supervising scientist report **196**

eriss research summary 2006–2007

DR Jones, C Humphrey,
R van Dam & A Webb (eds)

Australian Government
Department of the Environment, Water, Heritage and the Arts
Supervising Scientist

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

This Supervising Scientist Report is a summary of the 2006–2007 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 and Water Resources. ***eriss*** provides specialist technical advice to the Supervising Scientist, and conducts research into:

- the impact of uranium mining on the environment and people of the Alligator Rivers Region, and
- on the protection and management of tropical rivers and their associated wetlands.

The objective of the mining-related research is to develop best practice methodologies for the monitoring of impact, for the development of management guidelines to protect the environment during the operational phase of mine life, and for the development of closure criteria for the decommissioning and rehabilitation phase of mine life.

eriss also applies its expertise to carrying out of a limited program of contract research on the impacts of mining elsewhere in the northern tropics.

A significant program of work is also conducted on northern tropical rivers, with the Tropical Rivers Inventory Assessment Project (TRIAP) being the current focus of this activity. This four-year research program started in 2004 and is funded by the Natural Heritage Trust II and Land and Water Australia. The project is managed by ***eriss*** with collaborators from the University of Western Australia and James Cook University of North Queensland.

This report documents research projects undertaken by ***eriss*** over the 2006–07 financial year. The final section contains a summary of the non-uranium mining related external projects. Commercial-in-confidence projects have been excluded from this compilation. For additional information, readers are referred to the annual publications list (Appendix 1) that details all of the material published, and conference and workshop papers presented by ***eriss*** staff in 2006–07.

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 by consultation between the Alligator Rivers Region Technical Committee (ARRTC – 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.

eriss contributes to the addressing of the KKNs by applying its expertise in the following science fields:

- Ecotoxicology
- Environmental radioactivity
- Hydrological and geomorphic processes
- Monitoring and ecosystem protection
- Biophysical pathways and ecological risk assessment
- Aquatic chemistry

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.

Communication between research providers is addressed by stakeholder presentations and interactions at ARRTC meetings, as well as by conduct of formal multi-stakeholder workshops, and presentations at conferences and workshops. Communication with local Indigenous people and other regional stakeholders is an integral part of the project approvals and reporting processes. A dedicated officer for Indigenous Communications is stationed at the Jabiru field station for this purpose.

The following assumptions provided the context for the planning and design of the 2006–07 research program:

- Mining of uranium at Ranger is expected to cease in about 2008. This will be followed by milling until about 2020 and rehabilitation is expected to be largely completed by about 2026.
- The Nabarlek site has not reached a condition 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 this site is being used as an analogue for future rehabilitation at Ranger.
- Jabiluka will remain in a care and maintenance condition for some years, at least until mining ceases at Ranger.
- SSD will provide ongoing assistance to Parks Australia on matters relating to the assessment and rehabilitation of the cluster of small abandoned historic uranium mines in the South Alligator River Valley.

A review of the KKN framework was initiated by ARRTC at its 18th meeting in October 2006, recognising that since the original set of KKNs was formulated in 2004, the operational life of the site has been extended from about 2010 to 2020. It is envisaged that this review, and approval of an updated set of KKNs, will be not be completed until the middle of the 2007-08 financial year. Until this time reporting of research project outcomes will continue to be against the KKNs listed in Appendix 3.

This report has been structured under five main headings, consistent with the KKN framework:

1 Ranger – Current Operations

2 Ranger – Rehabilitation

3 Jabiluka

4 Nabarlek

5 General Alligators Rivers Region

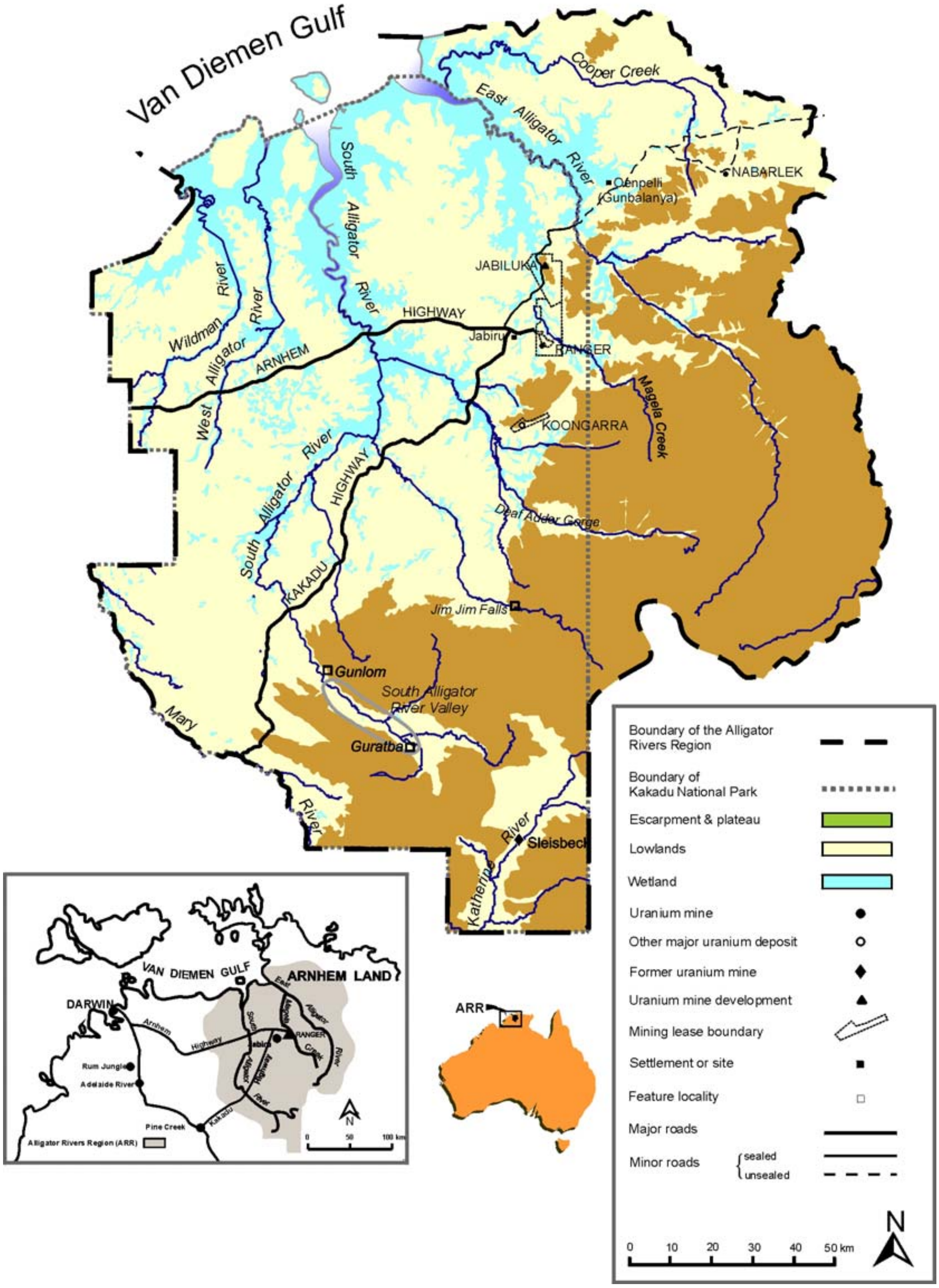
Three maps (following this Preface) provide the regional context for the locations that are referenced in the research papers. Map 1 shows Kakadu National Park and the locations of the Ranger Mine, Jabiluka project area, the decommissioned Nabarlek Mine, and the South Alligator River Valley. A schematic of the Ranger mine site is provided for reference in

Map 2. Map 3 shows the locations of billabongs and waterbodies used for the aquatic ecosystem monitoring and research programs for assessing impacts from the Ranger Mine.

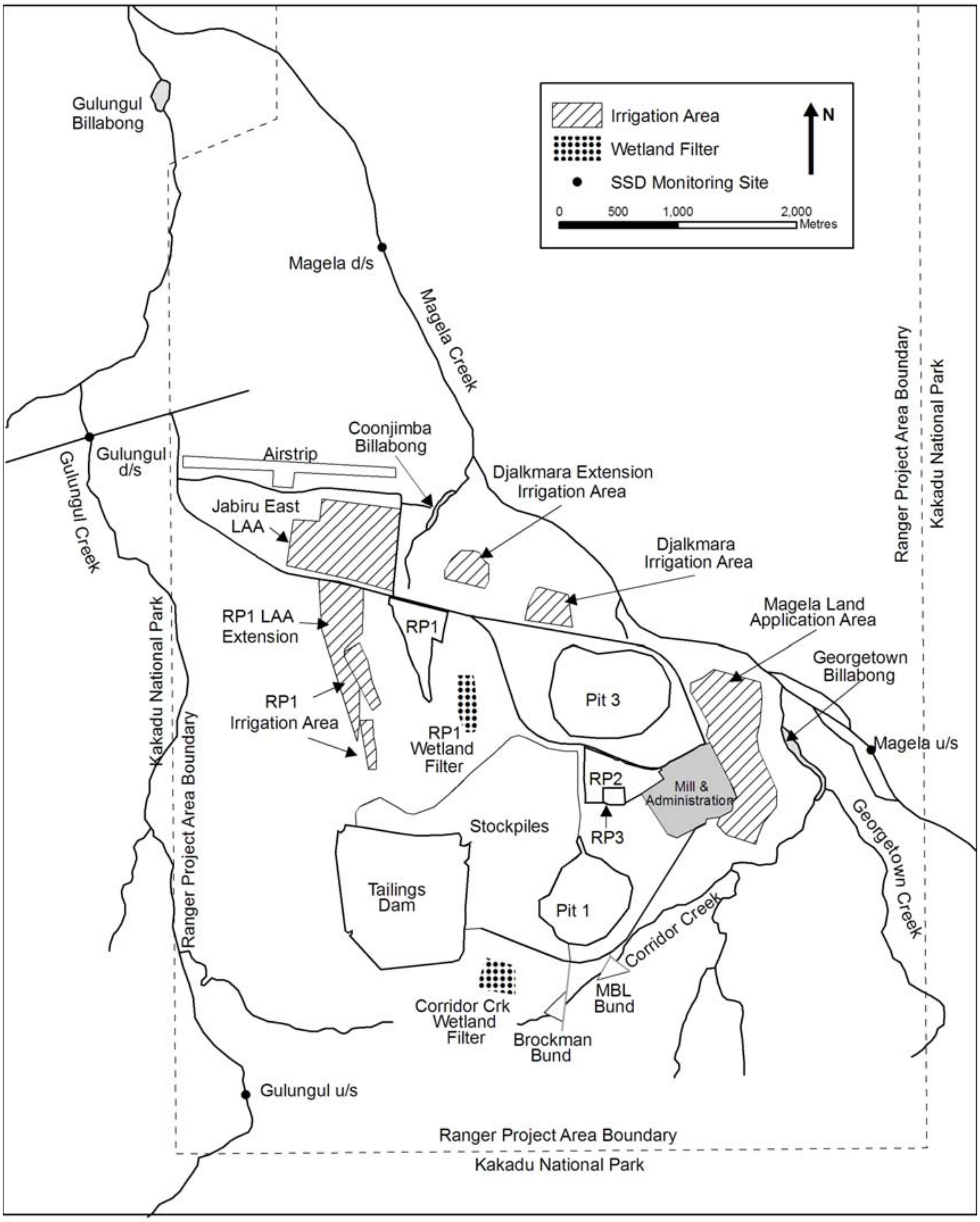
The final section of this report contains a summary of the non-uranium mining related external projects carried out in 2006–07.

