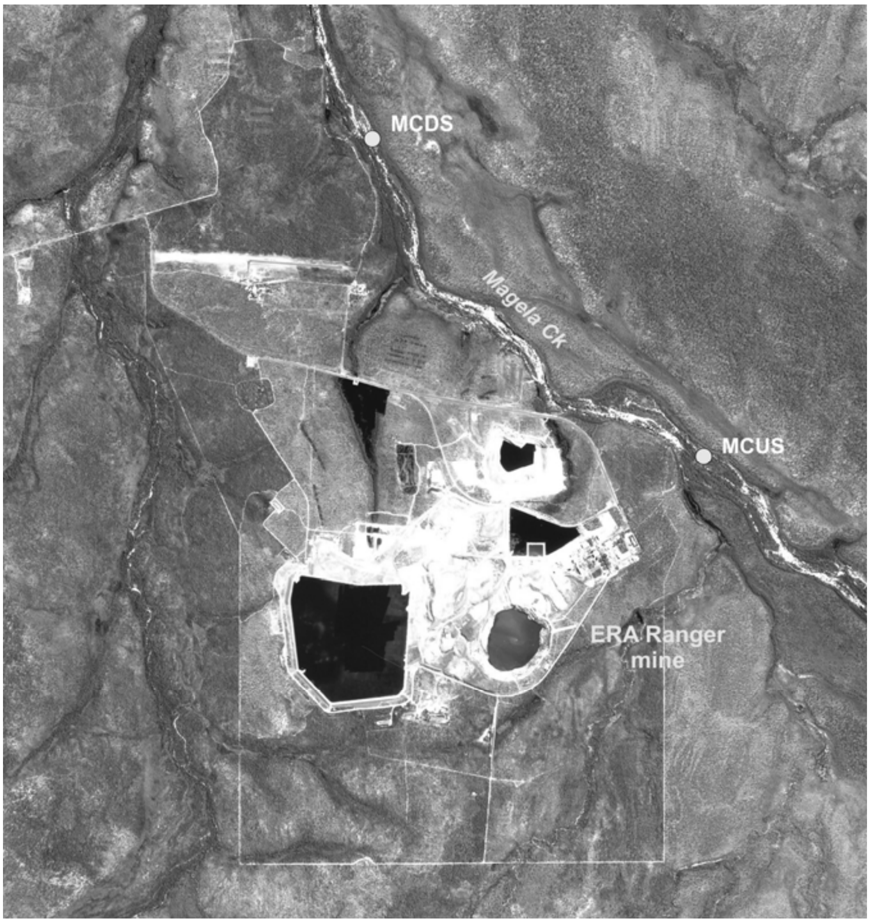

Figure 1 Location of the monitoring stations along Magela Creek

Trigger levels (which can be used for future impact assessment) for event mud loads have been derived for current pre-rehabilitation conditions using two complementary methods (Moliere & Evans 2008).

Impact assessment methods

BACIP Analysis (Method 1)

The basis of BACIP analysis (Stewart-Oaten et al 1986, 1992) is to produce a set of paired-site (P) 'difference' values by subtracting mud load data from the upstream (Control (C)) site from those at the downstream (Impact (I)) site. The mud load measured downstream of the disturbance is thus compared to that measured upstream on an event basis to assess impact.

'Trigger' values derived from statistical percentiles of the difference data distributions provide the basis for assessing the significance of the event mud load data for a particular event. For this work the BACIP analysis uses the differences between log-transformed event mud loads at the upstream and downstream stations along Magela Creek.

Event mud load data collected during 2005–06, 2006–07 and 2007–08 for Magela Creek were used to establish preliminary trigger values for the event-based BACIP analysis. Given that the log-transformed difference data are non-normally distributed, water quality management trigger levels are assigned to the 80th, 95th and 99.7th percentiles of the data. These trigger values represent 'focus', 'action' and 'limit' triggers, respectively (Supervising Scientist 2002), and define an escalating hierarchy of management response. Figure 2 is essentially a control chart plot that enables the significance of events to be assessed.

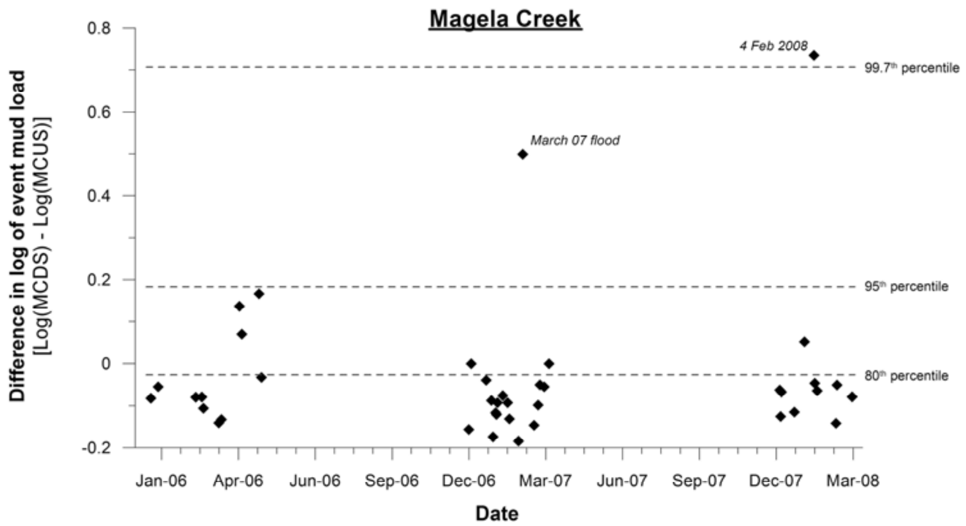

Figure 2 Temporal variation of the difference in the logarithms of the discrete event mud loads (indicated as ♦) measured in Magela Creek during the 2005–06, 2006–07 and 2007–08 wet seasons. The 80th, 95th and 99.7th percentiles of the difference in the logarithms of the event mud loads are marked so that potentially impacted events (indicated by date) can be identified.

Events that lie above the 95th percentile ('action' trigger) should be specifically investigated to determine the cause of the elevated mud load measured downstream relative to the load upstream. Events where mud loads measured downstream have exceeded the 'action' trigger level are highlighted by specific date labels in Figure 2.

Relationship between mud load and discharge characteristics (Method 2)

This approach uses a site-specific regression relationship between event mud load and corresponding event discharge characteristics.

Stepwise multivariate regression analysis between event mud load and a selection of discharge characteristics (such as total effective rainfall, maximum periodic rainfall intensity, total event runoff, total discharge of the rising stage of the hydrograph, maximum periodic rise and recovery period preceding the event) showed that the most significant discharge characteristics for predicting event mud load within the region are total event runoff and maximum periodic rise in discharge (Moliere & Evans 2008).

The form of the regression equation is as follows:

$$\text{Total mud load} = K\left(Q_T\right)^a Ri^b \qquad (1)$$

where Q_T is total discharge during the mud pulse, Ri is maximum periodic rise in discharge over 10 minutes and a, b and K are fitted parameters.

Event data collected between 2005 and 2008 were used to fit statistically significant regression relationships (Equation 1) between event mud load and corresponding event discharge characteristics for each site (Figure 3). Given that observed loads are normally distributed around the best-fit line (predicted loads), 'focus', 'action' and 'limit' trigger levels correspond to +1 standard deviation (SD), +2 SD and +3 SD from the 1:1 line, respectively (Supervising Scientist 2002). Events that plot above the +2 SD line ('action') have a significantly higher mud load than would be predicted using the corresponding event runoff characteristics for a non-impacted condition. Using the regression model approach, a potentially impacted event is one in which the mud load measured downstream of Ranger is significantly elevated compared to the corresponding event discharge characteristics (ie lies above the + 2 SD line). An impact is confirmed if the corresponding event mud load measured upstream of Ranger is not significantly elevated compared to the flow discharge characteristics (ie lies within + 2 SD of the fitted relationship).

Mud loads for discrete turbidity events observed in Magela Creek are shown plotted as dots against the predicted event mud loads in Figure 3. Possible minesite related events are identified by specific date labels.

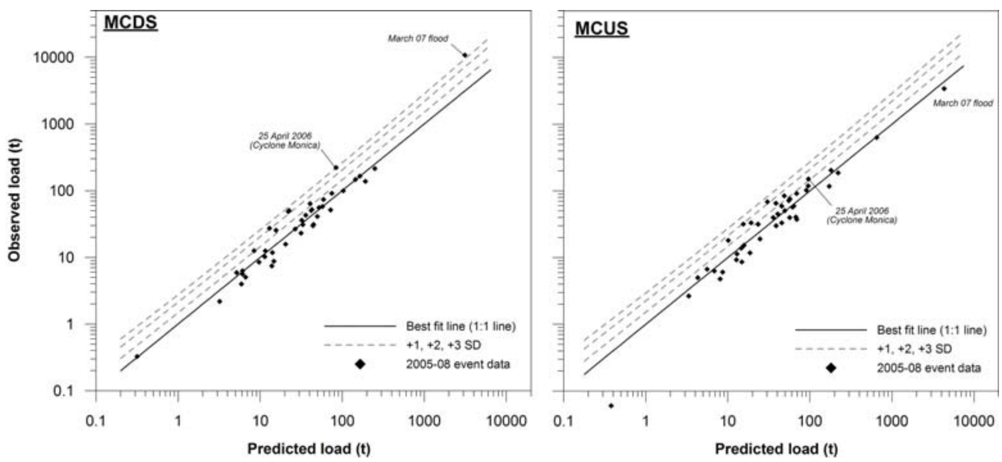

Figure 3 Event-based mud load relationships and associated +1, +2, and +3 standard deviation lines for MCDS (Left) and MCUS (Right). Discrete event data collected during the monitoring period are marked as dots, with potentially impacted events shown with event description and date.

Discussion

There is generally good correlation between the BACIP and regression model approaches to assess impact on mud loads. However, the results have shown that the impact assessment methods (treated separately) may not be reliable for all flow and storm conditions. For example, the event on 4 February 2008 at Magela Creek was associated with an isolated storm that occurred over a subcatchment between the upstream and downstream stations. BACIP analysis indicated that the mud load measured at the downstream station was elevated compared to that measured upstream (Fig 2) and, as such, warrants further assessment. However, the measured mud load during this event at the downstream station was well within the trigger levels derived from the regression model. That is, the mud load was reasonable for the flow conditions (Fig 3) and as such would not be considered impacted.

Therefore, it is recommended that a combination of BACIP and regression model techniques be used for impact assessment on mud load within the Magela Creek catchment. Using such an approach, an impacted event is one with a significantly elevated mud load compared to both the mud load measured upstream and the corresponding event discharge characteristics observed at the downstream site. Data collected along Magela Creek since 2005 show that there has been one event that would be classified as ‘impacted’ (ie identified as elevated by both BACIP and the mud load-discharge relationship impact assessment techniques). During the March 2007 flood, which was a record rainfall–runoff event for the region, the sediment management controls on the minesite area were simply overwhelmed (Moliere et al 2008). The subsequent input of turbid water from the site, including exploration drill pad areas to the east of Pit 3, into minesite tributaries and hence into Magela Creek, contributed to the elevated mud concentrations recorded downstream of Ranger (Figures 2 & 3).

Conclusions and future work

The combination of BACIP and regression model techniques has identified one possible mine-related impact on event mud loads measured downstream of Ranger since 2005. It is recommended that impact assessment of mud loads continue to use both approaches, given their complementary strengths and weaknesses.

Future measured event load data which plot above the ‘action’ trigger levels derived from these methods of sediment load analysis should prompt further investigation and management action, if required. The investigation should include analysis of the suspended sediment to ascertain if it has come primarily from the minesite or from adjacent un-impacted subcatchment areas. This distinction needs to be made since elevated sediment loads could result from the impact of an intense local storm event on land that is not impacted by mining activity, yet lies within the lease boundary. Water samples for this analysis will be provided by the event triggered autosamplers located at the downstream sites in Magela Creek.

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Part 3: Jabiluka

Monitoring sediment movement in Ngarradj

D Moliere, M Saynor & K Evans

Background

During the 2007–08 wet season, continuous rainfall, runoff and turbidity data were collected from JSC, the gauging station located downstream of Jabiluka within the Ngarradj catchment. Baseline suspended sediment and hydrology data have now been monitored at this site over a 10 year period. Data collection at two stations upstream of Jabiluka (ET and UM) ceased after the 2006–07 wet season. The locations of all three monitoring stations are shown in Figure 1.

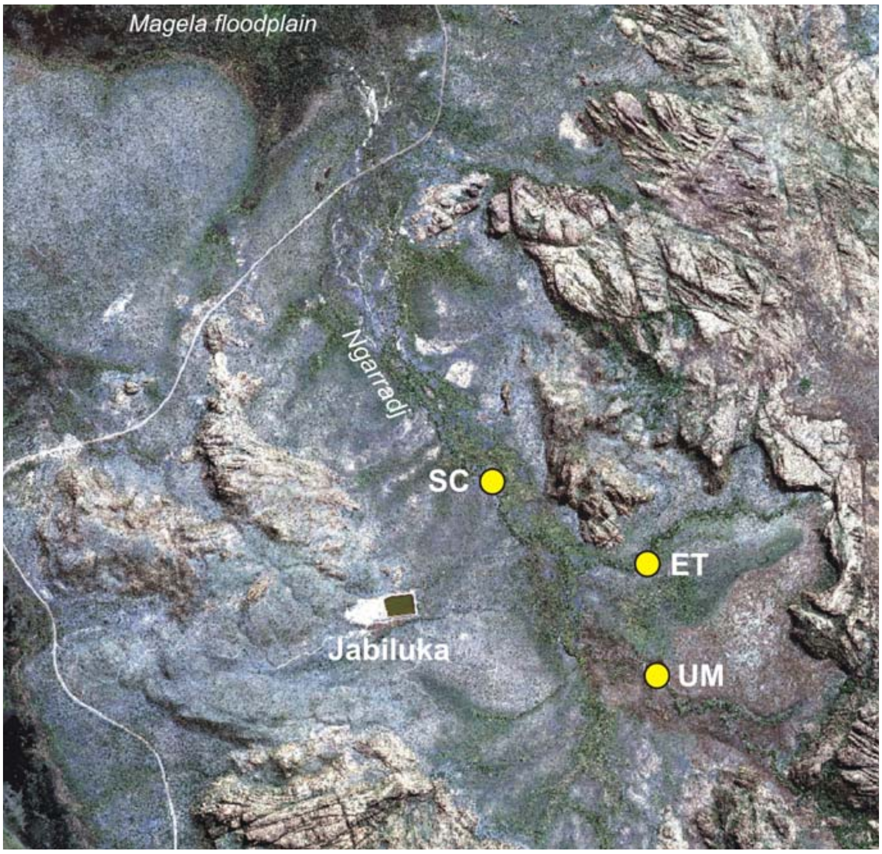

Figure 1 Location of the monitoring stations along Ngarradj

The fine suspended sediment (FSS) data, measured indirectly from turbidity, and hydrology data collected from the site will be used to derive indicators for minesite impact in the event that the Jabiluka project proceeds. The FSS data collected since the 2006–07 wet season will be especially important in the context of Cyclone Monica, which occurred in April 2006. In particular, comparison between the suspended sediment load data collected before and after this event to indicate whether there has been a change in 'pre-mining' sediment transport characteristics as a result of this catchment wide impact.

Progress

As discussed in the ARRTC 20 meeting paper for this project, an extraordinary rainfall event occurred over a 3-day period between 27 February and 2 March 2007, which resulted in the highest flood levels recorded within the Ngarradj catchment since 1998. Equipment at the downstream station was submerged by floodwaters, resulting in damage to equipment and data loss during the peak and subsequent recession of the flood. Consequently, the instrument platform at SC was elevated late in the 2007 Dry season at least 1 m above the peak water level height recorded during the March 2007 flood. Similar to that at the Gulungul Creek stations (see companion paper under this KKN – 'Development of catchment geomorphic characteristics of Gulungul Creek – gauging station upgrades'), the station was equipped with the same new type of datalogger. However, remote dial in access was not possible at this site since it is outside the coverage of the mobile phone network.

Data collected at SC during the 2007–08 wet season has not yet been analysed. It is planned that event FSS load data collected during the 2007–08 wet season will be plotted against the relationship between mud load and discharge characteristics derived from previous years data (Moliere & Evans 2008) to assess the sediment load charateristics for that season.

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Part 4: Nabarlek

Assessment of ingestion doses to people accessing the Nabarlek site

A Bollhöfer & B Ryan

Introduction

The main tasks remaining for the radiological risk assessment of the rehabilitated Nabarlek site include the determination of ingestion doses to Aboriginal people who may potentially access or spend time in the vicinity of the site pursuing customary bushtucker harvesting activities. Previous studies in the Alligator Rivers Region have shown that during the operational phase of a mine site, when access to the operational area is restricted, the downstream surface water pathway is the main potential contributor to ingestion doses via ingestion of mussels that bioaccumulate ^{226}Ra (Martin et al 1998).

Until recently the Nabarlek site could be easily accessed, despite not yet having been closed out by the regulatory authority, and the ingestion of terrestrial flora and fauna could potentially be an important component of the total ingestion dose. However, as of 30 June 2008, the mineral lease and associated responsibilities for managemen of the site has been taken over by the Adelaide-based company Uranium Equities Ltd (UEL). UEL is conducting an active exploration drilling program on and off-site, to investigate potential for below ground uranium anomalies. This work may impose access restrictions to the site in the future.

A number of studies have been conducted over the years measuring radionuclide activity concentrations in traditional Aboriginal foods and determining uptake factors for radionuclides, including terrestrial fauna and flora (Martin et al 1998, Martin & Ryan 2004, Ryan et al 2005). However, most of the data were acquired in the vicinity of Ranger mine or Jabiru, as accessibility of Nabarlek during the wet season, which constitutes the main fruiting season for many bushfoods, is poor. However, soil radionuclide activities in the Nabarlek region and activities in sediments of the riparian zones are known (Martin et al 2006, Frostick et al 2008) and concentration factors determined from other studies in Northern Australia can be used to model ingestion doses. These concentration factors are reported in Section 2.2.4: *Radiation exposure pathways associated with ecosystem establishment*.

Methods

Diet assessment

The model diet presented here has been modified from a study by Altman (1984) who investigated the dietary habits and the protein and calory intake of inhabitants of the Momega outstation in Central Arnhem Land, approximately 85 km east of the Nabarlek area. It assumes that 80 per cent of the protein intake in the diet is from traditional bush foods. According to Altman (2003) the estimates are still in general agreement with more recent estimates of the daily game intake for people living on outstations in Arnhem Land.

ICRP Publication 23 states that the per caput estimate of food supplies for Reference Man from Oceania (Table 122, page 349) is 677 kg/yr. If the shop bought food contributed 20% to the total food intake as suggested by Altman (1984, 2003), this would give an annual bushfood intake of approximately 538 kg/yr. The estimated annual intake of bushfood of local

Aboriginal people was collated in tabular form and provided to Aboriginal people living near Maningrida, Central Arnhem Land, for comment and correction (Table 1). For a 10 year old child the intake has been halved.

The model diet reported here is somewhat different to the diet reported for Aboriginal people in the Northern Kakadu region (Ryan et al 2008) as shop bought food is more difficult to access for people living in Arnhem Land (although most of the traditionally consumed vegetables are now replaced by shop bought items (Altman 1984)). The annual intake of wallaby (*Macropus agilis*) and pig (*Sus scrofa*) is higher than in the Northern Kakadu area, as is the annual intake of fish. The intake of feral water buffalo (*Bubalus bubalis*) and cattle (*Bos taurus*, *Bos indicus*) is large(r) as well, as they are prevalent throughout Arnhem Land.

Table 1 Estimate of the annual intake of bushfood of local Aboriginal people in Arnhem Land

Food item	Kg/y per person
Buffalo	200
Pig	120
Wallaby	70
Fish	70
Magpie goose + waterfowl	30
Turtle	10
Goanna	0
File snake	3
Crocodile	1
Shrimp	2
Mussels	2
Yam	20
Fruit	8
Water lily	2
Total food kg/yr	538

Typically smaller family groups spread throughout Arnhem Land by late August, when surface fresh water supplies disappear. However, the groups have a relatively small range in the early wet season and are virtually sedentary during the wet (Kohen 1995, Altman 1984). Consequently, we have assumed 1 month access to Nabarlek during the dry season only, taken into account seasonal hunting times.

Dose estimate

To assess the terrestrial pathway it was assumed that animals such as buffalo with a relatively large range spend 90% of the time grazing on environmental areas other than the Nabarlek mine site. Wallaby, with a much smaller range, were assumed to spend most of their time on-site, whereas pigs were assumed to spend about half of the year along the riparian zones along Cooper Creek in the greater Nabarlek area.

Table 2 in 'Bioaccumulation of radionuclides in terrestrial plants on rehabilitated landforms shows the concentration factors used in our assessment, and their origin' (this volume, 152–159).

^{226}Ra and ^{232}Th soil activity concentrations of 63 Bq·kg^{-1} and 20 Bq·kg^{-1}, respectively, determined from groundtruthing an airborne gamma survey (AGS) upstream of the Nabarlek site were used to provide a local background (environmental baseline) value. Area weighted soil activity concentrations within the fenced area (on-site) are 454 Bq·kg^{-1} and 41 Bq·kg^{-1}, respectively (Martin 2000). In addition, soil activity concentrations measured in sediment cores taken from overflow areas along Cooper Creek and the estimated contribution from the Nabarlek mine site (1 Bq·kg^{-1} for U and 2 Bq·kg^{-1} for ^{226}Ra) (Frostick et al 2008) were used as environmental and mine influenced soil activity concentrations, respectively, for pigs (and also for water lily), in conjunction with the respective concentration factors.

Radioactive equilibrium of all progeny in the soils was assumed, unless measurements of progeny were available, and average flesh radionuclide activity concentrations have been calculated from known concentration factors.

Buffalo/Cow: concentration factors (CFs) for feral buffalo from the Magela floodplain (Martin et al 1995, 1998) were used as an analogue for cattle, in combination with environmental and on-site soil activity concentrations.

Pig: CFs are from Martin et al (1998, 1995) from the Magela floodplain.

Wallaby: CFs have been determined from measurements of flesh activity concentrations in wallaby from around Ranger mine and Maningrida in western Arnhem Land (Ryan 2002, unpublished) and were used in the dose calculations, in combination with environmental and on-site soil activity concentrations.

Fish: CFs are form Martin et al (1995, 1998) and flesh activity concentrations have been estimated using the measured ^{226}Ra activity concentration in water in Cooper Creek.

Magpie go ose + waterfowl: Magpie Geese flesh activity concentrations from animals collected at Red Lily Billabong in the South Alligator disctrict and from Maningrida in Western Arnhem Land (Ryan 2002, unpublished) were used in the dose calculations.

Turtle: CFs for turtle have been determined for animals collected at Bowerbird Billabong (Martin et al 1998). These have been used in conjunction with ^{226}Ra activity concentration in water in Cooper Creek.

File Snake/Crocodile: CFs for file snake and crocodile flesh are from Martin et al (1998).

Shrimp: Freshwater shrimp activity concentrations were calculated from CFs in Martin et al (1998) and Cooper Creek water activity concentrations.

Mussels: Actual mussel flesh activity concentrations from mussels collected from the Goomadeer River in western Arnhem Land were used, rather than using concentration factors determined previously, as concentration factors depend on the location where mussels are sampled. It was furthermore assumed that for 1 month mussels were collected in Cooper Creek (activity concentration data for mussels are available from the 1980s), whereas the remaining 11 months mussels are collected from Goomadeer River, approximately 35 km east of Nabarlek.

Yams: CFs were used from Ryan et al (2005). In addition, it was assumed that a person spending 1 month at Nabarlek collects all yams consumed from the mine site.

Fruit: CFs were used from Ryan et al (2005). In addition, it was assumed that a person spending 1 month at Nabarlek collects all fruit from the mine site.

Flesh and plant activity concentrations were calculated using the input parameters described above. Using the model diet from Table 1, radionuclide activities (Bq) ingested per year per adult not accessing the Nabarlek site were estimated (Table 2). Radionuclide activities (Bq) ingested per year per adult accessing the site for one month are compiled in Table 3. It was assumed that children would consume about half the amounts eaten by an adult. The results for chilren are not shown here.

Using ingestion dose coefficients given in ICRP72 (1996) and the estimated annual consumption, total annual doses received via the ingestion of various bush foods for adults (and children) have been calculated and are shown in Figure 1.

Table 2 Ingested activities (Bq/y) from various bush foods and total annual ingestion dose in mSv for a person from Western Arnhem Land not accesing the Nabarlek site, using the model diet above

	^{226}Ra	^{210}Pb	^{210}Po	^{238}U	^{234}U	^{230}Th	^{228}Ra	Dose
Buffalo	1.5	2.0	61.8	0.4	0.5	0.2	0.5	0.08
Mussels	126.2	13.7	181.9	18.4	18.4	18.4	77.3	0.32
Pig	0.7	0.4	82.2	0.2	0.1	0.2	0.2	0.10
Magpie Goose	0.5	0.4	10.2	0.2	0.1	0.1	0.2	0.01
Fish	21.0	3.9	30.7	2.4	3.3	0.4	8.4	0.06
Wallaby	0.2	0.2	51.4	0.1	0.1	0.1	0.1	0.06
Yams	44.9	9.1	9.2	1.4	1.4	4.0	14.3	0.04
Turtle	1.3	1.8	46.3	0.2	0.2	0.1	0.5	0.07
Fruit	0.8	0.7	0.7	0.2	0.2	0.2	0.7	0.00
Filesnake	2.8	1.3	1.8	0.3	0.3	0.8	0.9	0.00
Crocodile	0.1	0.0	0.7	0.0	0.0	0.1	0.0	0.00
F/W shrimp	0.5	0.1	7.2	0.0	0.0	0.0	0.2	0.01
Water	1.3	0.2	7.2	0.4	0.6	0.5	0.3	0.01

Table 3 Ingested activities (Bq/y) from various bush foods and total annual ingestion dose in mSv/y for a person from Western Arnhem Land accessing the Nabarlek site for 1 month, using the model diet above

	^{226}Ra	^{210}Pb	^{210}Po	^{238}U	^{234}U	^{230}Th	^{228}Ra	Dose
Buffalo	1.6	2.1	65.0	0.4	0.5	0.2	0.5	0.08
Mussels	150.4	14.9	181.9	18.4	18.4	18.4	92.0	0.34
Pig	0.7	0.4	82.4	0.2	0.1	0.2	0.2	0.10
Magpie goose	0.5	0.4	10.2	0.2	0.1	0.1	0.2	0.01
Fish	26.2	4.9	38.2	3.1	4.1	0.6	10.5	0.07
Wallaby	0.3	0.3	77.9	0.2	0.2	0.2	0.1	0.09
Yams	68.2	13.8	13.9	2.1	2.1	6.0	21.6	0.06
Turtle	1.6	2.2	57.7	0.2	0.3	0.1	0.6	0.08
Fruit	0.8	0.7	0.7	0.2	0.2	0.2	0.7	0.00
Filesnake	4.3	2.0	2.7	0.4	0.4	1.2	1.4	0.01
Crocodile	0.1	0.0	0.9	0.0	0.0	0.1	0.1	0.00
F/W shrimp	0.6	0.1	9.0	0.0	0.0	0.0	0.2	0.01
Water	1.6	0.3	9.0	0.6	0.7	0.6	0.4	0.01

It can be seen from Tables 2 and 3 and Figure 1 that in terms of terrestrial food items, wallaby and yams are the biggest contributors to above environmental background ingestion doses. This is a worst case scenario, as it was assumed that during one months access only yams growing on-site were eaten, and that wallaby that are hunted in that month spent most of the year grazing on the actual fenced Nabarlek area. Mussels also contribute to above background doses however the mussel data set for the Nabarlek area is small, and actual doses are poorly described by the data set. Doses from the ingestion of fish are also above background, however, more surface water data are needed to perform a robust upstream-downstream comparison for aquatic food items as only little information on water radionuclide activity concentrations in Cooper Creek is available.