Dr DR Jones

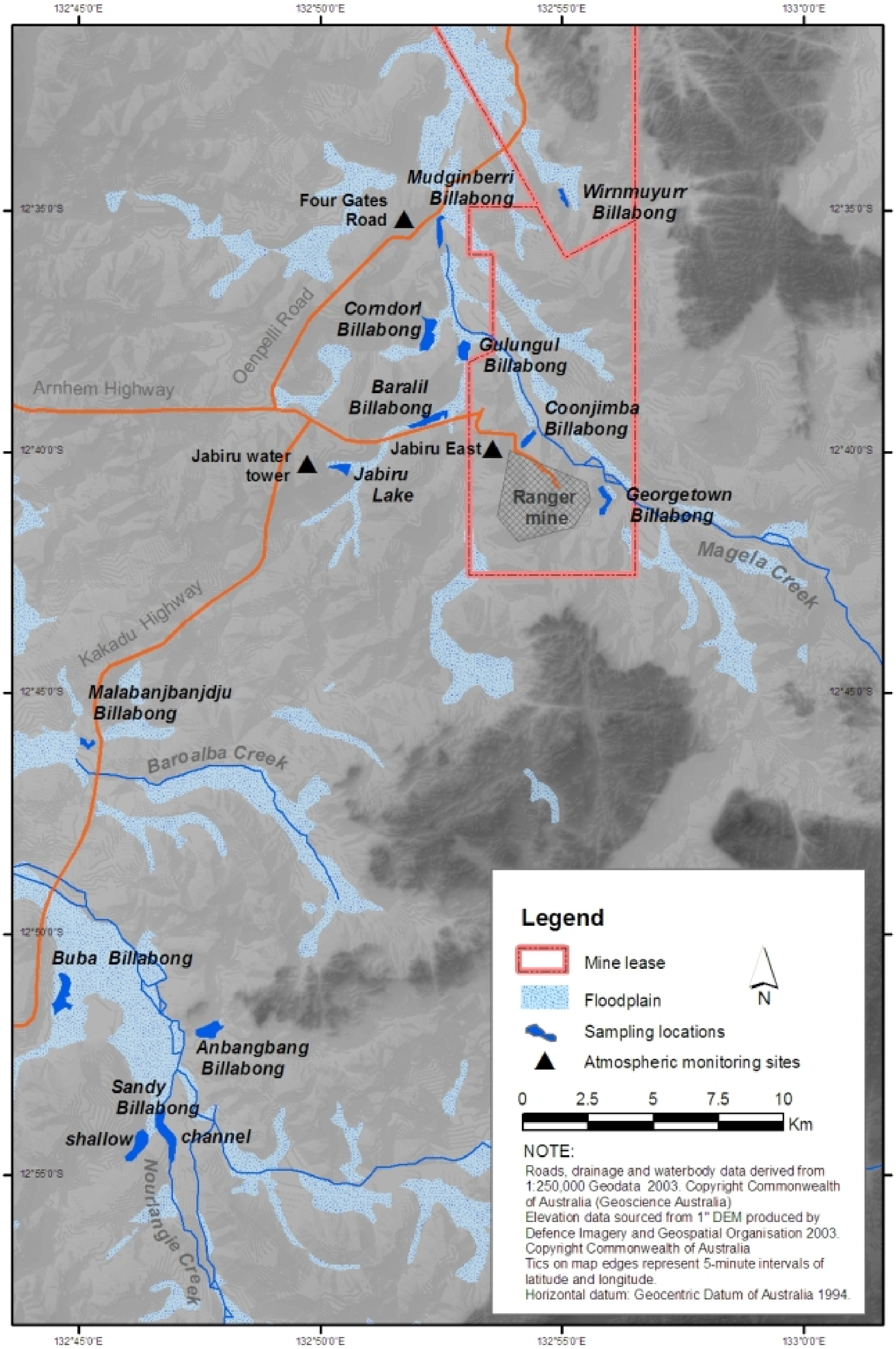
Director, Environmental Research Institute of the Supervising Scientist



Map 1 Alligator Rivers Region



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



Map 3 Location of water bodies and atmospheric monitoring sites used in the SSD's environmental monitoring programs

Part 1: Ranger – current operations

Contents[1]

1.2 ONGOING OPERATIONAL ISSUES

KKN 1.2.4 Ecotoxicology

1.3 MONITORING

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

1 List of papers grouped by Key Knowledge Need

Chronic toxicity of uranium in Magela Creek water to freshwater snail *Amerianna cumingi* and duckweed *Lemna aequinoctialis*

R van Dam, A Hogan & M Houston

Background

Over the past few years, uranium chronic toxicity data have been acquired for two additional species, the duckweed, *Lemna aequinoctialis*, and the freshwater snail, *Amerianna cumingi*. The majority of results were reported by van Dam et al (2006, 2007). Work required during 2006–07 to complete the project included the completion of several tests to better understand the observed loss of uranium from test solutions in the *A. cumingi* test system. Previous experiments have shown that, when integrated across the entire exposure duration, there typically was a 25% loss of uranium from the test solutions (see van Dam et al 2007). Following completion of the final experiments, this research was to be written up and submitted as a peer-reviewed journal manuscript.

Progress, results and discussion

Two experiments were undertaken, as follows:

i Assess the proportion of uranium lost from solution due to (i) adsorption to the test containers/tubes, (ii) adsorption to/absorption by deposited snail mucus and egg masses, and (iii) uptake by the snails; and

ii Assess the proportion of uranium lost from solution fur to the lettuce feed.

The design of Experiment 1 is illustrated in Figure 1. Snail test beakers (2 L) and tubes (six tubes per beaker) were set-up as per normal. Four treatments (A–D) were then prepared from the test beakers, and each treatment was exposed to both 48 and 470 µg/L uranium. The 'priming' of treatments for the deposition of snail mucus and egg masses was achieved by placing snails in the relevant beakers/tubes for 24-h prior to the uranium exposure.

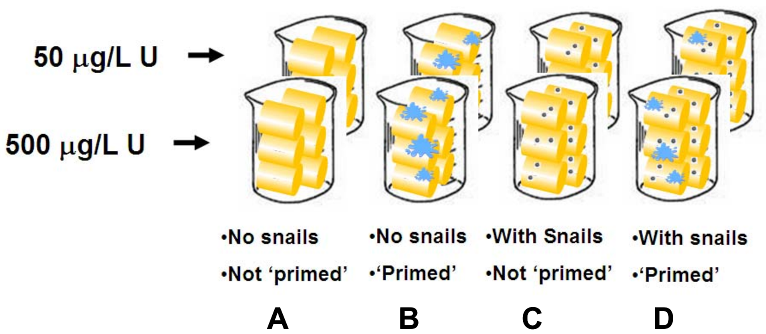


Figure 1 Design of experiment designed to assess proportion of uranium lost from solution due to (i) adsorption to the test containers/tubes, (ii) absorption by deposited snail mucus and egg masses, and (iii) uptake by the snails

Treatment A contained no snails and was not ‘primed’ with snail mucus and egg masses, hence it was assessing adsorption of uranium to the test containers/tubes only (ie. it acted as the experimental blank); Treatment B contained no snails but the test containers/tubes were ‘primed’ with snail mucus and egg masses (ie the snails left in for 24-h prior to uranium exposure were removed just prior to test commencement), hence it was assessing uranium adsorbed to/absorbed by the snail mucus and egg masses; Treatment C contained snails but the test containers/tubes had not been ‘primed’ with snail mucus and egg masses, hence it was assessing uranium lost primarily due to uptake by the snails; Treatment D contained snails and was ‘primed’ with snail mucus and egg masses, hence was assessing all the above loss pathways. The test was run for 24-h, as this is the standard period of test solution renewal for the *A. cumingi* test. At the end of the test, uranium was measured in the test waters as well as the whole snails.

The results for the 470 µg/L uranium treatments are shown in Figure 2. The loss of uranium from the test solutions across all treatments was between 56–64%, and there were no statistically significant differences between any of the treatments ($P < 0.05$). There were some statistically significant differences between the treatments exposed to 48 µg/L (data not shown), but overall, the loss of uranium from the test solutions across all treatments was still relatively tight, between 67–80%. The differences that were observed suggested that uranium loss was slightly less in treatments containing snails. Other than ‘reworking’ or remobilisation into the water column of uranium adsorbed to the test container/tube surfaces, it is unclear what could be causing this.

Uranium concentration in snails increased with increasing test solution uranium concentration (Table 1). However, the uranium uptake in the 48 µg/L and 470 µg/L test solutions accounted for only 4 and 6% of the total amount of uranium available, respectively. The results of this experiment indicated that the major loss pathway of uranium in the *A. cumingi* tests system is through adsorption to surfaces of the test container and tubes. Adsorbed uranium was considered not available to the snails, and, consequently, uranium concentration data for *A. cumingi* uranium toxicity tests were corrected accordingly (see van Dam et al 2007 for details).

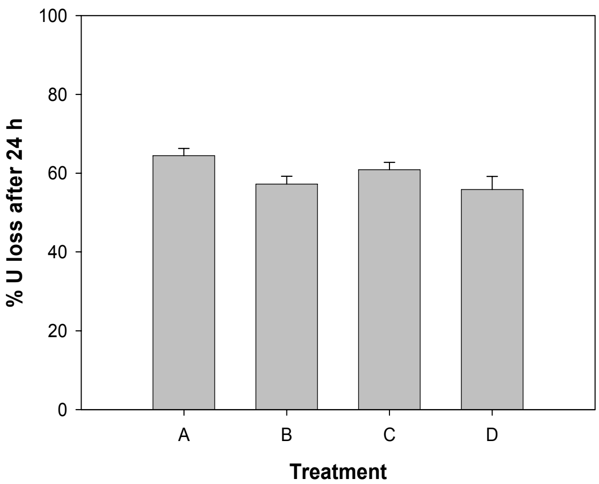


Figure 2 Percentage loss of uranium (spiked at 470 µg/L) from test solutions after 24-h for four different experimental treatments: A – no snails, surfaces not ‘primed’ with snail mucus/egg masses; B – no snails, surfaces ‘primed’ with snail mucus/egg masses; C – snails, surfaces not ‘primed’ with snail mucus/egg masses; D – snails, surfaces ‘primed’ with snail mucus/egg masses. There were no statistically significant differences between the treatments ($P < 0.05$).

Table 1 Concentration of uranium measured in the freshwater snail, *Amerianna cumingi*, following 24-h exposure to uranium

	Uranium treatment		
	Control (Magela Creek Water)	**48 µg/L**	**470 µg/L**
***A. cumingi* uranium concentration (mg/kg)**	0.05 (0.01)[a]	4.7 (0.6)	72 (6)

[a] Data represent means (±SEM) of three (Control treatment) or six (48 and 470 µg/L treatments) replicates.

Experiment 2 focused on the uptake of uranium by the lettuce fed to *A. cumingi* during testing. The results indicated that the lettuce took up approximately only 2–3% of the total uranium available in the test solutions. Consequently, the exposure concentration corrections based on the previous uranium fate tests were not adjusted based on these data.

Steps for completion

A journal manuscript, which incorporates both the *L. aequinoctialis* and *A. cumingi* uranium toxicity data is approximately 80% complete, and should be submitted before the end of 2007.

There is a need to revise the current uranium Limit for Magela Creek following the inclusion of the *L. aequinoctialis* and *A. cumingi* toxicity estimates into the uranium chronic toxicity dataset. However, this is not necessarily a straightforward exercise, as the most reliable toxicity estimates for *A. cumingi* are the IC values rather than the LOEC/NOEC values, which are the values currently in the dataset for the other species. Point estimate values, such as IC_{10} values, are generally considered more robust measures of toxicity than LOEC/NOEC values (OECD 1998). Consequently, the preferred approach is to reanalyse the existing uranium chronic toxicity data for the five species currently represented in the existing dataset, in order to calculate IC_{10} estimates, then derive a new Limit based on IC_{10} values. This task will be completed during 2007–08.

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

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

Background

Toxicity tests using freshwater species local to the ARR have been employed since the late 1980s to assess the toxicity of uranium. Some of the toxicity data have been used to derive a high reliability site-specific water quality Limit for Magela Creek downstream of Ranger, using the approach recommended by the Australian and New Zealand Water Quality Guidelines (WQGs; ANZECC/ARMCANZ 2000). The Limit of 6 µg/L was derived using chronic toxicity no-observed-effect concentration (NOEC) data, ranging from 18–810 µg/L, for 5 species (Hogan et al 2005). However, two of the NOEC values, 400 and 810 µg/L, 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 days exposure (+ 7 days 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.

The aims of this study were to (i) develop a 28 day larval growth chronic toxicity test, and (ii) use this test to assess the chronic toxicity of uranium, making comparisons to previous uranium toxicity data and discussing the implications for the uranium Limit for Magela Creek.

Toxicity test development

A 28 day larval growth test was developed for one of the fish species *M. mogurnda*, to assess the chronic toxicity of uranium[2]. In developing and optimising the toxicity test, various test design aspects were investigated, including test volume, fish stocking density, feeding regime and the influence of food ration on the fate of uranium in the test system. Initially, the study also attempted to develop such a protocol for *M. splendida inornata*. Despite numerous refinements to the experimental design including the development of a complex larval fish diet of commercial fish foods and live foods (*Paramecium*, *Chlorella* sp), adequate larval survival and growth over the 28 day test period could not be attained. Consequently, the remainder of the study focused on *M. mogurnda*.

The final test design for *M. mogurnda* was as follows: Newly hatched (<10 hours old) *M. mogurnda* larvae were exposed to various concentrations of uranium (control, 90, 184, 381, 768, 1397 and 3535 µg/L – measured concentrations) for 28 days. For each uranium 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–

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1 School of of Science and Primary Industries, Charles Darwin University, Darwin, NT.

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

28, respectively. A previous feeding trial experiment found that a similar feeding regime resulted in strong larval growth over a 28 day test period (Figure 1).

For the uranium toxicity test, various endpoints were measured or calculated based on data recorded throughout the 28 day exposure period. However, only larval survival, length and dry weight at the end of the 28 day exposure period are presented here. Analysis of the remaining data was being finalised at the time of preparation of this summary.

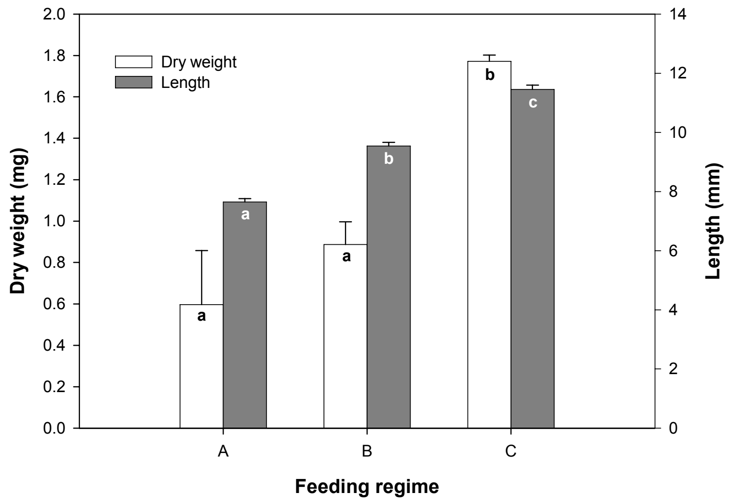

Figure 1 Effect of feeding regime on mean (±SEM; n=6) dry weight and length of larval purple-spotted gudgeon (*Mogurnda mogurnda*) over 28 days. Feeding regimes were as follows: *A.* 10–20 (days 1–6), 20–30 (days 7–23) and 30–40 (days 24–28) nauplii, once a day; *B.* 5–10 (days 1–6), 10–20 (days 7–23) and 20–30 (days 24–28) nauplii, twice a day; and *C.* 10–20 (days 1–6), 20–30 (days 7–23) and 30–40 (days 24–28) nauplii, twice a day. Bars for dry weight and length that do not have a letter in common are significantly different from each other ($P \le 0.05$).

Chronic toxicity of uranium

The effects of uranium exposure to larval *M. mogurnda* are shown in Figure 2. Exposure to 3535 µg/L uranium resulted in 100% larval mortality, with all fish dying within the first 24 hours of exposure. No significant mortality was observed at any of the other uranium concentrations. Larvae exposed to 1397 µg/L uranium exhibited significant reductions in length (13%; $P < 0.001$) and dry weight (30%; $P < 0.001$) relative to control larvae. Larvae exposed to uranium concentrations below 1397 µg/L showed no adverse effects to exposure over the test period relative to the control (unexposed) larvae. Based on larval length and dry weight, the lowest-observed-effect concentration (LOEC) and no-observed-effect concentration (NOEC) were 1397 and 768 µg/L, respectively.

The toxicity of uranium to *M. mogurnda* following 28 days exposure did not appear to be higher than that previously reported by Holdway (1992) following shorter (ie 7 and 14 day) 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 uranium on *M. mogurnda*, and that longer exposure periods will not necessarily result in a more sensitive response. Consequently, the historical 7 day fish toxicity test results used for the derivation of the current uranium Limit of 6 µg/L appear to be reasonably representative of uranium 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 effects of U.

Steps for completion

Further analysis and interpretation of the *M. mogurnda* chronic dataset will enable a closer examination of this issue. To complete this study, an additional 28 day larval growth toxicity test will be undertaken to more closely measure effects on *M. mogurnda* growth between the uranium concentrations of 768 and 3535 µg/L. In addition, uranium concentrations in larval fish tissues exposed for 28 days in both toxicity experiments will be measured.

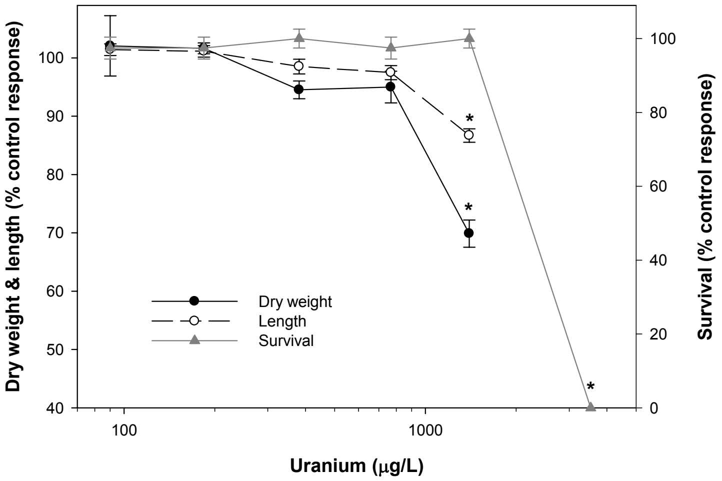

Figure 2 Effect of 28 days exposure to uranium on mean (±SEM) dry weight (n=4), length (n=37–39) and survival (n=4) of larval purple-spotted gudgeon (*Mogurnda mogurnda*), normalised against the control responses. Data points accompanied by an asterisk are signficantly different from the control responses ($P \leq 0.05$). Mean control responses (±SEM) were as follows: dry weight – 2.02 (±0.13) mg; length – 11.8 (±0.2) mm; and survival – 98 (±3)%.

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Toxicity of magnesium sulfate in Magela Creek water to tropical freshwater species

R van Dam, A Hogan, C McCullough[1] & C Humphrey

Background

Magnesium sulfate ($MgSO_4$) is a major contaminant of Ranger mine waste waters. For the past seven years, a PhD research project and a linked research project of the ***eriss*** Ecotoxicology Program has been assessing the effects of $MgSO_4$ on aquatic species and communities of Magela Creek. Two main approaches were used: (i) laboratory ecotoxicity tests using natural creek water on local test species, and (ii) field mesocosm experiments within the Creek on naturally-colonising biological communities. The project involved assessments of:

- The toxicity of Mg in Magela Creek water to six local freshwater species;
- The toxicity of Mg to aquatic communities in mesocosm experiments;
- The influence of calcium (Ca) on Mg toxicity to three local freshwater species; and
- The toxicity of Mg in the presence of elevated Ca (to maintain a Mg:Ca ratio of 9:1) to six local freshwater species.

Much of the research has been summarised previously (van Dam & Humphrey 2004, van Dam et al 2006, 2007). Briefly, key findings were that (i) Mg was markedly more toxic to several species than had previously been reported in the literature; (ii) this was thought to be due to the very low natural background Ca concentration in Magela Creek water; and (iii) maintaining a Mg:Ca (mass) ratio at ~9:1 provided protection against, and thereby reduced, Mg toxicity. The summary below updates much of the data and analyses reported by van Dam et al 2006).

Magnesium toxicity results

Several final laboratory ecotoxicity experiments were completed in early 2006–07, prior to the study's findings being presented at the Interact 2006 conference in Perth. Toxicity estimates for each species were calculated using non-linear regression (3 parameter logistic or sigmoidal) on data from 3–4 experiments, pooled after analysis of covariance was undertaken to confirm the validity of pooling the data.

The final toxicity results for Mg, in the presence of background Ca only (ie at Mg:Ca >9:1) and elevated Ca (ie at a Mg:Ca ~9:1) are presented in Table 1. In the presence of elevated Ca (ie at a Mg:Ca ratio of ~9:1), Mg was less toxic than in the presence of background Ca only. However, the magnitude of the difference varied widely between the species, possibly due to differences in species physiology and mechanisms of Mg toxicity. The ameliorative effect of increased Ca on Mg toxicity was likely to be due to an reduced ability for Mg to compete with Ca for Ca receptor sites, and therefore interfere with Ca-dependent processes.

[1] Centre for Ecosystem Management, Edith Cowan University, Joondalup, WA.

Derivation of magnesium water quality Limits

Using the BurrliOZ (Burr Type III distribution) species sensitivity distribution (SSD) approach recommended by ANZECC/ARMCANZ (2000), water quality Limits were derived for Mg in Magela Creek, at background Ca (ie Mg:Ca ratio >9:1) and at a Mg:Ca ratio of ≤9:1, based on the 99% species protection level (due to the high conservation value of the Magela Creek catchment). The resultant SSDs are shown in Figure 1, and the water quality Limit values (lower, upper 95% confidence limits) are as follows:

- When Mg:Ca ratio >9:1: **1.1 (0.8, 3.6) mg/L**
- When Mg:Ca ratio ≤9:1: **4.6 (2.1, 38) mg/L**

Table 1 Toxicity of Mg in the presence of only background Ca (ie Mg:Ca ratio >9:1) and in the presence of elevated Ca (ie Mg:Ca ratio ~9:1)to six local freshwater species

Species	Mg toxicity (mg/L)			
	@ background Ca (Mg:Ca >9:1)		@ Mg:Ca ~9:1	
	IC_{15}	IC_{50}	IC_{15}	IC_{50}
Chlorella sp. (72-h cell division rate)	87 (50, 134)[a]	1215 (695, 2127)	1184 (755, 1597)	4205 (3158, 5546)
Lemna aequinoctialis (96-h plant growth)	2.3 (1.9, 2.6)	4.4 (4.0, 4.8)	66 (33, 110)	629 (414, 956)
Amerianna cumingi (96-h embryo production)	5.7 (3.9, 7.3)	13 (11, 16)	10 (2, 25)	118 (53, 264)
Moinodaphnia macleayi (3 brood reproduction)	41 (34, 46)	63 (56, 70)	49 (31, 65)	122 (99, 151)
Hydra viridissima (96-h population growth)	2.6 (2.1, 3.2)	11 (10, 13)	345 (260, 412)	713 (646, 780)
Mogurnda mogurnda (96-h survival – acute)	12[b] (0.1,19.9)	41[c] (33, 50)	4009[b] (3850, 4025)	4054[c] (4046, 4063)

a Values in parentheses represent 95% confidence limits

b Represents LC_{05} value

c Represents LC_{50} value

Based on historical Mg and Ca concentration data, the Mg:Ca ratio in Magela Creek downstream of Ranger rarely, if ever, exceeds 9:1 (Figure 2). This is because mine waters that leave the site contain elevated concentrations of Ca in addition to Mg. Consequently, the Mg Limit that will be most often applied will be the one based on the Mg:Ca ratio of ≤9:1. Figure 2 compares the cumulative probability distributions of Mg concentration and the Mg:Ca ratio in Magela Creek downstream of Ranger (1985–2007), with the Mg Limit of 4.6 mg/L and the Mg:Ca 'trigger' ratio of 9:1. Based on historical and existing operations, the risks of exceeding the Mg concentration or Mg:Ca ratio trigger values appear negligible. However, it should be noted that EC data being produced by the continuous monitoring of water quality in Magela Creek suggest that the concentration of Mg downstream of Ranger may in fact exceed the Limit of 4.6 mg/L more often than indicated by both Ranger's statutory monitoring program and SSD's current routine monitoring program (see continuous monitoring summary under KKN 1.3.1). Both of these monitoring programs comprise weekly grab samples. The possible effects of transient exceedances of the 4.6 mg/L value can be examined, in part, using the results from the concurrent in situ/creekside toxicity monitoring program (see KKN 1.3.1).