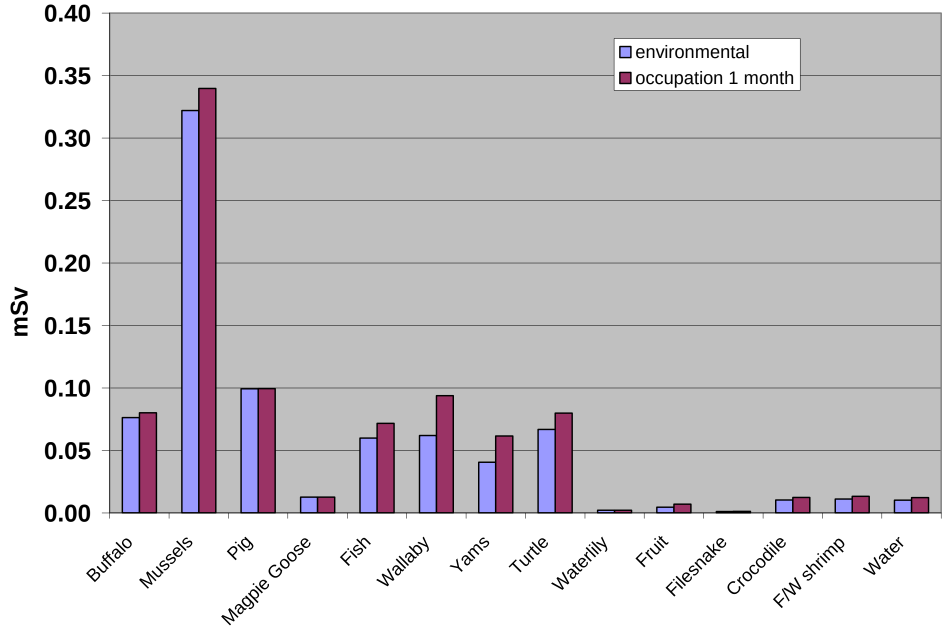

Figure 1 Typical background (environmental) annual doses from the consumption of various bush foods in Arnhem Land compared with doses received when harvesting bushtucker for one month from within the currrent fenced area at Nabarlek

In total the above background ingestion doses for one month's occupancy of the site would amount to approximately 0.1 mSv per annum.

Outstanding tasks

A robust upstream-downstream surface water dataset is needed to enable a true upstream downstream comparison for aquatic food items. Preliminary data, based on dry season sampling, indicate that downstream activity concentrations for radium are higher in Cooper Creek than upstream. However, more samples need to be collected from Cooper Creek, in collaboration with UEL, over the coming wet season to determine the median difference between upstream and downstream activity concentrations. In addition, mussels will be collected opportunistically to extend our data set and improve the accuracy of the contribution of this food item to the dose assessment.

Acknowledgments

Sharon Paulka and Jonas Klein from UEL are greatfully acknowledged for collecting surface water samples from Cooper Creek.

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Part 5: General Alligator Rivers Region

Undertake an ecological risk assessment of Magela floodplain to differentiate mining and non-mining impacts

J Boyden, A Petty[1], C Lehman[1] & P Bayliss[2]

Introduction

The Ecological Risk Assessment project is the final project of the 'Landscape-scale analysis of impacts' Program established in 2002 to help differentiate mining and non-mining impacts on the World Heritage and Ramsar listed Magela Creek wetlands downstream of the Ranger uranium mine. Ecological risk assessment allows the level of risk to the 'health' of ecosystems exposed to multiple stressors to be quantified in a coherent, robust and transparent manner. A high protection level for the biodiversity of aquatic ecosystems was used as the assessment endpoint, so conclusions here can be regarded as being appropriately conservative.

Two key results from the ecological risk assessment of the Magela floodplain reported at ARRTC 20 (October 2007) were that: (i) the overall findings of the landscape ecological risk assessment to date suggest strongly that non-mining landscape-scale risks to Magela floodplain should now receive the same level of scrutiny as that applied to uranium mining risks, including an assessment of what appropriate level of investment would be needed to manage these risks; and that (ii) of the landscape-scale ecological risks, damage from para grass > feral pigs > unmanaged fire. In this context it should be noted the current difference between non-mining and potential mining-related risk pathways may reduce when on-site water management systems at Ranger change in the transition between mine production and mine closure and rehabilitation. The transition between operations and closure requires a detailed and explicit risk assessment in itself.

Update: Fire and weed disturbance as landscape factors influencing vegetation development: implications for minesite revegetation

A key knowledge need identified by the Alligator Rivers Region Technical Committee (ARRTC) for rehabilitation of the final post-mining landform at Ranger is an understanding of the relationships between native vegetation communities and the physical environment. This knowledge is essential to identify revegetation management goals that optimise establishment of a sustainable vegetation community for rehabilitation of the site.

In this context, disturbance regimes (such as fire) are a key component of the physical environment affecting the development and composition of vegetation communities. Two landscape scale projects were undertaken during 2007–08 within Kakadu National Park (KNP) (including the Ranger Project Area) to: (i) assess the influence of fire on native vegetation

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communities; and (ii) to assess which environmental factors, including fire history, are significantly correlated with the distribution of two major grassy weeds, annual and perennial mission grass (*Pennisetum* spp). These two weed species are important because not only can they compete with native grasses and groundcover species but they can also increase fuel loads in areas in which they become established and hence increase the risk of high intensity fires.

The first study investigated the relationships between plant species diversity, ecological attributes of plants, and fire disturbance history (derived from Landsat satellite fire scar data) at 154 vegetation survey sites located through the Alligator Rivers Region (including the Ranger Project Area) and eastern Arnhem Land regions. The specific aims were to determine the relative importance of mean fire interval, late dry season fire frequency and early dry season fire frequency on the presence and absence of plant species. It was concluded that savannas are dynamic systems which are partially driven by variation in fire frequency and severity, thereby creating a mosaic of species that shift over time.

Sites with frequent fire, particularly late dry season (higher intensity) fire, were negatively related to species diversity. Furthermore, sites with longer mean intervals between burns were more likely to reflect a higher mix of species from different functional groups, such as ground cover species with a lifespan greater than 5 years. These findings are relevant to the management of the KNP region since changing disturbance regimes may alter the presence and diversity of different plant functional groups and species with differing life-histories. At the broader landscape scale, management incorporating a variety of fire regimes will promote biodiversity through the persistence of a savanna patch mosaic. The findings are also of relevance to minesite rehabilitation where an appropriate fire management strategy would be required during the early years following initial planting of vegetation. Exclusion of fire, especially higher intensity fire, for at least 5 years in the initial re-vegetation phase, will promote the establishment of ground and tree species that require fire-free intervals (of 5 to 10 years) to colonise and reproduce. Temporary fire exclusion may also aid in the establishment of deeper rooted perennial grasses that, compared to annual grass species, provide greater stability to soils and aid their biological development. After the vegetation establishment phase, fire frequency might be increased in line with the surrounding savanna woodlands in the Kakadu landscape. However, this would also need to assessed against other criteria used to measure the sucess of revegetation.

The second study used available spatial data on the distribution of annual and perennial mission grass in Kakadu and surrounds to assess the relationships with environmental factors such as fire history. Mission grass is commonly associated with disturbed areas, including minesites. As it is a potential threat to the successful rehabilitation of mine sites in the Alligator Rivers Region, it is important to understand the factors influencing the distribution of mission grass, as well as any potential ecological impact. A GIS approach has been used to discern broad scale factors that influence the distribution and fire behaviour of mission grass.

Results from the analysis indicate that mission grass can persist in a broad variety of vegetation communities, and that the strongest predictor of mission grass presence (apart from proximity to other patches of mission grass) is human activity (eg roads and settlements). This is likely due to a combination of the transportation of seed by vehicles, and increased disturbance near settled areas. Although cosmopolitan, the incidence of mission grass was reduced in areas of high overstory cover, prolonged seasonal inundation, or skeletal soils. Low lying, open habitats had the highest association with mission grass. Additionally a weak, yet significant, association was detected between fire and mission grass distribution in that lower fire frequency was associated with mission grass presence.

The findings support the current control regime, where eradication efforts focus on known colonies and new patches of mission grass as they appear, and controlling the spread of mission grass through human activity. Eradication and prevention of weed establishment is particularly important in areas of high local disturbance, such as the Ranger minesite and environs. Active weed control efforts, currently in place on the minesite, would in all likelihood be required to continue during the initial years following rehabilitation of the site, while the desired vegetation species are becoming established. This will also be important given the potential for the transport of seed from known weed colonies in the surrounding Park and on the mining lease.

Further investigation is warranted, particularly in regard to the fire management of mission grass affected areas. The more relevant questions regarding the interaction of fire and mission grass may be (i) what is the impact of mission grass on fire intensity rather than on frequency; and (ii) what is the impact of fire frequency on the spread of mission grass? Both of these would be best tested experimentally, although there may be some regions where the temporal record of mission grass distribution is sufficient to attempt such an analysis using the satellite record.

Definition of sediment sources and their effect on contemporary catchment erosion rates in the Alligator Rivers Region

MJ Saynor, G Staben, DR Moliere & JBC Lowry

Introduction

The 2006–07 wet season was the wettest on record for Jabiru. A total of 2600 mm of rain was recorded. Most of this (1940 mm) fell in February and March 2007, with 737 mm over a three-day period between February 28 and March 2 (Moliere et al 2007). The most intense period of rainfall occurred during the morning of 1 March 2007. The rainfall and the associated flood event was greatly in excess of a 1 in 100 year occurrence and had widespread impacts across the East Alligator River catchment in both the Magela Creek and also the East Alligator River. Along the gorge parts of the East Alligator River, vegetation was removed and large amounts of sediment were redistributed along the channel (images shown in Appendix 1). The removal of vegetation and movement of sediment along the East Alligator River in response to large flood events has not been studied in detail.

Fourteen landslips occurred in the upper part of the Magela catchment during this large event (Figure 1). An additional 10 to 15 landslips occurred within the catchment of the East Alligator River. A 3-dimensional stereoscopic investigation of 1982 aerial photographs images (most recent) at a scale of 1:50 000 scale of the area did not show evidence of scars of existing landslips suggesting that the landslips had all been generated during the large event in 2007.

The landslips occurred on well-vegetated weathered Oenpelli dolerite, which occurs as intrusions in the surrounding Mamadwerre sandstone (previously called Kombolgie sandstone). The landslips in the upper Magela catchment are of particular interest to SSD because they are a potential source of fine sediment to Magela Creek and hence their influence on baseline sediment loads in the creek need to be clearly understood. It is important to be able to distinguish sediment coming from this source from that originating from the minesite, in the event that higher than usual turbidity values are recorded downstream of the minesite.

Methods

The area of the landslips was initially mapped using remote sensing imagery. A two-day fieldtrip was undertaken in August 2007 to obtain detailed on-ground measurements of selected landslips. This groundtruthing was undertaken to provide sufficient calibration data to convert the rest of the landslip dimensions inferred from the remote sensing image to actual dimensions. During the field trip the opportunity was taken to conduct an aerial photographic survey of the East Alligator River to collect data for a pre- and post-flood comparision of bed and bank changes (Appendix 1).

On-ground measurements were made of the physical characteristics (including length, width, height of scarp face and slope angle) of several of the Magela Creek landslips. Samples of the slumped material (visually characterised by its bright red colour as a result of high iron oxide content) were collected for physical (particle size analysis and bulk density) and chemical analysis. Initial particle size analysis results showed that landslip material consists of predominantly fine-grained material with more than 60% of the particles being less than 63 µm in diameter. Material of this size range is easily eroded and transported overland by flowing water. Initial chemical analysis has shown that it contains very low concentrations of uranium and other heavy metals and hence it not likely to impact water quality parameters other than turbidity.