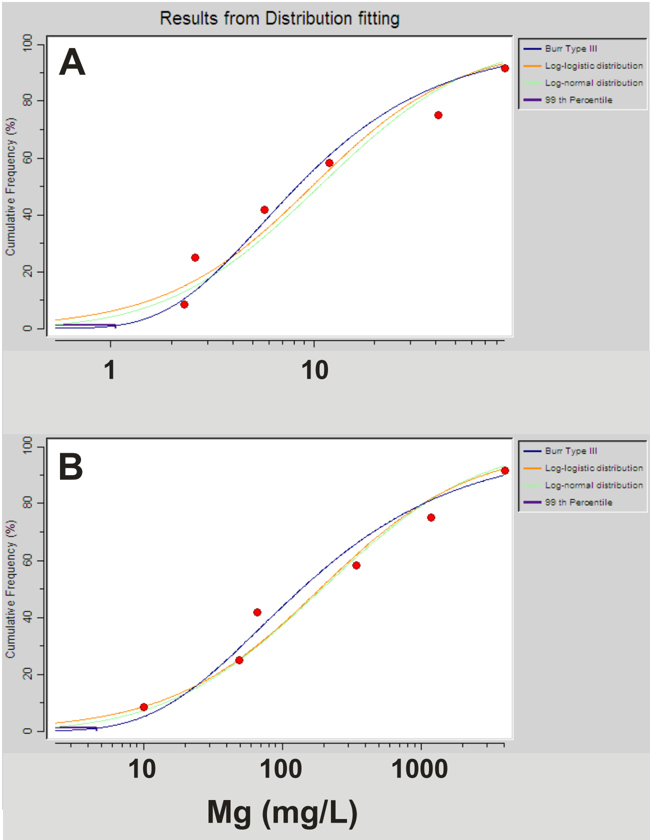

Figure 1 Species sensitivity distributions for Mg toxicity at (A) natural Magela Creek background Ca concentration (ie. ~0.2 mg/L) and (B) a constant Mg:Ca ratio of 9:1 (Note the different *x* axis scales). Data points represent the IC_{15} toxicity values for each species (LC_{05} values for *M. mogurnda*; see table 1) and are plotted as the cumulative frequency. The Burr Type III distribution is represented by the blue (darkest) fitted curve, and is the distribution used to calculate the trigger values for Mg (as recommended by ANZECC/ARMCANZ 2000).

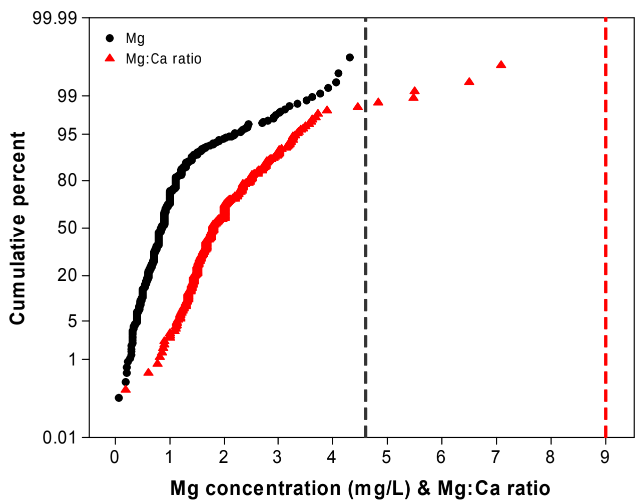

Figure 2 Cumulative probability distributions for Mg concentration and Mg:Ca ratio in Magela Creek downstream of Ranger, from 1985–2007 (data from ERA's database). The left reference line represents the Mg trigger value of 4.6 mg/L (when the Mg:Ca ratio is maintained at ≤9:1) and the right reference line represents the 'safe' Mg:Ca ratio of 9:1.

The Mg Limit of 1.2 mg/L which applies for a Mg:Ca ratio of >9:1 is of little relevance to the current context since the Ca concentration would need to be <0.13 mg/L for an exceedance to occur. According to historical (1985–2007) water quality data for Magela Creek downstream of Ranger this occurs <1% of the time. However, it is possible that in the long term following rehabilitation of the site, the current dominant condition of Mg:Ca <9:1 could change as solute inputs to the Creek become more influenced by the Mg-rich water contained in groundwater plumes originating from the tailings repositories in Pits 1 and 3.

Recently, alternative approaches to analysing toxicity data and deriving water quality Limits using toxicity data from multiple species have been under consideration (eg probability bounds analysis, hierarchical Bayesian mixed models). These efforts, in collaboration with Dr Bill Dixon (Arthur Rylah Institute, Melbourne) are still underway, and some initial results may be presented during the ARRTC meeting.

Steps for completion

A journal manuscript, which incorporates all the laboratory-based Mg toxicity data is approximately 50% complete, and will be submitted before June 2008. This project, and the proposed Mg trigger values, were recently presented to the Ranger Minesite Technical Committee (MTC). It is expected that the trigger values will be formalised in the near future.

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Ecotoxicological assessment of Ranger mine pond waters

R van Dam, A Hogan, M Houston & N Lee

Background

An extreme rain event (840 mm over 5 days) at Jabiru East in late February/early March 2007 resulted in a major increase in the water inventory and associated on-site water management issues for Ranger mine (Klessa et al 2007). Consequently, a number of options to reduce the volume of pond waters stored on site were proposed for urgent consideration and assessment. Given that flow in Magela Creek remained high for some time after the event (total discharge during March at station G8210009 was 535 GL, almost 50% greater than the mean annual discharge of 366 GL), the short-term direct release of untreated pond water from Retention Pond 2 (RP2) and/or Pit 3 to the creek was one of the water reduction initiatives that could have been implemented in the event that approval was given by the supervising authority. Pond water is derived from rainfall that falls on the active mine-site catchments, and typically has a uranium concentration between 2000 and 5000 µg/L.

In order to meet the primary objective of maintaining the natural biological diversity of aquatic ecosystems in the Alligator Rivers Region (as set out in the Environmental Requirements of the Commonwealth for the Operation of Ranger), it was essential that ecotoxicity testing be undertaken on pond water prior to any discharge off site. Site-specific toxicity of mine waters to a range of local aquatic organisms is undertaken to estimate dilution rates that provide environmental protection downstream of the mine.

Using such data, ERA and stakeholders could determine the ability to release pond water based on the existing flow regime in the creek. In addition, it was intended that the data would add to knowledge and subsequent decisions about water management options for future wet seasons.

Methods

The toxicity of RP2 water (dissolved uranium concentration – 1900 µg/L; pH – 6.8; EC – 1200 µS/cm) was assessed using the following five local species: unicellular green alga (*Chlorella* sp); macrophyte (duckweed; *Lemna aequinoctialis*); cnidarian (*Hydra viridissima*); cladoceran (water flea; *Moinodaphnia macleayi*); and a fish species (*Mogurnda mogurnda*). The test species were exposed to various RP2 water concentrations ranging from 0.3 to 100% RP2 water (diluted in Magela Creek water) and a Magela Creek water control. Three of the species most sensitive to RP2 water, *M. macleayi*, *H. viridissima* and *L. aequinoctialis*, were subsequently used to assess the toxicity of Pit 3 water (dissolved uranium concentration – 1600 µg/L; pH – 7.7; EC – 890 µS/cm), by exposing them to various Pit 3 water concentrations ranging from 0.25 to 100% (diluted in Magela Creek water) and a Magela Creek water control.

Results and discussion

RP2 water exhibited a wide range of toxicity to the species studied (Figure 1 & Table 1). Based on IC_{50}/LC_{50} values, the order of sensitivity of the five test species was:

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

With the exception of *H. viridissima* and *Chlorella* sp, the order of sensitivity corresponded reasonably well with the documented order of sensitivity to uranium for these species. However, absolute values of RP2 water toxicity did not necessarily correlate to known uranium toxicity, possibly due to contributions to toxicity from other potentially toxic metals (eg copper, lead, nickel, zinc) and/or a vast range of physico-chemical interactions that occur in complex mixtures, but which are not accounted for in single toxicant toxicity testing.

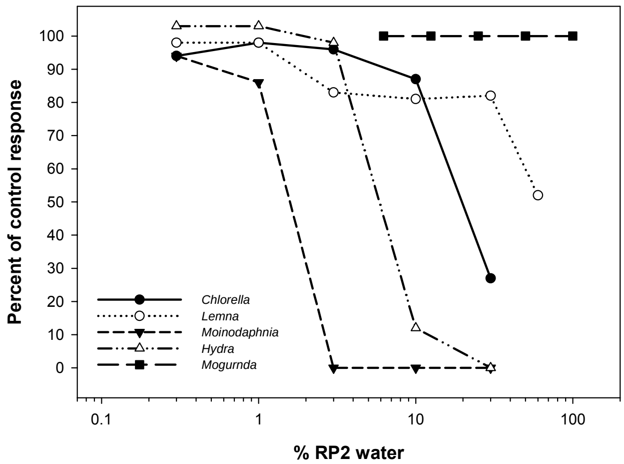

Figure 1 Combined concentration-response plot for each species, presented as a percent of the control response with regard to RP2 water concentration. Data points and standard error bars have been removed to aid visual interpretation.

Table 1 Summary of toxicity of RP2 water (expressed as % RP2 water) from Ranger mine to five local freshwater species

Species	IC_{10}[a] (95%CL)[b]	IC_{50}[c] (95%CL)
Moinodaphnia macleayi (cladoceran)	0.6 (0.2–0.8)	1.84 (1.81–1.86)
Hydra viridissima (green hydra)	3.5 (0–4.2)	6.9 (5.4–8.2)
Chlorella sp (unicellular alga)	7.5 (0–16)	22 (19–27)
Lemna aequinoctialis (duckweed)	2.1 (0.5–2.2)	>60
Mogurnda mogurnda (fish)	>100[d]	>100[d]

a IC_{10}: concentration that results in a 10% inhibition of response relative to the control response.

b 95% CL: 95% confitdence limits

c IC_{50}: concentration that results in a 50% inhibition of response relative to the control response.

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.

In order to calculate a protective dilution of RP2 water in Magela Creek water, a species sensitivity distribution was constructed using the IC_{10} (or, for *M. mogurnda*, LC_{05}) toxicity estimates for the five different species tested (see Figure 2; NB – a conservative value of 50% was assumed for *M. mogurnda*) using the statistical (Burr Type III) model used for the

Australian and New Zealand Water Quality Guidelines (ANZECC/ARMCANZ 2000). The concentration of sample that would be protective of 99% of species (due to the high conservation value of the receiving catchment), was extrapolated using this model and recommended as the 'protective' dilution rate.

Based on the resultant species sensitivity distribution (Figure 2), a protective dilution of 0.33% RP2 water (or 1 part RP2 water to ~300 parts Magela Creek water) would be expected to ensure 99% species protection for the downstream aquatic ecosystem. The uranium concentration at this dilution would have equalled approximately 6 µg/L, which is the same as the current site-specific uranium Limit of 6 µg/L.

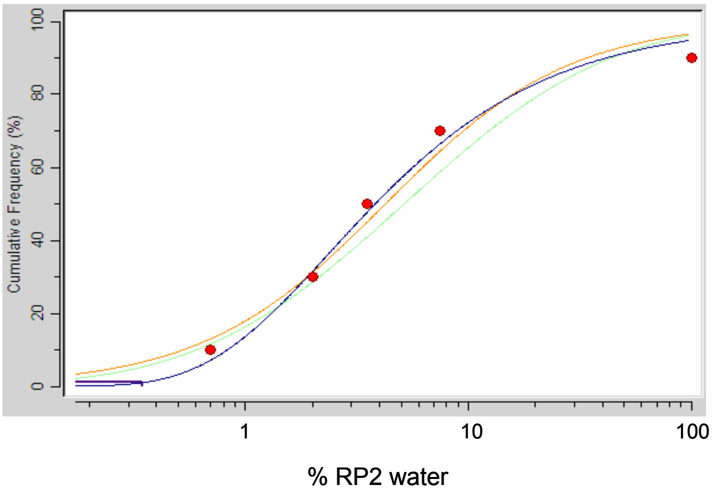

Figure 2 Species sensivity distribution based on IC_{10} values (and assuming a conservative value of 50% for *M. mogurnda*). The Burr Type III distribution is represented by the black (darkest) fitted curve and was the distribution used to calculate the protective dilution for RP2 water.

At this recommended dilution the concentration of magnesium would have been approximately 0.5 mg/L, which is well below the interim site-specific magnesium Limit of 4 mg/L and, in fact, is close to natural background concentrations of magnesium.

The toxicity of Pit 3 water to the three species assessed is shown in Figure 3 and summarised in Table 2. Based on IC_{50}/LC_{50} values, the order of sensitivity of the five test species was: *M. macleayi* >> *H. viridissima* >> *L. aequinoctialis*. A comparison of the toxicity of RP2 water and Pit 3 water to the three species (using analysis of covariance) found that the toxicity of Pit 3 water to *M. macleayi* and *H. viridissima* was not significantly different to that of RP2 water. However, the toxicity of Pit 3 water to *L. aequinoctialis* was significantly less than that of RP2 water.

The similarity in toxicity between the two water types for two of the species was not unexpected given the similar uranium concentrations and the fact that RP2 receives water pumped from Pit 3. The difference in toxicity between the two water types for *L. aequinoctialis* may have been due to differences in (i) test organism responses between the tests, (ii) important physico-chemical variables that influence metal bioavailability and toxicity (eg. pH), and/or (iii) concentrations of, or interactions between, potentially toxic metals other than uranium (eg Cu, Pb, Ni, Zn).

At the time of completion of the toxicity assessments, flow in Magela Creek was insufficient for the direct release of Pond Water at acceptable/protective dilutions (ie at least 1 in 300) to have significantly reduced the on-site water inventory. Consequently, this option was not progressed further. However, notwithstanding temporal variations in Pond Water composition/quality and how they may affect toxicity, the knowledge gained from the study will form a key part of the knowledge base required for the future evaluation of water management options at the Ranger mine.

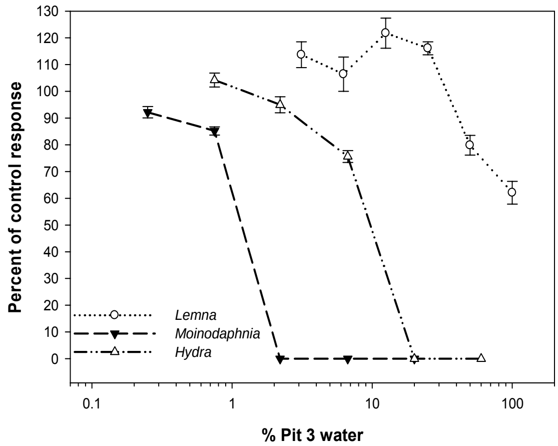

Figure 3 Combined concentration response plot for each species presented as a percent of the control response with regard to Pit 3 water concentration. Data points represent the mean (±SEM) of three replicates for *L. aequinoctialis* and *H. viridissima*, and 10 replicates for *M. macleayi*.

Table 2 Summary of toxicity of Pit 3 water (expressed as % Pit 3 water) from Ranger mine to three local freshwater species

Species	IC_{10}[a] (95%CL)[b]	IC_{50}[c] (95%CL)
Moinodaphnia macleayi (cladoceran)	0.4 (0.2–0.6)	1.35 (1.31–1.38)
Hydra viridissima (green hydra)	2.9 (0.9–5.3)	11 (10–12)
Lemna aequinoctialis (duckweed)	34 (31–37)	>100

a IC_{10}: concentration that results in a 10% inhibition of response relative to the control response.

b 95% CL: 95% confidence limits

c IC_{50}: concentration that results in a 50% inhibition of response relative to the control response.

References

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Toxicity of treated pond water from Ranger uranium mine to five local freshwater species

R van Dam, A Hogan, M Houston & N Lee

Background

In December 2005, the toxicity of treated Pond Water from the Ranger water treatment plant was assessed. The treated Pond Water had a uranium concentration of ~4 µg/L. The results, summarised by van Dam et al (2007), indicated that the treated Pond water was non-toxic to four of the five species assessed. The fifth species, *Moinodaphnia macleayi*, was adversely affected by treated Pond Water, with ~25 and 50% reductions in repdroduction for test animals exposed to 50 and 100% treated Pond Water, respectively. At the time, two hypotheses to explain the effect on *M. macleayi* were proposed:

1 Enhanced uranium toxicity due to a lack of dissolved organic carbon (DOC) in the treated water

2 A nutrient/essential ion deficiency due to the minerally deficient nature of the treated water.

Methods

To test these hypotheses, the toxicity of uranium to *M. macleayi* 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 (NMCW), were assessed and compared. The 3-brood (6 day) reproduction tests were run concurrently. The comparison not only served to assess for differences in the toxicity of uranium in the two types of water, but also provided an indication of the performance of *M. macleayi* in SMCW alone (ie in the absence of uranium).

Results and discussion

The results of the toxicity comparison are shown in Figure 1 and summarised in Table 1. It was evident that the reproductive performance of *M. macleayi* was significantly lower in SMCW than NMCW (see control reproduction values in Table 1), regardless of exposure to uranium. This may have been due to the lack of organic nutrients and/or essential ions in the SMCW (although the test animals are fed daily with a mixture of *Chlorella* sp and fermented food supplement, FFV; Riethmuller et al 2003), and suggests that the previously observed effect of treated Pond Water on *M. macleayi* may have been due to such a factor/s.

However, it was also evident from the experiment that the toxicity of uranium in SMCW was significantly higher than in NMCW. Based on the IC_{50} values, uranium toxicity was about three fold higher in SMCW than in NMCW. Thus, while the data from this experiment suggest that the nutrient/ion deficient nature of treated Pond Water is the most likely reason for the adverse effect it has on *M. macleayi*, it would appear that enhanced uranium toxicity has the potential to be a contributing factor. Further experiments are needed to gain a better understanding of this.

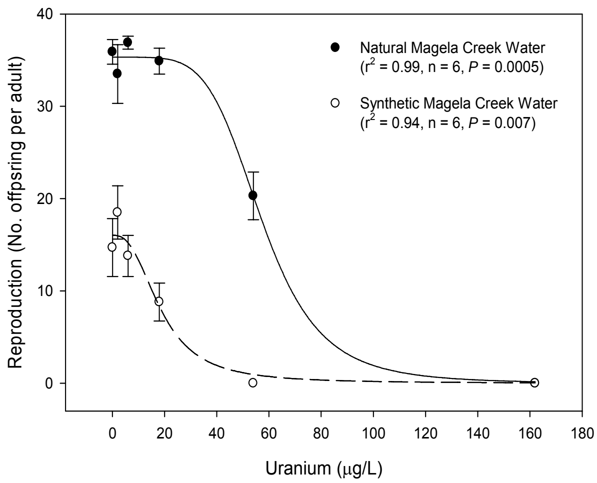

Figure 1 Effect of natural and synthetic Magela Creek water on the reproduction of, and uranium toxicity to, *Moinodaphnia macleayi*. Data points represent the mean (±SEM) of 10 replicates. Both curve fits represent a 3-parameter logistic model.

Table 1 Summary of the effect of natural and synthetic Magela Creek water on the reproduction of, and uranium toxicity to, *Moinodaphnia macleayi*

Diluent	Control offspring/adult (±SEM)	Uranium toxicity (µg/L)	
		IC_{20} (95% CL)	IC_{50} (95% CL)
Natural Magela Creek water	35.9 (±1.3)	44 (37, 49)	57 (53, 63)
Synthetic Magela Creek water	14.7 (±3.1)	11 (<1, 19)	18 (11, 35)

Based on measured uranium concentrations in treated Pond Water from the existing Pond Water treatment plant during 2006–07 (ERA 2006, 2007a), the IC_{20} and IC_{50} concentrations for uranium in SMCW were exceeded approximately 15% and 7% of the time, respectively[1]. These results would suggest there is a need to better quantify the potential for uranium in treated Pond Water to be toxic to *M. macleayi*. It should be noted, however, that treated Pond Water does not represent a threat to the off-site environment, because it is not discharged directly to Magela Creek, but to a range of on-site locations including RP1 and the wetland filters (ERA 2007b).

Update on July 2007 treated Pond Water toxicity study

In July 2007, the toxicity of treated Pond Water from a mobile (reverse osmosis) water treatment unit at Ranger was assessed, again using the routine suite of five species. On this occasion, the treated Pond Water had a uranium concentration of 8 µg/L. By and large, the results were the same as for the 2005 toxicity study. Treated Pond Water exhibited low to no toxicity for all species except *M. macleayi*, where the effect was very similar to that reported in the 2005 study (Figure 2). The results of the recent study, which will be reported in full to

[1] One value (ie 72 µg/L) from the 2006–07 Pond Water permeate uranium dataset was excluded from the analysis because it exceeded the water quality criterion for uranium in Pond Water permeate of 40 µg/L. Such exceedences are reflective of a problem with the water treatment process, and are not indicative of typical water quality during normal operating conditions. When exceedences are detected, discharge from the Pond Water treatment plant to the various on-site wetland filters and retention ponds is ceased.

ARRTC in 2008, have further affirmed the need to elucidate the the cause/s of the adverse effects exhibited by *M. macelayi* exposed to treated Pond Water, as well as the potential for uranium in treated Pond Water to exhibit toxicity, and the reasons for this toxicity.

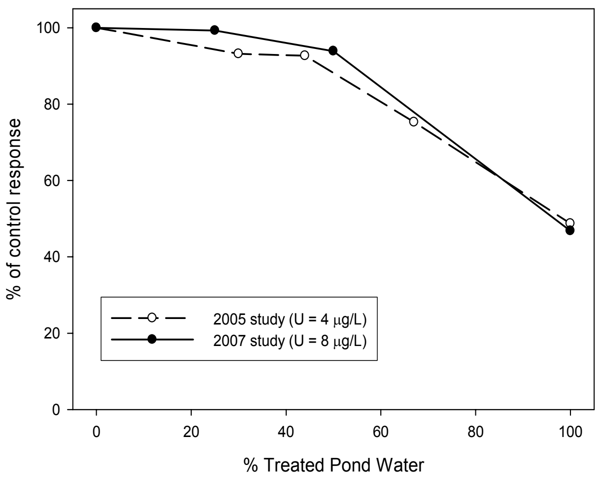

Figure 2 Toxicity of treated Pond Water in 2005 (on-site water treatment plant) and 2007 (mobile water treatment unit) to *Moinodaphnia macleayi*. Data points represent the mean of 10 replicates (error bars not shown).

Steps for completion

This program of work includes the assessment of the toxicity of treated process water, which is expected to occur in 2008. Following this, the research will be published as a Supervising Scientist Report and possibly a peer-reviewed journal article.