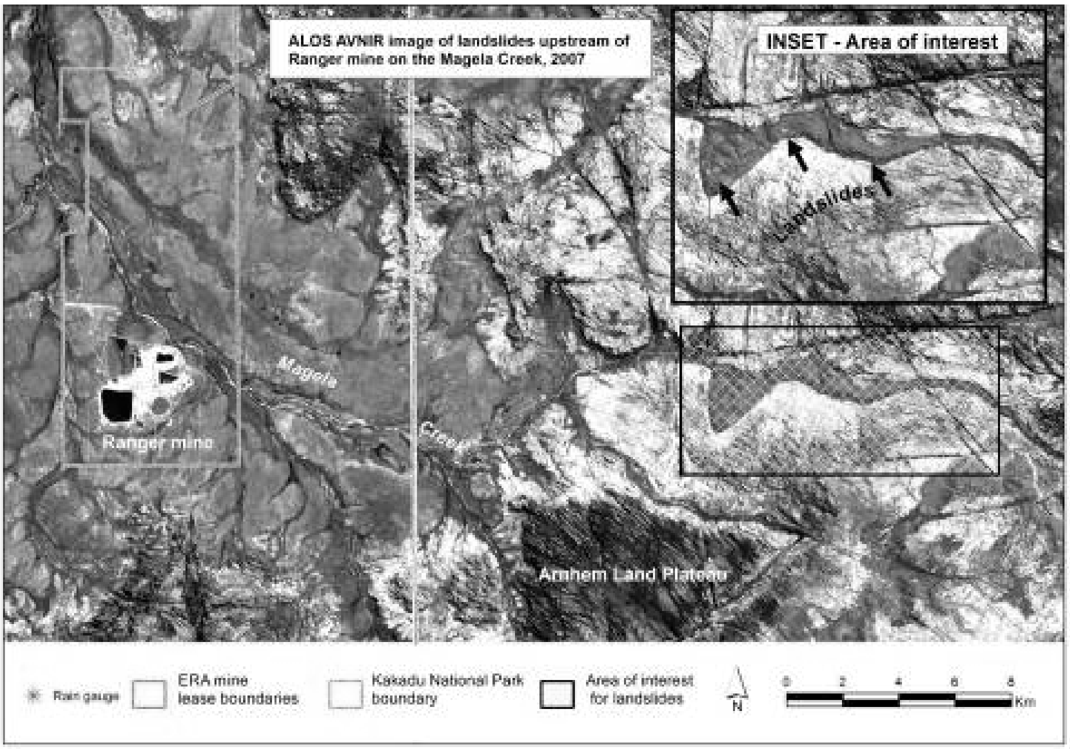

Figure 1 Location of the Landslips in the upper Magela Catchment in relation to Ranger mine

Results

The areas of the landslips measured on the ground are shown in Table 1. The slope angles ranged from 17° to 28° with an average of 21°. Areas derived from remotely sensed tended not to be able to distinguish between eroded material and deposited material and hence were approximately 1.3 larger than the field measured areas. This conversion factor was applied to all of the remotely identified landslips in the Upper Magela Catchment, to generate a total area of 0.3 km^2 (32 816 m^2) of sediment that was moved.

Although not all landslips were able to be measured in the field, the data that were acquired indicated an average depth of 2 m (Figure 2). Therefore the volume of the sediment that was moved during the landslip events was approximately 65 632 m^3. The average bulk density of the moved sediment was 1.19 t/m^3, which gives approximately 78 102 tonnes of sediment that was moved and potentially available for overland transport by future rainfall events.

Table 1 Landslip area comparisons generated by two different methodologies

Landslip no.	*Remotely sensed area (m²)*	*Measured areas (m²)*
1	5200	5630
2	530	250
3	840	250
8A	2720	1620
8B	7160	6420
9	16910	12220
Total	**33360**	**26390**

Figure 2 Field measurement of the one of the landslips in the Upper Magela Catchment. The approximate depth of the landslip at 2 m is illustrated.

Sediment dispersion in Magela Creek 2007–2008 wet season

During a routine field inspection of the SSD monitoring pontoons near the Ranger mine in January 2008 it was observed that the water was visibly red in colour (Figure 3). The source of this discoloration was traced by helicopter upstream to the vicinity of the landslips. With the landslips already being researched, and by tracing the source of the sediment back to these landslips, it was possible for timely and accurate information to be given to all stakeholders and other interested parties. In particular, the rapid conclusion that the source of the red water was not coming from Ranger mine provided assurance to local traditional owners that the source was natural.

During the 2007–08 wet season there were five occasions when the water in Magela Creek turned visibly red at the Magela Creek gauging station near Ranger mine (G8210009). In all cases when the river turned red there was a pronounced turbidity spike recorded at the Magela Creek pontoons. However, the water level did not exhibit the usual rise (Figure 4) that historically has been associated with higher turbidity levels in Magela Creek, indicating that the rainfall that triggered the events was localised in the vicinity of the landslips rather than being catchment wide. That the total area of the landslips is only 0.3 km^2, compared with the 605 km^2 catchment area of Magela Creek at the G8210009 monitoring point downstream of the mine, shows how much this highly erodible material can contribute to turbidity levels in Magela Creek. Indeed it is estimated that the landslips contributed approximately 50% of the total load of fine suspended sediment in Magela Creek for the 2007–08 wet season.

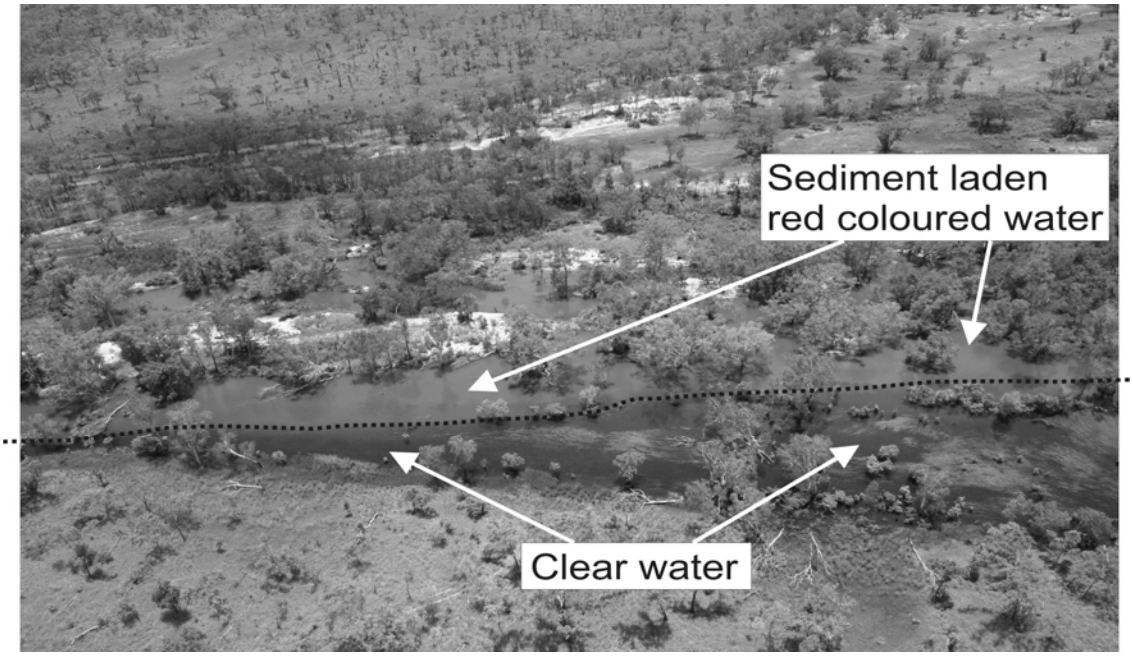

Figure 3 Red coloured sediment laden plume in Magela Creek downstream of G8210009 on 5 February 2008

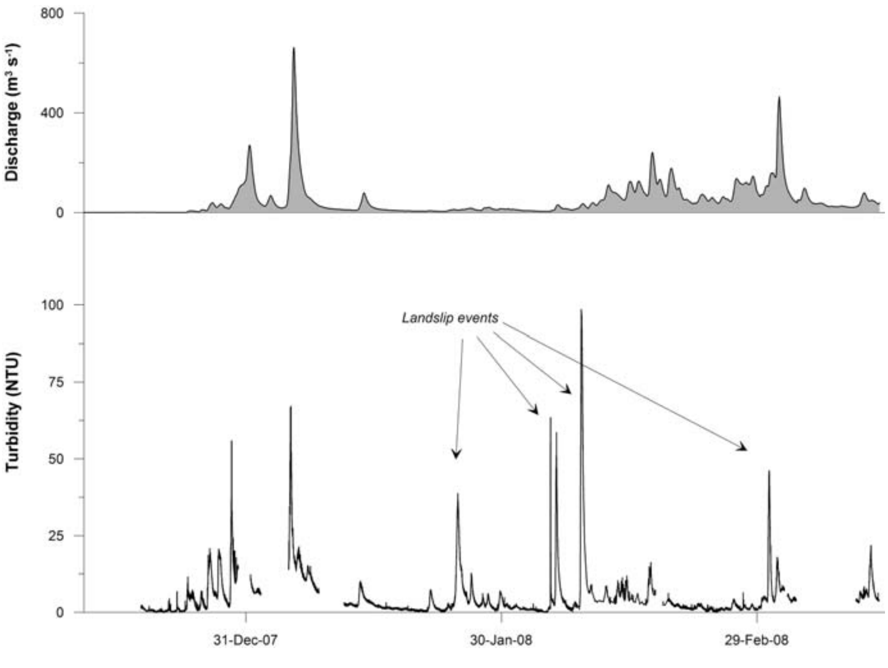

Figure 4 Wet season hydrograph for Magela Creek, showing high turbidity levels with low discharges

Rain gauge installation

An electronic rain gauge was installed on one of the hills above the landslips in late February 2008 to provide an early warning of the likely occurrence of turbidity pulses from the landslips. The rain gauge is able to be accessed by dial-in telemetry. The Bureau of Meteorology radar archive is also being investigated to see if it can be used to identify local weather systems that are likely to produce rain of sufficient intensity to mobilise the landslip material. The real time output of the rain gauge near the landslip may be able to be used to calibrate the radar images. If this technique proves to be successful, weather radar could possibly be used to obtain quantitative measurements of rainfall intensity generated by localised storm cells across the landscape.

Movement of sediment in the East Alligator River

During the large flood event large amounts of sediment were deposited in the river channel downstream of the lower gorge, approximately 1 km upstream of Cahills Crossing. In this context ***eriss*** was asked to comment on the deposition of sediment along the river section upstream of Cahill's Crossing which reduced the distance upstream that boat tour operators could negotiate (Figures 5 & 6). The advice provided was that this was the result of a natural process (ie a pulse of bedload moving through the catchment) and that it would clear in time (at least several wet seasons) as it had in the past. Dreging of the material was not recommended as a solution. It would be a long and inefficient processes as sediment would be distributed and re-deposited along the dredged sections.

Figure 5 East Alligator River upstream of Cahills Crossing in 2006 prior to the large movement of sediment. Points A, B, C are common points in both images.

Figure 6 East Alligator River upstream of Cahill's Crossing in 2007 after the large movement and deposition of sand during the flood

Further work

Further physical and chemical characterisation of the material from the landslips is underway to determine if there are any unique markers that can be used to distinguish landslip material from the fine sediment originating from the rest of the Magela catchment. The landslip sites also present an opportunity to track the natural re-establishment of vegetation. This is important in the context of reducing through time the contribution of the exposed landslip material to sediment load in Magela.

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Appendix 1

The following images illustrate changes that have taken place in the upper gorge section of the East Alligator River approximately 60 km upstream. The images were taken 8 months apart, before and after the flood event.

Upper section of the East Alligator River taken in July 2006

Same section of the East Alliator River as above, taken in April 2007 after the large rainfall event. River water level was similar at the time both of these photographs were taken. Note the removal of vegetation and the deposition of sand on both banks. The top of the right bank sand bar is estimated to be at least 10 m above baseflow conditions. Further upstream, on the left bank, bank scour is evident with removal of sand and vegetation. The level to which scour occurred is clearly seen.

Remediation of the remnants of past uranium mining activities in the South Alligator River Valley

A Bollhöfer, L Dunn[1], K Pfitzner, B Ryan, M Fawcett[2] & DR Jones

Introduction

In the 1950s and 60s the South Alligator River valley was prominent for its mineral and uranium mining activities. The main uranium deposits in the upper valley were characterised by high uranium contents with concentrations up to 2.5% U_3O_8. During peak activity, at least 16 different uranium ore bodies were discovered in the area, with 13 of them eventually mined (Waggit 2004). The South Alligator mill treated high-grade uranium ores from the Rockhole mine and the nearby O'Dwyers, Sterrets and Teague mines, which produced over 13 400 tons of very high grade uranium ore. This mill was located a few hundred metres east of Rockhole Mine Creek, near its confluence with the South Alligator River, and tailings were deposited on the ground with very little containment. When mining ceased in 1964 the minesites, mill and camps were abandoned and no substantial effort was made to rehabilitate or clean up the area.