To further test the hypotheses related to the increased sensitivity of *M. macleayi* to uranium in SMCW, an experiment will be undertaken exposing *M. macleayi* to various concentrations of uranium in one of three treatments: (i) SMCW; (ii) NMCW; and (iii) SMCW to which dissolved organic carbon has been added at the concentration measured in NMCW. This experiment will be an acute exposure, with the animals not being fed, in order to eliminate the food as a source of organic material.

To address the potential for uranium to be toxic in treated Pond Water, it would be possible to test treated Pond Water (with a low starting uranium concentration; eg <1µg/L) to which additional uranium (at various concentrations) has been spiked. Such a study would be relevant and informative to the overall issue and associated management of treated Pond Waters.

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

R van Dam, A Hogan, M Houston & N Lee

Background

Over the past three 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. Reference toxicant control charts were developed for four of the five routine testing species during 2004–05 and 2005–06. The aims for 2006–07 were to:

1 Establish a reference toxicity testing program and associated control charts for the cladoceran, *Moinodaphnia macleayi*; and

2 Continue with the reference toxicity testing programs for *Chlorella* sp, *Hydra viridissima*, *Lemna aequinoctialis* and *Mogurnda mogurnda*.

Progress

In total, 28 reference toxicants tests (*Chlorella* – 4; *Lemna* – 3; *Moinodaphnia* – 9; *Hydra* – 4 *Mogurnda* – 8) were completed during 2006–07, 21 of which provided valid results. The results of these tests are summarised in Table 1, with the control charts presented in Figure 1.

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 generally valid. However, for two of the reference toxicity tests, control growth was unacceptably low. For one test (770G) this may have been due to the pH of the synthetic soft water (SSW) test medium being too low (pH ~5). Following this, procedures for ensuring adequate equilibration time after the pH of SSW has been adjusted to a value of 6 were tightened.

L. aequinoctialis

The *L. aequinoctialis* reference toxicity test has proved to be problematic. The key 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 (esp. PO_4) and trace elements added to the test medium to facilitate plant growth, also interact with uranium, greatly reducing its bioavailability and toxicity. 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 growth medium). Despite some promising indications, consistent results could not be obtained. This issue is still under investigation. Another 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).

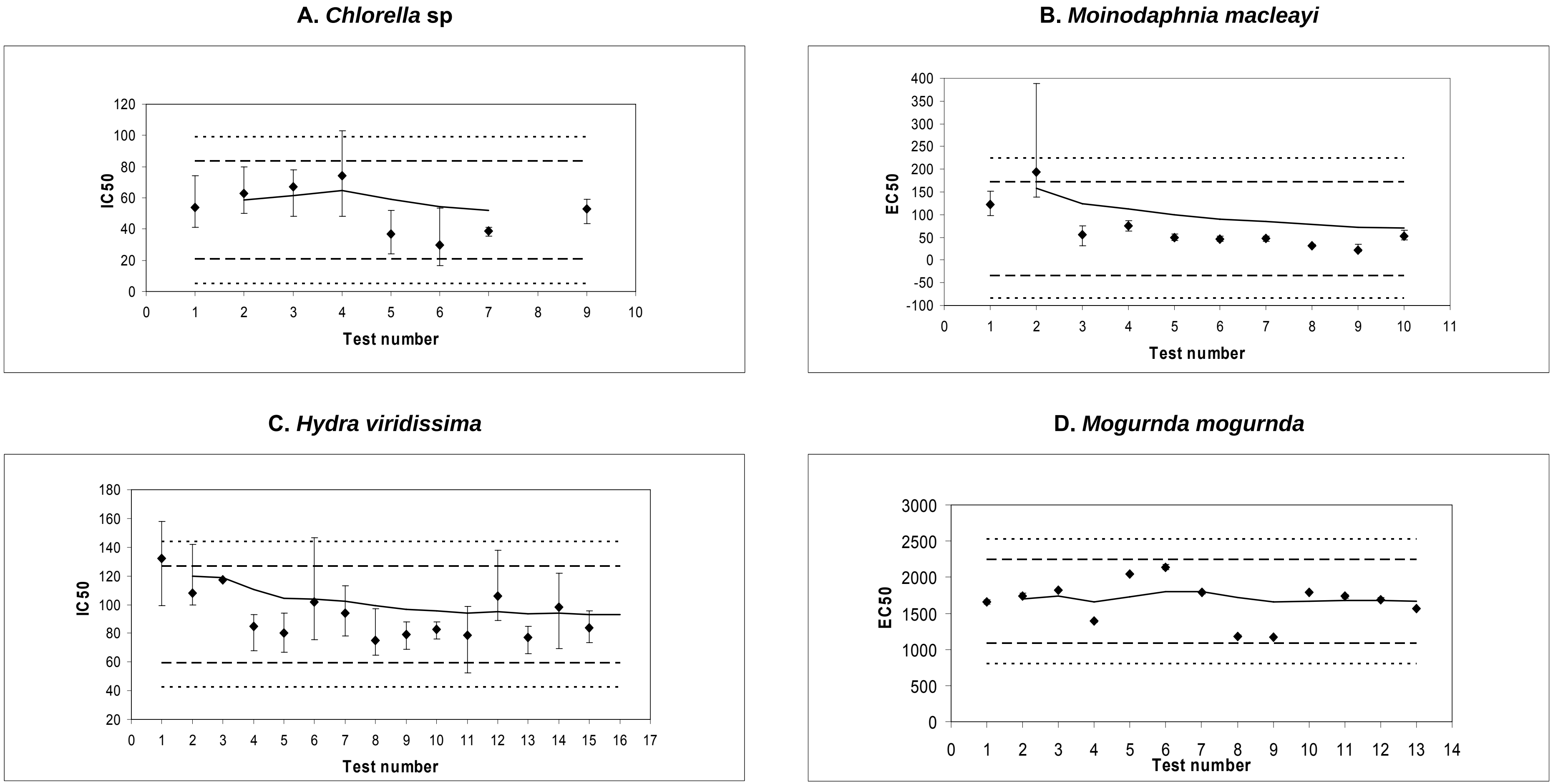

Figure 1 Reference toxicant control charts for A. *Chlorella* sp, B. *M. macleayi*, C. *H. viridissima* and D. *M. mogurnda*, as at September 2007. Data points represent IC_{50} or EC_{50} toxicity estimates. Reference lines represent the following: dotted lines – upper and lower 99% confidence limits; broken lines – upper and lower warning limits; unbroken line – running mean.

Table 1 Summary of uranium reference toxicity test results for 2006–07

Species & endpoint	Test Code	IC_{50} (µg/L)	Valid test?	Comments
Chlorella sp (72-h cell division rate)	770G	NC[a]	No	Low pH; insufficient control growth
	774G	39 (35, 41)[b]	Yes	
	783G	22 (14, 28)	No	Insufficient control growth
	839G	53 (44, 59)	Yes	
Lemna aequinoctialis (96-h population growth)	779L	NC[b]	No	Insufficient control growth[c]
	787L	>2000	No	No effect at highest conc[c]
	825L	>2040	No	Insufficient effect (18%) at highest conc[c]
Moinodaphnia macleayi (48-h immobilisation)	762I	122 (98, 152)	Yes	
	763I	194 (139, 388)	Yes	
	776I	57 (32, 75)	Yes	
	788I	76 (64, 87)	Yes	
	797I	50 (43, 57)	Yes	
	801I	47 (41, 53)	Yes	
	827I	48 (41, 52)	Yes	
	832I	32 (30, 35)	Yes	
	837I	22 (18, 34)	Yes	
Hydra viridissima (96-h population growth)	760B	78 (53, 98)	Yes	
	771B	106 (89, 138)	Yes	
	796B	77 (66, 85)	Yes	
	830B	98 (69, 122)	Yes	
Mogurnda mogurnda (96-h sac fry survival)	768E	1175 (1149, 1197)	Yes	
	773E	1167 (1107, 1239)	Yes	
	780E	NC	No	Excessive mortality at all concentrations except control; suspect chemistry error[d]
	781E	>2500	No	No effect at highest conc; suspect chemistry error [d]
	782E	1790 (1618, 1916)	Yes	
	792E	1738 (1652, 1889)	Yes	
	798E	1690 (1594, 1774)	Yes	
	841E	1570 (1495, 1649)	Yes	

a NC: Not calculable. See ‘Comments’ column for reason.

b Values in parentheses represent 95% confidence limits.

c See text for discussion of ongoing problems associated with the *L. aequinoctialis* test protocol.

d See text for discussion

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. After the first two tests, a rapid decline in the EC_{50} was observed. Initially, this was attributed to an increase in sensitivity to uranium. However, over the course of the year, it became evident that the first two toxicity values were anomalous. It is unclear why these values were so much higher than the subsequent ones. They are still influencing the control/warning limits, however, this influence will decrease as more tests are completed.

H. viridissima

All four reference toxicity tests for *H. viridissima* were valid. There are no problems associated with this protocol.

M. mogurnda

The majority of reference toxicity tests for *M. mogurnda* were valid. Only two tests were invalid, one which resulted in excessive mortality across all treatment except the control, and the other which resulted in no effect at the highest concentration. In both cases, measured uranium data showed that errors in preparing the uranium test solutions almost certainly resulted in the unusual results.

Planned testing in 2007–08

The reference toxicity testing programs for *Chlorella* sp, *H. viridissima*, *L. aequinoctialis* and *M. mogurnda* will continue in 2007–08, with the aim of completing at least four tests per species. In addition, further testing will take place to try to resolve the issues of the *L. aequinoctialis* reference toxicity test protocol.

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Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer & A Esparon

Introduction

The principal emission to the atmosphere from the Ranger mine and mill are:

- radon from the mine pit, the ore stock piles and the waste rock dumps,
- dusts from the mine pit, the primary crusher and the calciner, and
- sulfur oxides from the acid plant and power station.

Dusts from the above sources are primarily a radiological hazard for workers and members of the public because of the relatively high activity concentrations of the long-lived nuclides of the uranium series. Similarly, the inhalation of radon and, in particular, the short-lived radon decay products (RDP) gives rise to radiation exposure. It has been shown that the greatest fraction of natural exposure of humans to radiation originates from the inhalation of radon decay products (Porstendörfer 1994).

Dust and radon are monitored by ERA on site and in Jabiru and Jabiru East and reported in their quarterly and annual *Radiation Protection and Atmospheric Monitoring Program* reports, respectively. ***eriss*** monitors the atmospheric radon and dust concentration at Jabiru and Jabiru East, as well as Four Gates Road radon station in the vicinity of the Mudginberri community (see atmospheric monitoring sites in Map 3).

The results of the atmospheric monitoring program are periodically compared with results of ERA's atmospheric radiological monitoring program. In addition, at Mudginberri Four Gates Rd Radon Station radon gas is measured continuously on a half hourly basis, 24/7, and data are downloaded bimonthly. This station is regarded as a radon background and reference site, and has continuously acquired radon data for the past 6 years.

Results

Radon pathway

Figure 1 shows the quarterly RDP data from Jabiru and Jabiru East measured by ***eriss*** from early 2002 to June 2007. Median RDP concentrations [$\mu J \cdot m^{-3}$] for 2002–2006 at Jabiru and Jabiru East are 0.039 and 0.064, respectively. Jabiru East data are generally higher and show more variation due to the vicinity of Jabiru East to the mine pit and ore stockpiles, which are 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 the annual cycle of wet and dry seasons, with superimposed diurnal variations. Airborne radon concentrations are generally lower during the wet season, as radon exhalation from the soil slows down with increasing soil moisture content. Other factors such as soil ^{226}Ra activity concentration, soil morphology, and vegetation cover influence the radon exhalation as well (Conen 2004, Tanner 1964, 1980) and have been investigated in a separate project (Lawrence et al submitted).