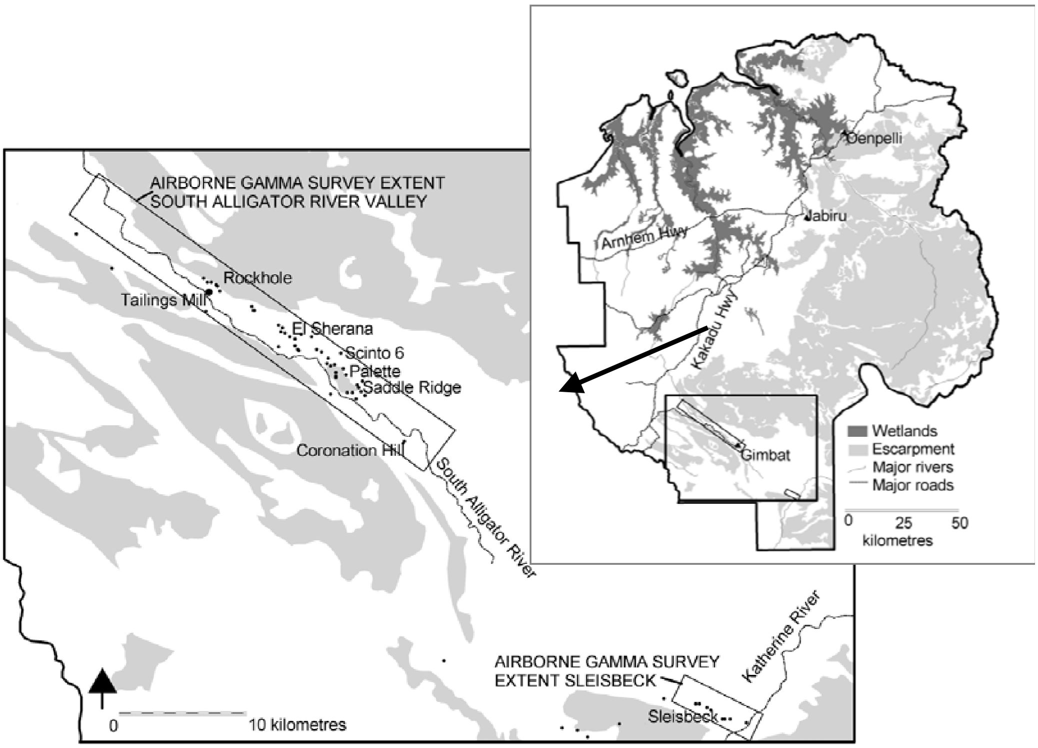

Figure 1 Alligator Rivers Region, with a detailed excerpt of the southern area showing the extent of two airborne gamma surveys conducted in 2000 and 2002, the location of known uranium anomalies (from MODAT database) and some historic mining and milling areas (Supervising Scientist 2003)

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Gamma surveys conducted by staff from SSD in 1984 confirmed the presence of radioactive tailings in the vicinity of the Rockhole mill. Most of the tailings present within the footprint of the original tailings deposition area were removed by Pacific Gold Mines NL between 1985 and 1986. In 1991, a routine survey revealed that residual tailings had become exposed in an erosion gully next to Gunlom Road directly opposite the old South Alligator mill tailings dam area. After the conclusion of this survey a program for rehabilitation of the abandoned mine sites was put into place and hazard reduction works were carried out in 1991 and 1992. During this period the South Alligator mill, surrounding buildings were demolished and the rubble and most of the remaining tailings waste were buried in trenches and covered with earth. Since completion of the hazard reduction works, routine bi-annual inspections of these sites are carried out by staff of SSD (Waggit 2003).

An airborne gamma survey was flown in 2000 to undertake precise mapping of the surface distribution of radiological material in the valley (Pfitzner & Martin 2000). The airborne gamma survey revealed that there was still some radiological contamination across the footprint of the original tailings area, extending to an area between the road and the South Alligator River (Pfitzner et al 2001). Groundtruthing was conducted in 2001. Dose estimate calculations made for three potentially affected classes of people – Park visitors, Rangers and local Aboriginal people – indicated that no further clean up action on radiological grounds alone (based on the most stringent International Atomic Energy Agency guidelines) was required (Bollhöfer et al 2002).

In 1996 land granted to the Gunlom Aboriginal Land Trust was leased to the Director of National Parks to be managed as part of Kakadu National Park. The lease agreement requires the Director to implement a plan of environmental rehabilitation of Guratba (Coronation Hill) and other mine sites and associated workings in the area so as to limit and where possible reverse the impact on the environment of any mining activities previously carried out. To that end, in May 2006, the Australian Government allocated $7.3 million over four years to the Director of National Parks.

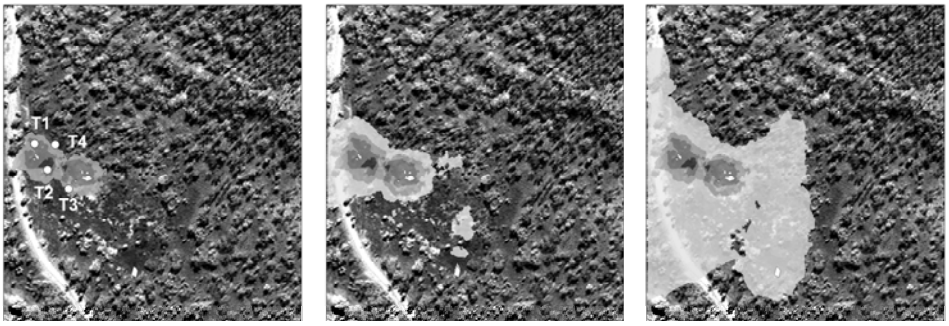
The rehabilitation program is being managed by Parks Australia, on behalf of the Director, and SSD is providing specialist assistance with the radiological assessment. The work is now well underway and includes: removal of residual physical hazards; remediation of non-radiologically contaminated areas; and cleanup of residual (radiological and non-radiological) mine-derived materials and ultimate burial of these latter materials in an engineered facility in the vicinity of the former El Sherana airstrip. Seasonal measurements of groundwater elevation, quality and baseline characterisation of radiological conditions of the soil profile are being done by SSD to provide input to the design process.

Results

Characterisation of contamination at the Rockhole residues site

A full characterisation of the tailings footprint area involved determining the areal extent of contamination, as well as the depth to which tailings-derived radiological material had penetrated into the soil profile beneath where the tailings were originally deposited. This survey was needed to quantify the volume of material to be removed to ensure effective remediation of the area, and hence to specifiy the design size of the engineered containment to hold it. In addition the data collected will provide the basis for deriving a radiological cleanup criterion, such that a clear distinction can be made between background and impacted material during the site excavation process.

A high resolution groundbased gamma survey was conducted in 2007 to determine the area of contamination at the former tailings dam footprint (Bollhöfer et al 2007). External gamma dose rate was measured at a spatial resolution of approximately 10 m using calibrated environmental dose rate meters. From the results, contour lines were produced which were overlaid on an Ikonos high resolution satellite image of the Rockhole residue area (Figure 2). An area between the road and the South Alligator River (to the top left edge of images in Figure 2) that contained some tailings derived material had been investigated in detail previously in 1999 (Tims et al 2000) and stabilised against erosion by covering it with rock. It is pending ultimate removal as part of the overall rehabilitation program.



1.2 µGy·hr^{-1} **0.9 µGy·hr^{-1}** **0.6 µGy·hr^{-1}**

Figure 2 Extent of the area at the original tailings dam footprint for various dose rate thresholds, and location of trenches 1 to 4 (T1–T4)

The gamma survey identified an area with elevated gamma dose rates of up to 3 micro grays per hour [µGy·hr^{-1}]. Several trenches were dug within this area with a backhoe to determine the depth of penetration of the radionuclides. Additional trenches were dug at the periphery of the area where dose rates had reduced to ~ 1 µGy·hr^{-1}. The walls of the trenches were sampled at a vertical resolution of 5 cm at the top 25–50 cm and at further 20 cm increments down to a total depth of 1–1.5 m.

The collected samples were measured for their activities of uranium and thorium series isotopes using high resolution gamma spectrometry in the ***eriss*** radioanalytical laboratories. The extent of penetration of radionuclides into the soil was found to be approximately 20 cm for Trenches 1–3 and approximately 1 m for Trench 4. The soil radionuclide activity concentrations (Bq·kg^{-1}) present in the respective layers were then converted to expected terrestrial gamma dose rates in air [µGy·hr^{-1}] using conversion factors recommended by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR 2000). Results are shown in Figure 3.

These data will be used to derive radiological cutoff criteria to guide the contractors who will be carrying out the remedial works on site. These same criteria will be applied to the cleanup of the thin layer of radiological tailings-derived material lying between the road and the South Alligator River.

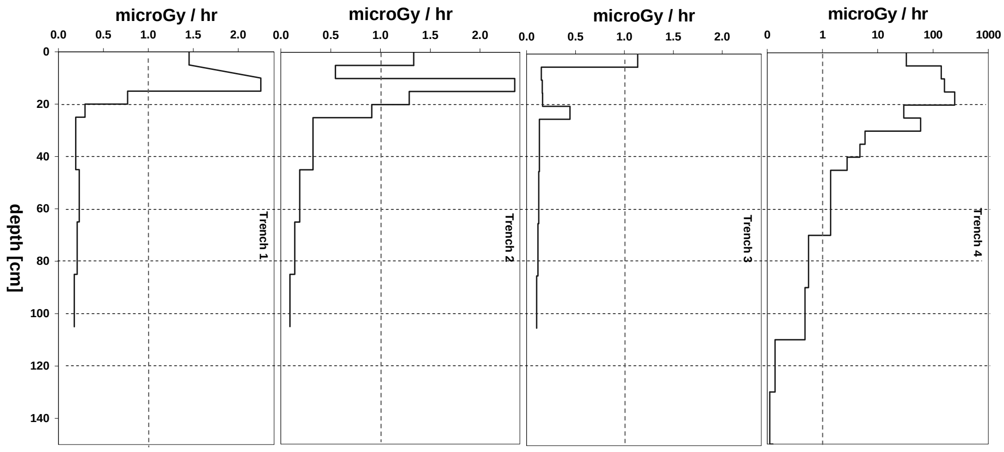


Figure 3 Terrestrial gamma dose rates calculated from radionuclide activity concentrations measured in trenches 1-4. The dashed vertical lines indicate a value of 1 µGy·hr^{-1}.

Assessment of the planned containment site

It is planned to build an appropriate capacity containment to bury the remnants of historic mining activities in the South Alligator River valley. Currently, a site located in the vicinity of the old El Sherana Airstrip close to the historic El Sherana mine is being assessed. A gamma radiation survey was conducted and Figure 4 shows that external gamma dose rates in the area average 0.13 µGy·h^{-1}, which is typical of the regional background including cosmic background.

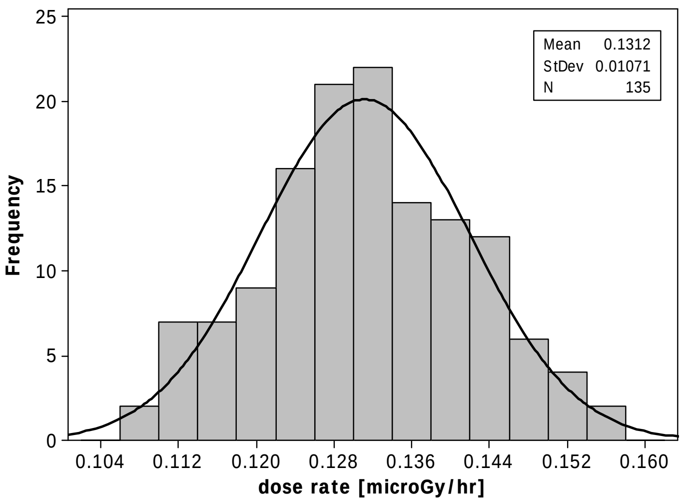


Figure 4 Histogram of γ-dose rates measured at the planned containment site at the El Sherana Airstrip

These measurements can be used to assess the performance of the containment after the placement of radioactive residues from the area and to demonstrate whether the containment meets the requirement to reduce the radiation levels at the surface to near background levels.

The final site selection for the containment also depends on a set of criteria related to potential for interaction with groundwater. In particular there should be sufficiently low hydraulic conductivity and the local water table should not rise to within 5 metres of the buried waste. ***eriss*** has been undertaking groundwater monitoring in the area of the proposed containment to assess seasonal variation in groundwater elevation and flow and to determine baseline groundwater quality. Figure 5 shows the location of the six groundwater monitoring bores that were installed in 2006. Continuous logging groundwater level monitoring sensors were placed in each bore in January 2007 and a permanent automated weather station was installed mid 2007.

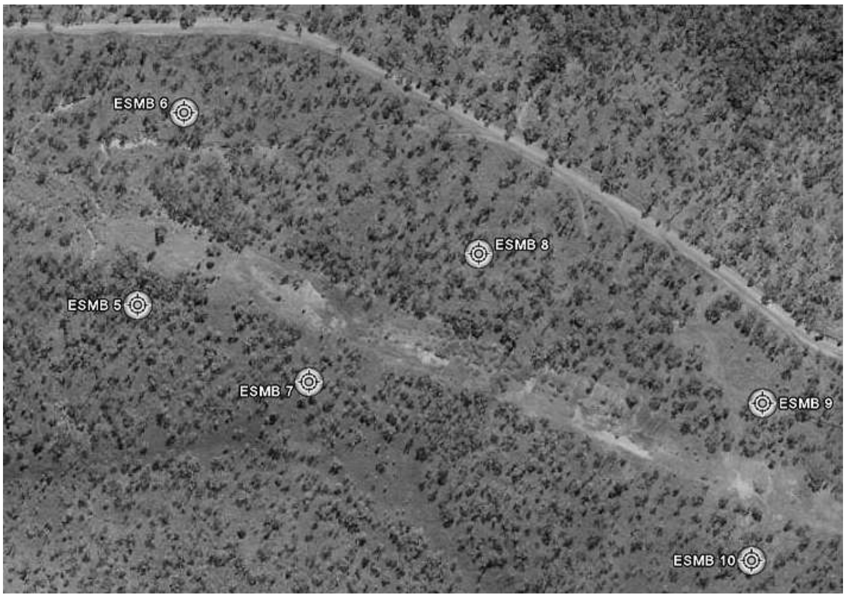


Figure 5 El Sherana airstrip groundwater monitoring bore locations

The depth of the bores range from 13.4 and 14.1 metres below ground surface for ESMB08 and ESMB09, respectively, to 20 metres for ESMB06 and ESMB07. Bores 8 and 9 have remained virtually dry, whereas the other four bores have water in them all year. The ground water levels range from 16 metres below the surface at the end of the wet season to more than 18 metres below the surface at the end of the dry season for ESMB06. This bore is situated at the furthermost upstream groundwater location and is at a higher elevation relative to the other bores. ESMB10 is the bore at the lowest relative elevation and at the furthermost groundwater downstream location. It exhibits groundwater levels of 7 metres below the surface at the end of the wet season and 12 metres below the surface at the end of the dry season. All bores show a slow and steady rise through the wet season and generally little or no response to individual storm events, with a slow lowering of groundwater levels during the almost rainless dry season. Further testing is under way to determine hydraulic conductivities in the area.