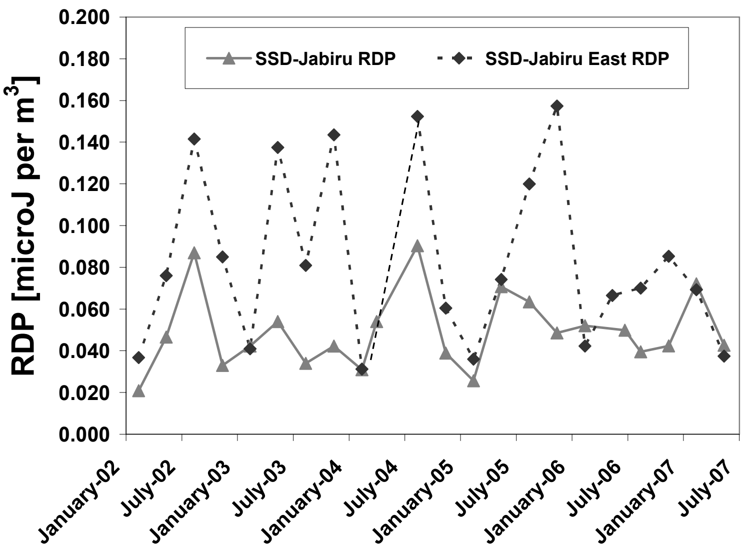

Figure 1 Radon decay product concentration measured by SSD at Jabiru and Jabiru East

Since the exposure due to naturally occurring RDP in the region is about 0.5–1 mSv per year, one of the challenges of determining the mine-related dose due to the inhalation of RDP is distinguishing between the mine-derived and natural background signals. Ranger uranium mine determine the mine origin RDP using a wind correlation model developed by ***eriss***, and subsequently calculate 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 (in brackets) at Jabiru and Jabiru East (ERA, 2007) and the calculated total annual doses. 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 $\mu J/hr/m^3$. The mine related dose calculated for 2006 is small and amounted to about 1% of the public dose constraint of 0.3 mSv per year from a single source.

Table 1 Average radon decay product concentrations at Jabiru, Jabiru East and Mudginberri, and associated total and mine derived annual doses received at Jabiru in 2004–2006

		2004	2005	2006
RDP concentration [$\mu J \cdot m^{-3}$]	Jabiru East	0.095 (0.103)	0.097 (0.097)	0.066 (0.071)
	Jabiru	0.063 (0.079)	0.052 (0.088)	0.046 (0.039)
	Mudginberri	0.048	0.054	0.075
Total annual dose [mSv] Jabiru		0.61 (0.76)	0.50 (0.85)	0.44 (0.38)
Mine derived dose [mSv] at Jabiru		0.014	0.037	0.003

Dust pathway

Atmospheric dust concentration, or long lived alpha activity (LLAA) concentration, is routinely monitored by both, ***eriss*** and ERA at the above monitoring sites. Figure 2 shows the long lived alpha activity at Jabiru and Jabiru East measured by ***eriss*** from early 2002 to June 2007. In 2007, permanent dust samplers were installed at Jabiru and Four Gates Rd radon station, to allow for permanent sampling of LLAA at the monitoring sites.

Similar to the variations in 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 vicinity to the mine. The median measured from 2002–2006 at Jabiru and Jabiru East amounts to 0.00013 and 0.00019 Bq·m^{-3}, respectively. The median over this period at Mudginberri amounts to 0.00012 Bq·m^{-3}.

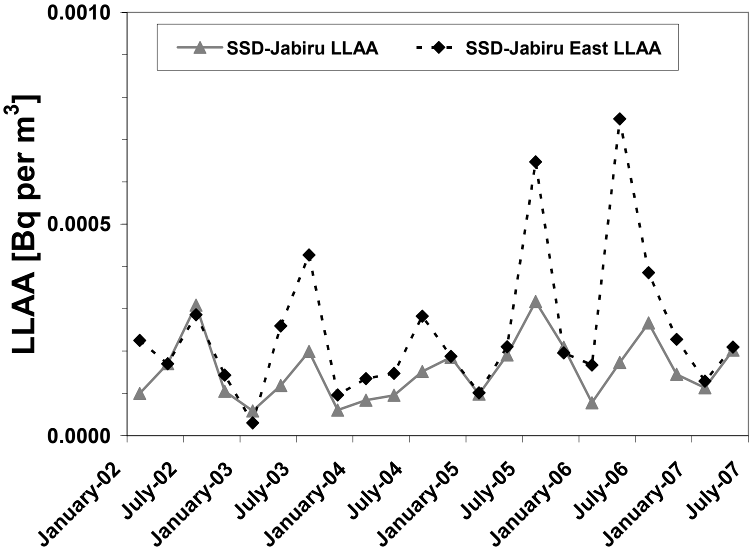

Figure 2 Long lived alpha activity concentration measured by SSD at Jabiru and Jabiru East

Table 2 gives the average LLAA concentration data measured by ***eriss*** and reported by ERA (ERA 2007; in brackets) for the past three years and the calculated annual doses from the inhalation of LLAA at Jabiru. Doses were 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).

Table 2 Average long lived alpha activity concentrations at Jabiru, Jabiru East and Mudginberri, and associated total annual doses received at Jabiru in 2004–2006

		2004	2005	2006
LLAA concentration [Bq·m^{-3}]	Jabiru East	0.00019 (0.00027)	0.00029 (0.00075)	0.00038 (0.00043)
	Jabiru	0.00013 (0.00011)	0.00020 (0.00014)	0.00017 (0.00033)
	Mudginberri	0.00015	0.00014	0.00014
Total annual dose [mSv] Jabiru		0.008 (0.011)	0.012 (0.031)	0.016 (0.018)

The mine contribution to the dust inhalation doses at Jabiru has been determined using stable lead isotope ratios and is much lower than at Jabiru East. Assuming a person living at Jabiru and working at Jabiru East, this person would receive an average annual dose from the inhalation of radiogenic dust originating from the Ranger mine of approximately 0.002 mSv (Bollhöfer et al 2006).

Steps for completion

The routine monitoring of dust and radon progeny is continuing at the thee monitoring sites in the Alligator Rivers Region, Jabiru, Mudginberri Four Gates Road Radon Station and Jabiru

East. In 2007, continuous RDP monitors will be acquired and tested, and permanently deployed at the Jabiru and Four Gates Road radon stations. This will enable ***eriss*** to acquire continuous RDP concentration data and allow simultaneous measurement of radon and decay product concentrations at Four Gates Road radon station. This data set will permit diurnal and annual variations of the equilibrium factor to be resolved.

It was also planned to deploy a radon monitor at the Koongarra deposit as part of the radon network in the Alligator Rivers Region. However, access to the site was not granted and this aspect of the project is on hold.

Summary

Monitoring of the radon and dust exposure pathways over the past 5 years has shown that the only significant contribution to radiological exposure of the public via inhalation is the inhalation of mine-derived radon decay products. Although the contribution is much less than the public dose constraint for a single source of 0.3 mSv per year and is of no concern, atmospheric monitoring continues to provide re-assurance to the public that exposure via inhalation of mine-derived radionuclides remains very low.

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Overview of SSD's stream monitoring program for Ranger, 2006–07

C Humphrey, B Ryan & D Jones

The SSD operates an integrated chemical (including radiological), physical and biological monitoring program to ensure protection of the aquatic ecosystems of the ARR from the operation of uranium mines in the region. This stream monitoring program is an independent, assurance program, unlike the compliance and check water chemistry monitoring programs of the mining company (Ranger) and the NT government regulator respectively.

The techniques and 'indicators' used in the monitoring program are underpinned by the outputs of research programs that have been carried out over many years. However, the program is not static, with ongoing improvements being made both as a result of more recently acquired knowledge and by the implementation of new technologies as these become available. In particular, the 2006–07 wet season represents only the second year for which continuous water quality parameters have been measured directly in Magela Creek.

The scope of the monitoring program satisfies two important needs of environmental protection: (i) the early detection of significant changes in measured indicators to avoid short or longer term ecologically important impacts; and (ii) assessing ecological or ecosystem-level effects by way of measured changes to surrogate indicators of biodiversity. The suite of monitoring techniques implemented by the SSD to meet these requirements is summarised below.

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

- *Water physico-chemistry*:
  - Spot measurements: includes pH, electrical conductivity (EC), suspended solids, uranium, magnesium, manganese and sulfate (weekly sampling during the wet season) and radium (fortnightly), and
  - 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 monitoring* of reproduction in freshwater snails and survival of fish fry (four-day tests conducted at fortnightly intervals) using creekside and in situ methods;
- *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 biennually in the late dry season).

(ii) Assessment of changes in biodiversity

- *Benthic macroinvertebrate communities* at stream sites (sampled at end of each wet season);
- *Fish communities in billabongs* (sampled at the end of each wet season).

The results from the stream monitoring program and monitoring support tasks and the outcomes of reviews of research programs are summarised below. The complete results for the water quality and biological monitoring programs are reported on the SSD web site (http://www.environment.gov.au/ssd/monitoring/magela-bio.html).

Chemical and physical monitoring of surface waters in Magela and Gulungul Creeks

J Brazier & K Turner

Routine weekly sampling program in Magela Creek

The water quality objectives for Magela Creek and the measures of success in meeting those objectives are provided in Iles (2004). Radium-226 activity concentrations in Magela Creek for 2006–07 are reported elsewhere ('Surface water radiological monitoring in the vicinity of Ranger and Jabiluka'). For other water chemistry analytes, the first samples for SSD's 2006–07 wet season surface water monitoring program were collected from Magela Creek on 14 December 2006. The creek was sampled weekly up to and including the 2 August 2007, with flow ceasing in the week thereafter.

Several distinct flow events influenced the water quality in Magela Creek in the 2006–07 wet season resulting in some peaks in the water quality parameters measured. However, the numerical water quality guidelines and limits set to protect the aquatic ecosystems downstream of the mine were not exceeded, indicating the ecosystem remained protected. Available biological monitoring data (described later in this section) also indicated that the environment remained protected throughout the season.

Between December and mid-January, Magela Creek had variable flow and at times almost ceased to flow due to lack of consistent rainfall. Magnesium (Figure 1) and sulfate levels at the downstream site for the first two weeks of flow, and again in February, were the highest observed in Magela Creek since SSD commenced its routine monitoring program in 2001. During the 1990s and up to 2001, similar levels (and higher) were consistently observed by ERA during routine monitoring. During all these periods, SSD's biological monitoring results showed no adverse effects.

Whenever 'high' magnesium (Figure 1) and sulfate concentrations, and consequently high electrical conductivity (EC), have been observed at the downstream site (as described above), this is always associated with a lack of corresponding increase in the same solutes at the upstream site, implying a mine-site contribution. In mid February 2007, an SSD investigation traced the major source of the magnesium and sulfate to Retention Pond 1 (via Coonjimba Billabong) by measuring EC in situ along the Magela Creek west channel, commencing from the downstream compliance site G8210009 and moving upstream until the EC source was found. Approximately 1.4 km upstream of G8210009, the EC in natural drainage entering the western channel was recorded at 420 µS/cm while only metres upstream of this point, the EC in the west channel decreased abruptly to 23 µS/cm. The drainage line was traced back to Coonjimba Billabong and its entry point to the west channel of Magela Creek. The point-source nature of the high conductivity drainage water to Magela Creek clearly indicates that Coonjimba Billabong was the source of the increased magnesium and sulfate at the downstream compliance site. A summary of the results from this investigation is shown in Figure 2.