Acknowledgments

Thérèse Fox, Nadine Riethmuller and Rob Muller are thanked for support before and during the fieldwork.

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Part 6: Knowledge management and communication

Spatial and remote sensing data management review

J Lowry

Introduction

Over the last eight years, SSD has invested substantial resources in the acquisition and creation of spatial data. However, SSD has reached an important cross-roads, in terms of continuing to acquire, collect and crucially, manage spatial datasets. Unless an appropriate management system is put in place, the cost-effectiveness of acquiring, generating and maintaining spatial data will be reduced, as increasing resources will be required to store the data. At the same time, the value of data of unknown quality or of data lacking accurate documentation will steadily diminish. Conversely, the spatial datasets collated by SSD have the potential to significantly increase their value if dedicated resources were made available to support data management.

Consequently, the current status of data management protocols and the utilisation of network server space for handling of GIS and remote sensing data, and photographic images was reviewed internally over the period of February – June 2008. The review built on, and enhanced existing data management recommendations (eg Finlayson & Bayliss 1997) to develop new protocols and guidelines for spatial data management which reflected the current versions of software now in use, and current national and international data management standards and procedures.

The brief specifically called for the review project to:

- Collate the procedures and protocols developed by the Spatial User Group (SUG) for spatial and remote sensing data, to produce an SSD operations manual for the handling of these types of data;
- Oversee implementation of the partitioning of the Network Attached Storage (NAS) system to create a dedicated space for storage of spatial and remote sensing data;
- Oversee SSD-wide implementation of the new data management operations protocols by negotiation of data migration timelines with the primary data custodians;
- Establish a cataloguing and directory structure framework and place all primary acquired GIS and remote sensing datasets into the corporate server environment;
- Develop and implement a strategy for offline or slower hard-drive archiving of completed project material – ultimately this could be applied to all non-current (say greater than 3 y old) project material stored in SSDX;
- Develop and implement a strategy for cataloguing and storage of digital photographic images using as a basis the database developed by Andrew Esparon:
  - In the first instance implement a framework for storing of all future digital images on a common platform;
  - Develop a strategy for the progressive migration of historical images to the new platform;

- Identify current impediments to effective sharing and access of GIS and remote sensing data with ERIN (Environmental Resources Information Network) in Canberra, and identify hardware upgrades needed to address this; and
- Present a report to the Technical Data Management Group outlining the review findings and recommendations, and present a final consolidated report to the SMM for consideration.

Results

The key output from the project at the end of June was a data management plan, with a list of conclusions and short and long-term recommendations for the creation and management of spatial data. The report also identified and developed a number of standards and operating procedures for the creation and management of spatial data. These included the quality assessment and quality checking procedure for datasets as they are created and added to the corporate spatial database (Figure 1).

The cost and resource implications of the recommendations contained in the review are currently being considered by management within SSD, with certain elements allocated a higher priority. Key recommendations made in the report include:

- The need to recognise data management as an integral element of every project, and to ensure adequate resourcing is provided and recognised for this;
- Signficant resources will be needed to ensure staff involved in the creation, manipulation and analysis of spatial data have adequate training and time to undertake and implement the data management protocols and QA/QC procedures recommended in the report;
- The establishment of a dedicated data management position to oversee the management and storage of spatial data;
- If configured appropriately, existing hardware and software infrastructure could be used to implement an efficient data management environment which mirrors, on a smaller scale, the environments and standards developed by ERIN in Canberra. There are significant support benefits in ensuring the infrastructure and data management protocols are aligned or similar to those applied by ERIN.

Steps for completion

The technical recommendations contained in the report are in the process of being implemented.

The critical element to ensure the success of the data management strategy is ensuring that sufficient resources (training, time allocation) are provided to all staff involved in the creation, manipulation and management of spatial data.

The strategic staffing issue regarding coordination of data management in SDD is yet to be formally considered by the Senior Management Group.

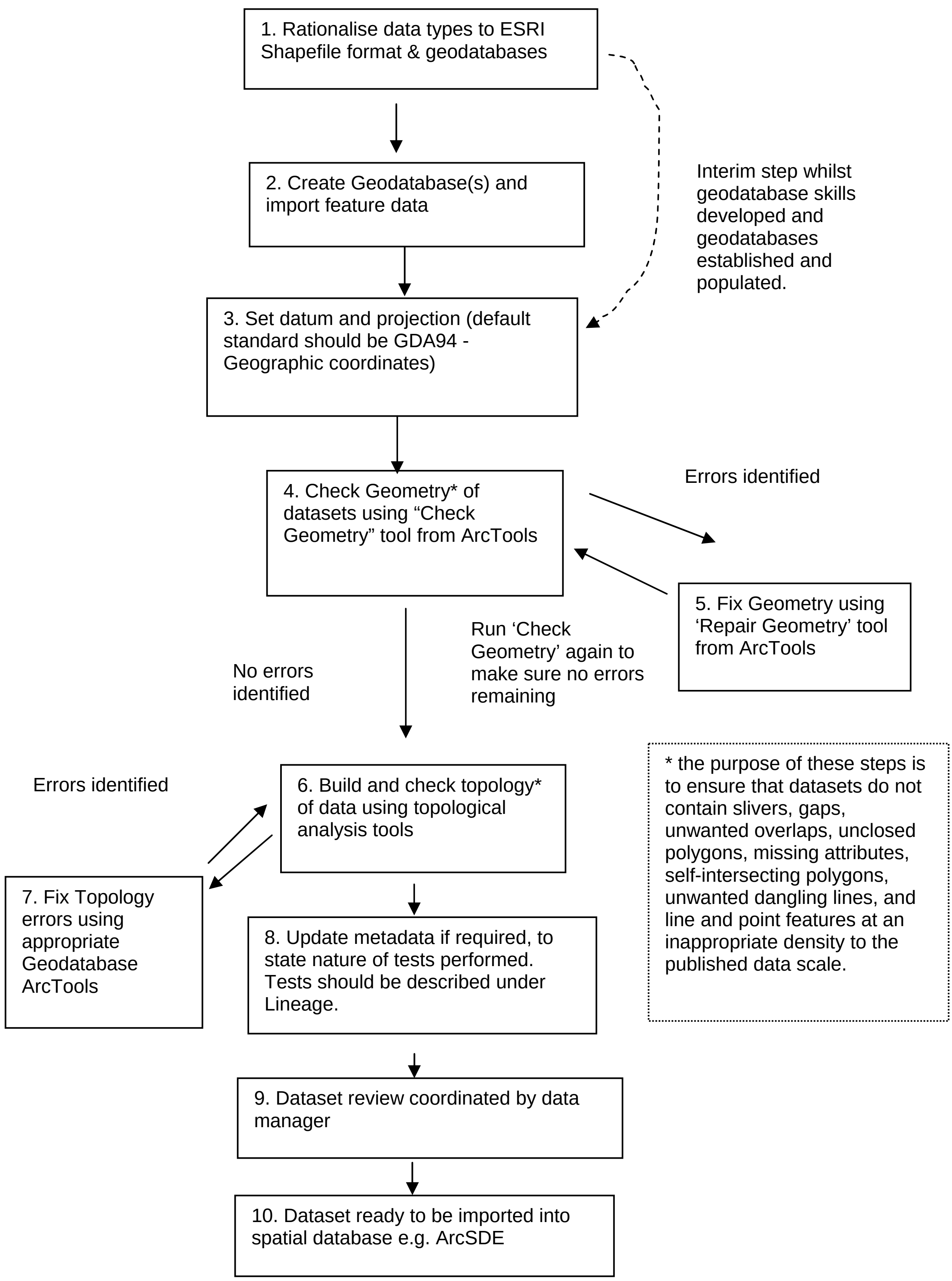

Figure 1 QA/ QC procedure for vector datasets

Acknowledgments

The spatial data review and related data management plan was produced through a consultative and iterative process with input provided by members of the ***eriss*** Spatial Users Group (SUG) and the ***eriss*** research and data management group (ERDMG). Valuable input has also been provided by Damian Woollcombe and Mike Maslen of ERIN, and Che Diggens of AMS Pty Ltd.

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Research consultancies

This section contains a summary of non-uranium mining related research consultancies carried out by ***eriss*** during 2007–2008. Most of these reports are commercial-in-confidence and are not available for public release.

List of non-uranium mining related research consultancies

Environmental Research Institute of the Supervising Scientist 2008. Understanding resources and risks in northern Australia: Establishment of a GIS and online mapping tool. Unpublished report to the Northern Australia Land and Water Taskforce by the Environmental Research Institute of the Supervising Scientist, January 2008.

Harford A & van Dam R 2007. Modification of toxicity testing protocols using a coral larva and the diatom *Nitzschia closterium* to assess marine contaminant issues from Alcan Gove operations – Interim Report, December 2007. Commercial-in-confidence report to to Alcan Gove Pty Ltd.

Harford A, Hogan A & van Dam R 2008. Ecotoxicological testing of receiving waters downstream from Newmont Woodcutter's mine site. Final Report, June 2008. Commercial-in-confidence report to Earth Water Life Sciences Pty Ltd.

Harford A, van Dam R, Hogan A, Tsang J, Parry D, Adams M, Stauber J & Negri A 2008. Modification of toxicity testing protocols using the diatom *Nitzschia closterium* and coral *Acropora tenuis* to assess the effects of waste water discharges from Rio Tinto Gove operations – Final Report, April 2008. Commercial-in-confidence report to Rio Tinto Alcan Gove Pty Ltd.

Humphrey C, Buckle D & Camilleri C 2009. A macroinvertebrate survey of stream sites associated with Territory Resources Frances Creek iron ore project, April 2008. Commercial-in-confidence report to Earth Water Life Sciences.

Humphrey C & van Dam R 2007. Environmental effects of magnesium sulfate-rich seepage waters from Argyle Diamond Mine, Report No. 6, Synthesis and Recommendations. Commercial-in-confidence report to Argyle Diamonds Pty Ltd.

Humphrey C, Fox G, Chandler L, Brazier J, Cammilleri C & Hanley J 2008. An assessment of the effects of mine waste waters arising from the Redbank copper mine on downstream macroinvertebrate community March 2008. Commercial-in-confidence report to Redbank Mines Ltd.

Humphrey C, Storey A, Buckle D, Chandler L, Hanley J, Creagh S & Camilleri C 2007. An assessment of the effects of seepage arising from the Argyle Diamond Mine upon stream biota sampled in 2006 and 2007: Summary results. Commercial-in-confidence report to Argyle Diamonds Pty Ltd.

Humphrey C, van Dam R, Storey A, Chandler L, Hogan A & Buckle D 2008. Assessment of the effects of $MgSO_4$-rich wastewater discharges from Argyle Diamond Mine on downstream aquatic ecosystems: Synthesis of a three-year (2006–08) study. Commercial-in-confidence report to Argyle Diamonds Pty Ltd.

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Tropical marine toxicity testing in Australia: a review and recommendations

RA van Dam, AJ Harford, MA Houston, AC Hogan & AP Negri[1]

Background

Developmental pressure across Australia's northern coastal catchments will increase rapidly in the near future. These areas are important strongholds for marine biodiversity and contain some of the least impacted marine habitats in the world. Consequently, such development must take place in an environmentally sustainable manner and needs to be underpinned by sound scientific knowledge. A study was undertaken in 2006 to 1) review the current state of the science for toxicity testing methods that have been developed for, or could be adapted to, Australian tropical marine species and environments and 2) use the information to identify the research and development needs to develop an appropriate suite of Australian tropical marine toxicity test methods. The component of the review focusing on water colum toxicity testing was accepted in 2008 for publication in a Special Issue of the *Australasian Journal of Ecotoxicology*, titled, *Tropical Ecotoxicology in Australasia*. A summary of the outcomes of this component of the review is provided below.

Outcomes

Sixteen taxonomic groupings (from 11 broad taxa groups) were reviewed and their suitability in routine toxicity testing protocols was assessed. The review revealed that there is a paucity of fully developed, regionally-relevant marine toxicity testing methods for Australian tropical marine species. Currently, just two fully developed routine sub-lethal/chronic test protocols exist, both of which are for tropical marine microalgae (*Nitzschia closterium* and *Isochrysis* aff. *galbana*), while sub-lethal tests using various tropical coral species have also been applied regularly. Numerous other Australian tropical marine species have been used for acute toxicity testing. In order to meet minimum requirements recommended by ANZECC/ARMCANZ (2000) for site-specific assessments, additional sub-lethal/chronic toxicity tests need to be developed. This review identified a number of different tropical marine species that may be suitable candidates in a suite of toxicity test protocols. The development of such methods will require a large R&D effort, and regulators, industries and community stakeholders should all have an interest in ensuring that these important knowledge gaps are addressed.

Acknowledgment

The review was funded by Rio Tinto Alcan – Gove operations, as part of the Marine Health Monitoring Program.

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Tropical rivers inventory and assessment project

R van Dam, R Bartolo & P Bayliss[1]

'Australia's tropical rivers – an integrated data assessment and analysis', more commonly known as the 'Tropical Rivers Inventory and Assessment Project' (TRIAP), was completed during 2008. Funded by Land & Water Australia and the Natural Heritage Trust 2, TRIAP was a collaborative effort between ***eriss***, James Cook University and the University of Western Australia, with additional involvement of Charles Darwin University and the University of Wageningen (The Netherlands). The project commenced in late 2004 and aimed to provide an information base to support the management of Australia's tropical rivers. It consisted of three sub-projects: (i) mapping and inventory, (ii) risk assessment of key threats, and (iii) development of a framework for evaluating ecosystem services.

The project examined 51 catchments across northern Australia (from Broome in the west to the western tip of Cape York), covering some 1 192 000 km^2. Three focus catchments, the Fitzroy River (Western Australia), Daly River (Northern Territory) and Flinders River (Queensland), were assessed in more detail.

Inventory and mapping of aquatic ecosystems

The key aim of sub-project 1 was to collate and analyse the existing spatial data for key biophysical attributes of the northern tropical rivers, namely, water quality, hydrology, geomorphology, estuaries, riparian vegetation, macroinvertebrates, fish, aquatic reptiles and waterbirds. A key conclusion following consolidation of the data was that there are substantial information/data gaps across the region.

A number of useful products have arisen from this sub-project, despite the limitations imposed by the lack of data. These included: a hydrologic classification based on streamflow and catchment characteristics; geomorphic classifications at two scales (whole of study area and catchment) based on landform and soil characteristics; a preliminary model for prediction of the distribution of riparian vegetation; and an in-depth discussion of the key water quality variables driving ecological function in tropical rivers and how they could be used as the basis for future classification schemes.

This study has provided the most comprehensive descriptions yet of a number of the key biophysical attributes of the northern tropical rivers, and also served to highlight where further research effort is required.

Major risks to aquatic ecosystems

Sub-project 2 demonstrated the utility of applying a tiered assessment approach for ecological risk assessment for tropical rivers. This involved gaining an initial broad understanding of the extent and status of the ecological assets and the threats they face, followed by formal semi-quantitative methods to assess and compare the significance of the threats, followed by detailed quantitative risk analyses of high priority threats to specific ecological assets. These assessments utilised and also built upon the information base compiled during sub-project 1,

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and were guided by information obtained from workshops and discussions with stakeholders. Various sensitivity and uncertainty analyses were also undertaken to test the rigour and robustness of the modelling approach, and identify areas for model improvement.

Highlights of this study included: the first comprehensive description of the key ecological assets of, and threats to, the northern tropical rivers; the advantages of adopting spatially explicit risk modelling to prioritise catchments in terms of their relative risk of multiple threats or pressures to multiple ecological assets; and the utility of Bayesian approaches (ie. a type of quantitative risk modelling that can integrate quantitative information with qualitative expert knowledge) in quantitative risk assessments.

Focusing in on one aspect of the study, the spatially explicit relative risk modelling across the whole study area identified eight catchments as being of higher relative risk of impacts from multiple threats/pressures. These were the Adelaide, Finniss/Darwin, Daly and Mary Rivers in the NT, and the Mitchell, Gilbert and Leichhardt Rivers in Qld. No catchments in WA were identified as being at higher relative risk. However, this result may have been due to issues associated with spatial data resolution and quality for this region.

The five most significant current threats to the ecological assets of the region's aquatic ecosystems were found to be (from a total of 19 included in the model): cattle grazing (on natural vegetation); feral pigs; poorly managed fire; aquatic weeds; and mining. Climate change and sea level rise are emerging problems for the region's aquatic ecosystems that threaten to cause much greater impacts than currently exist.

Trialling an ecosystem services framework

Sub-project 3 was undertaken by six Masters students from the University of Wageningen in the Netherlands. It was a small project that focused on two regions within the study area – the lower Mary River (NT) and the Daly River (NT). Through stakeholder interviews and the use of existing information, a framework for analysing the value of ecosystem services provided by the wetland and riverine ecosystems was populated and trialled. The analyses drew heavily on the conceptual framework developed for the Millennium Ecosystem Assessment (MA; www.millenniumassessment.org) where ecosystem services are defined as 'the benefits people obtain from ecosystems'.

It was established that local communities and other stakeholders are highly dependent on the wetlands in many ways, including ecological (eg wetlands of national importance, rare and endemic species), socio-cultural (eg cultural heritage, spiritual and existence values, sense of place, recreation) and economic reasons (eg water use, agriculture, carbon sequestration, tourism). The ecosystem services framework also considered the trade-offs and competing interests between these values in the context of the current policy setting, and evaluated the associated management implications.

Reports and information

TRIAP's comprehensive final reports can be accessed via the TRIAP page on the web site of the Supervising Scientist Division (www.environment.gov.au/ssd/tropical-rivers). In addition, a DVD package of the project's GIS coverages has been produced for use for future projects on northern tropical rivers.

Appendix 1 SSD publications and presentations for 2007–08

Journal papers (in press or published)

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

Frostick A, Bollhöfer A, Parry D, Munksgaard N & Evans K 2008. Radioactive and radiogenic isotopes in sediments from Cooper Creek, Western Arnhem Land. *Journal of Environmental Radioactivity* 99, 468–482.

Hancock GR, Loughran RJ, Evans KG & Balog RM 2008. Estimation of soil erosion using field and modelling approaches in an undisturbed Arnhem Land catchment, Northern Territory, Australia. *Geographical Research* 46(3), 333–349.

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Conference papers – published

Bartolo R 2008. Women in remote sensing and photogrammetry – results of the SSI women in spatial survey. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Bartolo R & Bayliss P 2007. Considerations in the design of a remote sensing framework for monitoring tropical coastal wetlands. In *Proceedings of the 28th Asian Remote Sensing Conference*, 12–16 November 2007, Putra World Trade Centre, Kuala Lumpur, Malaysia. Published online: Section TS13: Coastal Zone/Oceanography(2): http://www.aars-acrs.org/acrs/proceedings2007.php

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Appendix 2 ARRTC membership and functions

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

ARRTC membership

ARRTC comprises:

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

ARRTC functions

The primary functions of ARRTC are:

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

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

c to make recommendations to the Minister [for the Minister for the Environment, Heritage and the Arts] on:

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

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

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

Appendix 3 ARRTC Key Knowledge Needs

As a result of the extension in mine life and the conduct of a further three years of research since the original key knowledge needs (KKNs) and timeline priorities were established, it was judged by the Alligator Rivers Region Technical Committee (ARRTC) that a revision of the KKNs should be conducted in 2007–08. This was done and a revised list of KKNs approved by the committee. Both the original (2004–2006) KKNs and the new (2008–2010) KKNs are listed in this appendix so that the updates that have been made can be clearly identified. Appendix 3A contains the 2004–2006 KKNs; Appendix 3B contains the 2008–2010 KKNs.

Appendix 3A Alligator Rivers Region Technical Committee Key Knowledge Needs 2004–2006: Uranium mining in the Alligator Rivers Region

Overall objective

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

Background

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

The assumptions made for uranium mining operations in the region are:

- Mining of uranium at Ranger is expected to cease in about 2008. This will be followed by milling until about 2011 [revised to 2014 during 05/06] and final rehabilitation expected to be completed by about 2016 [revised to 2019 during 05/06].
- Nabarlek is decommissioned but has not reached a status where the NT Government will agree to issue a Revegetation Certificate to the mine operator. Assessment of the success of rehabilitation at Nabarlek is ongoing and is being used as an analogue for rehabilitation at Ranger.
- Jabiluka will remain in a care and maintenance condition for some years, at least until mining ceases at Ranger.
- It is unlikely that any proposal will be brought forward for mining at Koongarra in the foreseeable future.

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

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

Specifically, under the Ranger Environmental Requirements (ERs):

The company must ensure that operations at Ranger are undertaken in such a way as to be consistent with the following primary environmental objectives:

(a) maintain the attributes for which Kakadu National Park was inscribed on the World Heritage list;

(b) maintain the ecosystem health of the wetlands listed under the Ramsar Convention on Wetlands (ie the wetlands within Stages I and II of Kakadu National Park);

(c) protect the health of Aboriginals and other members of the regional community; and

(d) maintain the natural biological diversity of aquatic and terrestrial ecosystems of the Alligator Rivers Region, including ecological processes.

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

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

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

(a) revegetation of the disturbed sites of the Ranger project area using local native plant species similar in density and abundance to those existing in adjacent areas of Kakadu National Park, to form an ecosystem the long-term viability of which would not require a maintenance regime significantly different from that appropriate to adjacent areas of the park;

(b) stable radiological conditions on areas impacted by mining so that, the health risk to members of the public, including traditional Aboriginal owners, is as low as reasonably achievable; members of the public do not receive a radiation dose which exceeds applicable limits recommended by the most recently published and relevant Australian standards, codes of practice, and guidelines; and there is a minimum of restrictions on the use of the area;

(c) erosion characteristics which, as far as can reasonably be achieved, do not vary significantly from those of comparable landforms in surrounding undisturbed areas.

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

- Ranger – current operations
- Ranger – rehabilitation
- Jabiluka
- Nabarlek
- General Alligator Rivers Region
- Knowledge management and communication.

1 Ranger – current operations

ARRTC believes that the knowledge (research) needs relating to the current management of the uranium mining operations in the ARR would be best organised within a risk management framework. Such a framework would permit the various risks to the ARR to be assessed using a consistent, quantitative methodology and to be placed in priority order. Risk management is built on the use of quantitative predictive models to link threats or stressors with potential adverse ecological effects.

eriss is undertaking some ecological risk assessment work, but we believe this needs to be upgraded and made the central focus of the research programme. Proposals for research should then be assessed in terms of how the knowledge generated will contribute to the management of risk from the mining operations.

1.1 Reassess existing threats

Surface water transport of radionuclides: Using existing data, assess the present and future risks of health problems to the Aboriginal population eating bush tucker potentially contaminated by the mining operations bearing in mind that the current traditional Aboriginal owners derive a significant proportion of their food from bush tucker.

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

1.2 Ongoing operational issues

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

Land irrigation: Investigations are required on shallow groundwaters in the land irrigation areas adjacent to Magela Creek as a diffuse source of contaminants. Contaminants of interest/concern in addition to radionuclides are magnesium, sulfate and manganese. Further, the status of the irrigation areas in relation to decommissioning requirements (including radiological risk) needs to be assessed. Water quality models will be linked to knowledge of ecological effects.

Wetland filters: The key research issue associated with wetland filters in relation to ongoing operations is to determine whether their capacity to remove metals (principally uranium) from the water column will continue to meet the needs of the water management system in order to ensure protection of the downstream environment. Related to this is a reconciliation of the solute mass balance particularly for the Corridor Creek System.

Ecotoxicology: Although a great deal of ecotoxicological research and assessment has been undertaken, there are still a number of key issues that remain to be addressed including uranium toxicity measurements for two additional local native species, completion of research on the toxicity of magnesium including the ameliorative effects of calcium, and an assessment of the toxicity of manganese. Other issues that should be considered could include the relationship between dissolved organic matter and uranium toxicity and the effects of suspended sediment on aquatic biota.

Assurance programme for radionuclide surface water transport: Further research on surface water dispersion of radionuclides is not considered necessary on the basis of risk. However, a continuing programme of monitoring of radionuclides in surface water and in aquatic biota is considered necessary to provide assurance for Aboriginal people who source food items from the Magela Creek system downstream of Ranger.

Radiation exposure of workers: Further work should be considered in three areas: (a) a more robust examination of radon loss from dust particles, (b) development of a system which measures the concentration of radioactive dust and radon progeny in the breathing zone of a worker whilst wearing respiratory protection, and (c) measurement of the AMAD (activity Median Aerodynamic Diameter) and solubility of ore and product dusts in a range of exposure scenarios.

1.3 Monitoring

Surface water, groundwater, chemical, biological, sediment, radiological monitoring: Routine and project-based chemical, biological, radiological and sediment monitoring should continue. There is very little research required for the continued implementation of these programmes although there is scope for some specific research and analysis in relation to the review of the occupational radiological monitoring programme. More specifically, ARRTC supports the design and implementation of a new risk-based radiological monitoring programme based on a robust statistical analysis of the data collected over the life of Ranger.

2 Ranger – rehabilitation

Mining and milling at Ranger is likely to cease by about 2011 [revised to 2014 in 05/06]. Closure of the Ranger mine requires a large number of decisions, many of which will be dependent upon high quality scientific and technical information. The generation of this information will be the major focus of Ranger over the next five years. It will also be necessary to develop a holistic monitoring strategy, based on the risk assessments (and the associated models) recommended above, that aims to quantify changes in the identified high risk areas or test outcomes predicted by the models.

2.1 Landform design

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

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

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

Groundwater modelling: In addition to the seepage and runoff issues discussed above, there is a specific need to address the existence of mounds under the tailings dam and waste rock stockpiles. Models are needed to predict the behaviour of groundwater and solute transport in the vicinity of these mounds and options developed for their remediation to ensure that on-site revegetation can be achieved and that off-site solute transport from the mounds will meet environmental protection objectives.

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

Radiological characteristics of the final landform: The characteristics of the final landform from the radiological exposure perspective need to be determined and methods need to be developed to minimise radiation exposure to ensure that restrictions on access to the land are minimised. Radon emanation rates, gamma dose rates and radionuclide concentrations in dust need to be determined and models developed for both near-field and far-field exposure. The pre-mining radiological conditions should also be assessed so that estimates can be made of the likely change in exposure rates compared to pre-mining conditions.

Testing of 'trial' landforms: Current landforms at Ranger and at other sites such as Nabarlek should be used to test the various models and predictions for water quality, geomorphic behaviour and radiological characteristics at Ranger.

Final landform design: The detailed design for the final landform at Ranger should be determined taking into account the results of the above research programmes on surface and ground water, geomorphic modelling and radiological characteristics.

2.2 Ecosystem establishment

Development and agreement of closure criteria from ecosystem establishment perspective: Closure criteria for ecosystem establishment need to be established at both the broad scale and the specific. At the broad scale, agreement is needed, particularly with the traditional owners and within the context of the objectives for rehabilitation incorporated within the ERs, on the general strategy to be adopted on habitat types to be incorporated and the species composition of trees, shrubs and grasses to be established on the landform. At the specific scale, criteria are needed that will be used to assess the success of ecosystem establishment. These would include, for example, targets for species density and abundance and measures of faunal return.

Characterisation of terrestrial and aquatic ecosystem types at analogue sites: To implement the revegetation strategy for Ranger mine, an understanding of the relationships between vegetation communities and key geomorphic features (parent material, slope, effective soil depth, internal drainage characteristics) in surrounding areas of Kakadu National Park is essential in identifying sustainable and achievable ‘landscape’ analogues (or target habitats) for the final, post-mine landform at Ranger. Identification and description of these landscape analogues is also the first step in developing robust, measurable, ecologically-based criteria for assessing revegetation performance, function and success.

Establishment and sustainability of ecosystems on mine landform: Research on how the landform, vegetation, fauna habitat, hydrology and geochemistry will be reconstructed at Ranger is essential. Noting that there are no good examples in the wet-dry tropics of successful reclamation of hard rock mines, priority needs to be given to this research. Research sites should be established that demonstrate an ability to reconstruct an ecosystem, even if this is at a relatively small scale. Issues that need to be addressed include species selection, seed collection germination and storage, propagation of recalcitrant species, nursery production of seedlings, fertiliser strategies including application methods and direct seeding techniques. Other issues requiring investigation include the return of fauna habitat, potential plant toxicity problems from waste rock, the exclusion of weeds and the effects of fire, hydrology and erosion on the rehabilitation strategy.

Radiation exposure pathways associated with ecosystem re-establishment: Bioaccumulation studies conducted to date have focused on aquatic animal and plant species because of their importance of the aquatic transport pathway, particularly during the operational phase of uranium mining operations. Information on radionuclide uptake by terrestrial animals and plants is required to enable a radiological risk assessment to be carried out for the revegetation programme. This needs to be coupled with estimates of terrestrial bushfood consumption by local Aboriginal people. Another radiological issue that requires assessment is the potential for tree roots to penetrate any radon barriers that form part of the rehabilitated landscape.

2.3 Groundwater dispersion

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

Geochemical characterisation of source terms: Investigations are needed to characterise the source term for transport of contaminants from the tailings mass in groundwater. These will include determination of the permeability of the tailings and its variation through the tailings mass, strategies and technologies to enhance settled density and accelerate consolidation of tailings, and pore water concentrations of key constituents. Assessment is also needed of the effectiveness (cost and environmental significance) of paste and cementation technologies for increasing tailings density and reducing the solubility of chemical constituents in tailings.

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

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

2.4 Water treatment

Active treatment technologies for specific mine waters: Substantial volumes of process water retained at Ranger in the tailings dam and Pit 1 must be disposed of by a combination of water treatment and evaporation during the mining and milling phases of the operation and during the rehabilitation phase. Research priorities include treatment technologies and enhanced evaporation technologies that can be implemented for very high salinity process water.

Passive treatment of waters from the rehabilitated landform: Sentinel wetlands may form part of the final landform at Ranger. Research on wetland filters during the operational phase of mining will provide information relevant to this issue. However, there is a need to assess the long-term behaviour of physical and biotic components of wetlands and the ecological health of wetlands which are used to treat runoff from the proposed rehabilitated landform.

2.5 Monitoring

A monitoring programme to assess the success of rehabilitation at Ranger will be essential. Prior to its design and implementation, clear and agreed closure criteria will be needed as indicated above. These criteria should be used to determine the design of the monitoring programme.

Monitoring of the rehabilitated landform: A new management and monitoring regime for the rehabilitated Ranger landform needs to be developed and implemented. It needs to address all relevant aspects of the rehabilitated landform including ground and surface water quality, radiological issues, erosion, flora, fauna, weeds, and fire.

Off-site monitoring during and following rehabilitation: A monitoring regime for the downstream environment is also required to assess rehabilitation success with respect to protection of the downstream environment. This programme should address the dispersion of contaminants by surface water, ground water and via the atmosphere.

3 Jabiluka

The Jabiluka project has now entered a long-term care and maintenance phase. It is ARRTC's view that ongoing monitoring will be required throughout this period. In addition, a review is needed of knowledge that would be required prior to any proposal to develop Jabiluka. In particular, it will be necessary to identify and implement any projects considered essential in providing this knowledge well in advance of any development plans.

3.1 Monitoring

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

3.2 Research

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

4 Nabarlek

Nabarlek is decommissioned but has not reached a status where the NT Government will agree to issue a Revegetation Certificate to the mine operator. Since Nabarlek is the first Australian uranium mine of the modern era to complete operations and be rehabilitated, ARRTC believes that Australia needs to ensure that an overall assessment of the success of rehabilitation at Nabarlek is carried out. The Nabarlek site should also be used as an analogue for rehabilitation at Ranger and projects at Nabarlek should be designed to address specific issues of concern at Ranger.

4.1 Success of revegetation

Revegetation assessment: The principal ongoing issue at Nabarlek is the poor revegetation. Assessment of the adequacy of revegetation at the site should continue and, following its completion, management options should be developed and submitted to the mine-site technical committee for its consideration.

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

4.2 Assessment of radiological, chemical and geomorphic success of rehabilitation

Overall assessment of rehabilitation success at Nabarlek: The current programme on erosion, surface water chemistry, groundwater chemistry and radiological issues should be continued to the extent required to carry out an overall assessment of the success of rehabilitation at

Nabarlek. In particular, all radiological exposure pathways should be evaluated and a comprehensive radiation dose model for Nabarlek should be developed.

5 General Alligator Rivers Region

5.1 Landscape scale analysis of impact

Apart from regular refinement of procedures for the current monitoring programmes, a potential major future research area is the possible development of broader, landscape scale programmes that would enable possible effects of mining to be distinguished from those arising from other causes. Such a programme was recommended by the Independent Science Panel of the World Heritage Committee. Initial studies have been undertaken. However, ARRTC believes that, before committing further resources to this programme, a review of the programme to assist in determining future priorities needs to be undertaken.

Re-assess and prioritise the landscape programme: A review is required, within a modelling conceptual and risk assessment framework, of the landscape wide programme to determine options and priorities for the future development of this programme.

5.2 South Alligator River valley rehabilitation

The focus of work to develop and implement a rehabilitation strategy for historic uranium mining related sites in the South Alligator Valley is the identification of a suitable site for the burial of radiologically active mining residues such as uranium ores or sediments contaminated with tailings. Parks Australia is responsible for this programme. Once potential sites have been identified based upon hydrology, access, stability, cultural and other considerations, groundwater investigations will be required to ensure that the site meets requirements for minimum separation between the base of the repository and top of the water table.

Assessment of mine sites in the South Alligator River valley: SSD conducts regular assessments of the status of mine sites in the SAR valley, provides advice to Parks Australia on technical issues associated with its rehabilitation programme and occasionally conducts a low level radiological monitoring programme, primarily for assurance purposes. ARRTC believes these should continue.

5.3 Develop monitoring programme related to West Arnhem Land exploration activities

Mining exploration is proceeding in the eastern area of the ARR in Arnhem Land outside the Kakadu National Park. In order to overcome the common problem of inadequate baseline data for correctly identifying the cause of environmental change, the SSD and NLC have jointly advocated the strategic collection of regional baseline information on aquatic ecosystems in areas adjacent to mining exploration sites in the ARR.

Baseline studies for biological assessment in West Arnhem Land: In areas adjacent to mining exploration sites, ARRTC believes there is a need to determine a baseline for (a) rare, threatened and endemic biota and (b) indicator species or groups such as macroinvertebrates.

5.4 Koongarra

There are currently no plans for the development of the Koongarra uranium prospect. However, it is ARRTC's view that, subject to the prioritisation of available resources, an ongoing base-line data collection programme could be established and the value of Koongarra as an analogue for pre-mining radiological conditions at Ranger could be investigated.

Baseline monitoring programme for Koongarra: A low level monitoring programme should be developed for Koongarra to provide baseline data in advance of any possible future development at the site. Data from this programme may also have some relevance as a control system for comparison to Ranger, Jabiluka and Nabarlek.

Analogue information for pre-mining conditions at Ranger: The value of Koongarra as an analogue site for pre-mining radiological conditions at Ranger should be investigated. There are some pre-mining radiological data for Ranger but the value of these data could be greatly enhanced if it could be extrapolated, through the use of an undisturbed analogue site such as Koongarra, to provide further information on parameters such as pre-mining gamma dose rates, radon exhalation, and radioactivity concentrations in dust.

6 Knowledge management and communication

The Alligator Rivers Region is one of the most studied regions in Australia. Consequently, a very large amount of knowledge has been accumulated over the years on this system. The stimulus for the research is that knowledge-based management of the uranium mines is the best approach to ensuring minimal risk to the ARR.

ARRTC believes that additional emphasis needs to be put on knowledge management and exchange in the next five years. Key aspects that will need to be addressed include the following.

6.1 Integrated framework

Development of an integrated framework: This has already commenced within a landscape analysis framework and is linked with the development of conceptual models of the ARR recommended above. Such an integrated framework will assist with the communication where the scientific information is relevant, and how it informs on the various risks to the system and its people from the uranium mines.

6.2 Uncertainty analysis

Uncertainty analysis of data and communication: People involved in the management of natural resources rarely have all the information they need. Even in the ARR, where a very large amount of research has been undertaken on the possible impacts of uranium mining, there is still much not known about the risks. ARRTC believes that management of the mining operations would be improved if the uncertainties in the risk assessment were explicitly identified and communicated. Additionally, those high risk areas where the uncertainty is great would be targeted for more research. It is expected that current work on the development of conceptual models of the ARR will clarify many of these uncertainties.

6.3 Effective communication channels between research providers

Establishing effective communication channels between and within research providers: There are a large number of organisations undertaking research in the ARR including SSD, EWLS, ERA, Parks Australia North and CSIRO. Given limited resources, it is critical that research is not being duplicated or previous studies repeated. ARRTC believes that communication between the various research providers could be improved and become more formalised to ensure better outcomes for all parties.

6.4 Effective communication to stakeholders

Effective communication of science to stakeholders: There are a large number of stakeholders with direct and indirect interests in uranium mining in the ARR. It is critical that the results of the high quality research being undertaken in the ARR is communicated to all stakeholders in the most relevant format. ARRTC believes that the various research providers need to target their communication strategies more specifically to the various stakeholder groups.

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

Overall objective

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

Background

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

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

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

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

Specifically, under the Ranger Environmental Requirements (ERs):

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

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

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

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

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

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

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

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

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

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

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

1 Ranger – Current operations

1.1 Reassess existing threats

1.1.1 Surface water transport of radionuclides

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

1.1.2 Atmospheric transport of radionuclides

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

1.2 Ongoing operational issues

1.2.1 Ecological risks via the surface water pathway

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

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

1.2.2 Land irrigation

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

1.2.3 Wetland filters

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

1.2.4 Ecotoxicology

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

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

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

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

1.2.5 Mass balances and annual load limits

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

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

1.3 Monitoring

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

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

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

2 Ranger – Rehabilitation

2.1 Reference state and baseline data

2.1.1 Defining the reference state and baseline data

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

2.2 Landform

2.2.1 Landform design

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

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

2.2.2 Development and agreement of closure criteria from the landform perspective

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

2.2.3 Water quality in seepage and runoff from the final landform

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

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

2.2.4 Geomorphic behaviour and evolution of the landscape

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

2.2.5 Radiological characteristics of the final landform

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

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

2.3 Groundwater dispersion

2.3.1 Containment of tailings and other mine wastes

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

2.3.2 Geochemical characterisation of source terms

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

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

2.3.3 Aquifer characterisation and whole-of-site model

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

2.3.4 Hydrological/hydrogeochemical modelling

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

2.4 Water treatment

2.4.1 Active treatment technologies for specific mine waters

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

2.4.2 Passive treatment of waters from the rehabilitated landform

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

2.5 Ecosystem establishment

2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective

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

2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites

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

2.5.3 Establishment and sustainability of ecosystems on mine landform

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

2.5.4 Radiation exposure pathways associated with ecosystem re-establishment

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

2.6 Monitoring

2.6.1 Monitoring of the rehabilitated landform

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

2.6.2 Off-site monitoring during and following rehabilitation

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

2.7 Risk assessment

2.7.1 Ecological risk assessments of the rehabilitation and post rehabilitation phases

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

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

2.8 Stewardship

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

3 Jabiluka

3.1 Monitoring

3.1.1 Monitoring during the care and maintenance phase

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

3.2 Research

3.2.1 Research required prior to any development

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

4 Nabarlek

4.1 Success of revegetation

4.1.1 Revegetation assessment

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

4.1.2 Development of revegetation monitoring method

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

4.2 Assessment of radiological, chemical and geomorphic success of rehabilitation

4.2.1 Overall assessment of rehabilitation success at Nabarlek

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

5 General Alligator Rivers Region

5.1 Landscape scale analysis of impact

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

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

5.2 South Alligator River valley rehabilitation

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

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

5.3 Develop monitoring program related to West Arnhem Land exploration activities

5.3.1 Baseline studies for biological assessment in West Arnhem Land

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

5.4 Koongarra

5.4.1 Baseline monitoring program for Koongarra

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