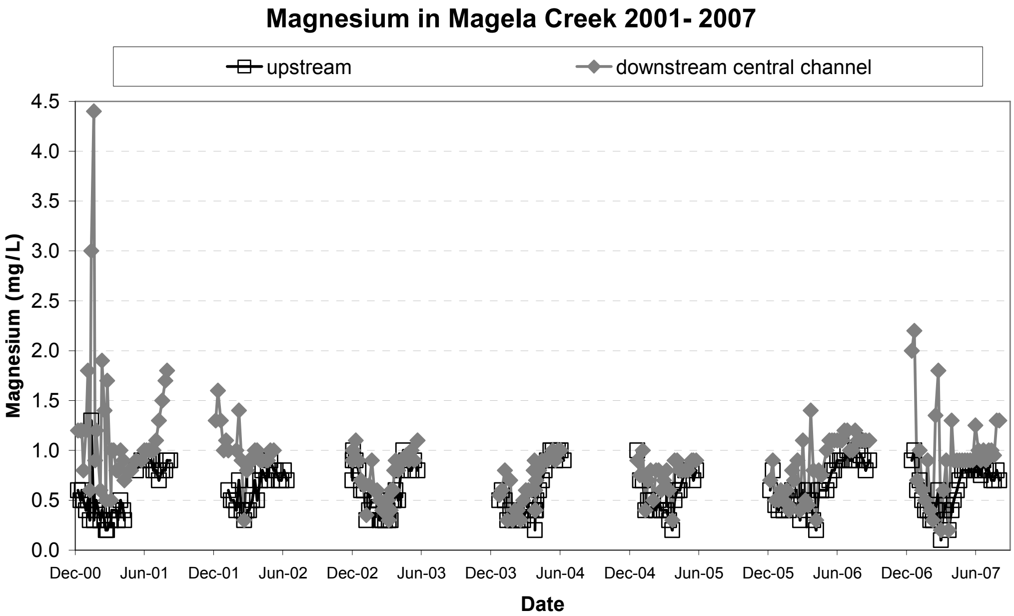

Figure 1 Magnesium concentrations in weekly grab samples (<0.45 µm filtered creek water) from Magela Creek 2001–2007 (SSD data)

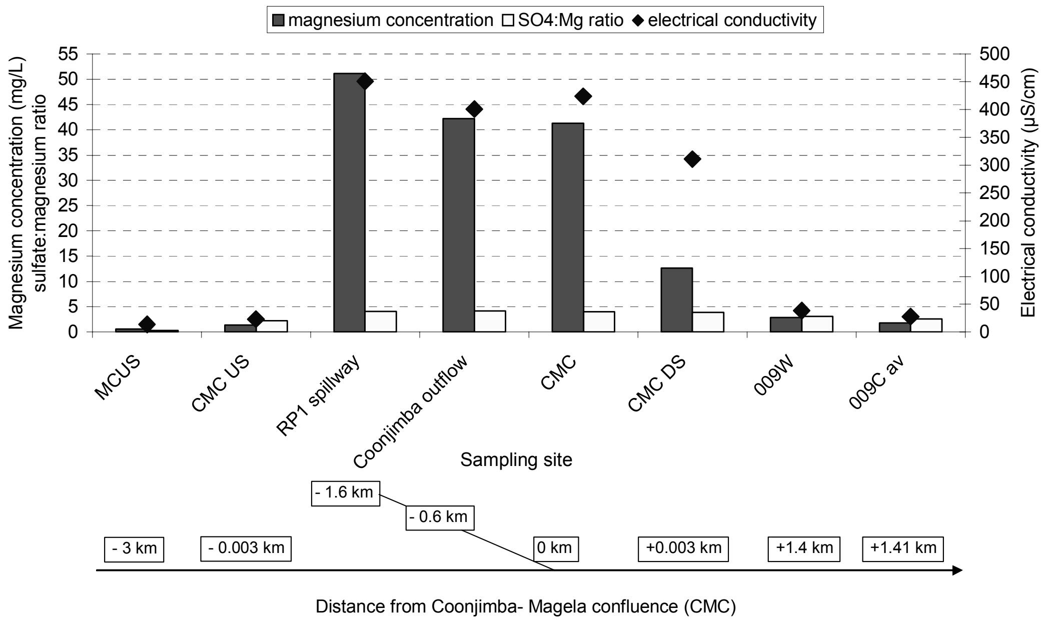

Figure 2 Tracking the source of elevated magnesium, sulfate and electrical conductivity in Magela Creek, February 2007. The distance trajectories depicted on the X axis represent Magela Creek (horizontal axis) and Coonjimba Creek (oblique axis) with direction of stream flow indicated by arrow. Site codes and description are as follows: MCUS is the statutory Magela upstream compliance site; 009C is the statutory Magela downstream compliance site (central channel); 009W is adjacent to 009C, but in the west channel; CMC is the Magela-Coonjimba drainage confluence point, where elevated Mg, SO_4 and EC were observed to drain into the Magela Creek west channel; CMC US and CMC DS[1] were sampling sites in the Magela Creek west channel, a few metres upstream and downstream of CMC respectively.

[1] The discrepancy between field-measured EC and laboratory-measured Mg at CMC DS is presumably an artefact of very poor mixing of waters at this site, located on the Magela-Coonjimba confluence. Separately collected or measured samples, in this case, can vary considerably in water quality.

In addition to confirming the source of the magnesium and sulfate, an assessment was made as to whether the increased concentration in salts represented an increase in solute load to Magela from the mine site or was instead a result of reduced dilution afforded by Magela Creek at low creek flows. Solute loads (based on continuous EC data as a surrogate) for the corresponding 2005–06 and 2006–07 wet season periods were compared (see section below, 'Continuous Monitoring of Water Quality', for discussion of EC and solute relationships). This showed that the solute load to the creek was not higher than the comparable period of the previous 2005–06 wet season. The increased magnesium and sulfate concentrations at that time of the season were likely a result of decreased water volume in Magela Creek (compared to previous years) allowing for less dilution of incoming salts. Later in the 2006–07 season, however, although the magnesium concentrations were lower, the magnesium load was actually higher (see 'Continuous monitoring of water quality in Magela Creek' description below for details).

During a five day period commencing on 27 February 2007, the Magela catchment (as well as the Gulungul and Ngarradj catchments) experienced record rainfall, with the gauging stations being covered by rising water at both the upstream and downstream sites. The discharge measured over the three day period corresponded to the highest flood levels recorded in Magela Creek since recording began in 1971. The Ranger mine shut down its mining and milling operations during this high rainfall period and concentrated on its on-site water management. SSD's data showed that there were no guideline or limit exceedances for any of the variables in the water samples collected during this period.

The highest uranium concentration measured by SSD during this high rainfall period was 0.15 µg/L on 8 March which is about 2.5% of the limit value of 6 µg/L (Figure 3). ERA increased their sampling frequency during and after this intense rainfall period and recorded a uranium concentration of 1.21 µg/L (~20% of the limit) on 9 March at the downstream compliance point. This higher uranium value also coincided with a peak in EC of 30 µS/cm recorded by the SSD continuous monitoring sonde. Since the uranium value was in excess of the 'Action trigger', the stakeholder-agreed value beyond which an investigation or contingency plan must be instigated, increased management action was required by ERA. However, such actions were already being undertaken by way of contingency water management, increased monitoring in the creek and investigations of the water quality in various on-site waterbodies.

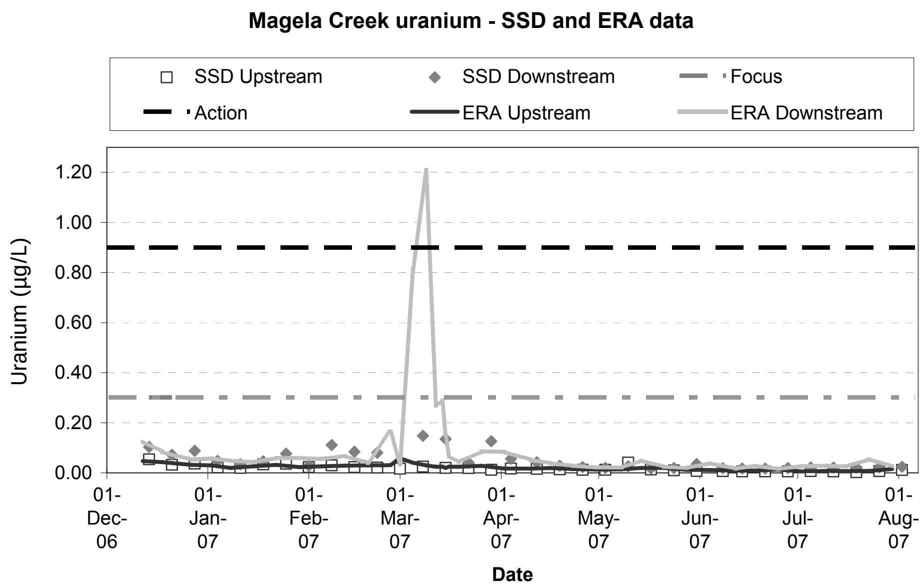

Figure 3 Uranium concentrations measured in Magela Creek by SSD and ERA between Dec 2006 and June 2007

There is historical precedent for uranium concentrations in Magela Creek reaching around 1 µg/L. Values close to (or greater than) 1 µg/L were recorded in Magela Creek prior to the remediation works conducted by ERA in 2000 to improve the water quality of RP1 (Figure 4).

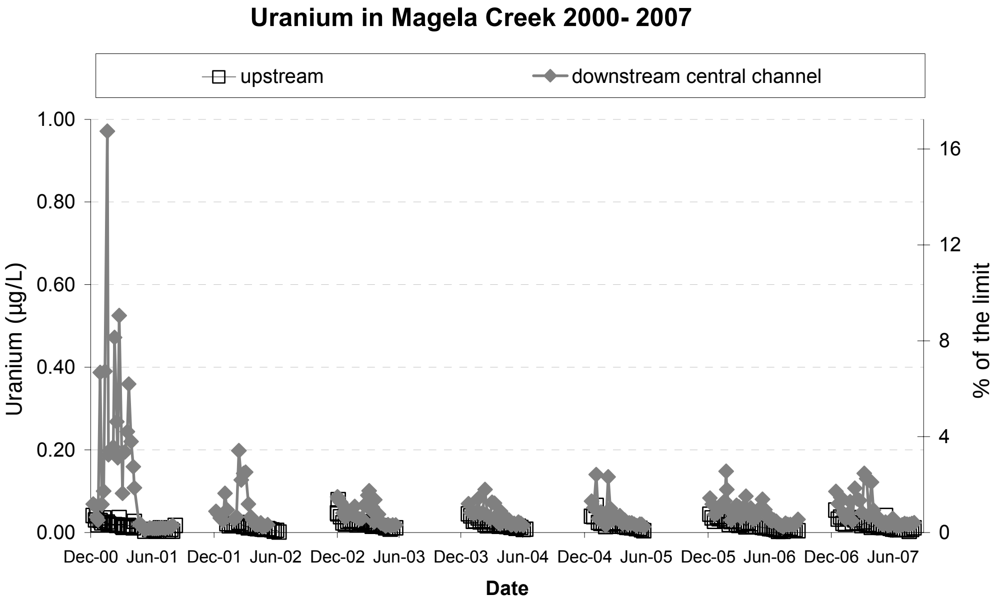

Figure 4 Uranium concentrations in Magela Creek since the 2000–01 wet season (SSD data)

The uranium concentrations measured by both SSD and ERA since the flood episode have been within the usual improved uranium concentration range observed since 2000 (Figure 3).

Chemical and physical monitoring of Gulungul Creek

The first water chemistry samples for SSD's 2006–07 wet season surface water monitoring program were collected from Gulungul Creek on 4 January 2007 – the first week after continuous surface water flow was observed in the creek. Weekly sampling continued throughout the season until 28 June 2007 when the last samples were collected. On 4 July 2007, key stakeholders agreed that continuous surface flow had ceased in Gulungul Creek and monitoring of the creek was no longer required.

Overall, the water quality in Gulungul Creek was good during the 2006–07 wet season, providing reassurance that the aquatic environment of the creek remained protected from impacts of mining activities at the Ranger mine site. Available biological monitoring data for the creek (described later in this section) also indicate that the environment remained protected throughout the season.

Between 27 February and 2 March 2007, the Gulungul Creek catchment (like the Magela and Ngarradj catchments) experienced an extreme rainfall event resulting in overbank flow at the upstream and the downstream sites. SSD weekly monitoring data for the period (not shown in this report, but available at www.environment.gov.au/ssd/monitoring/gulungul-chem.html) showed that turbidity increased at both the upstream and downstream sites consistent with the increase in stream flow, while concentrations of most other variables were similar at the upstream and downstream sites. However, uranium, although only just above 0.2 µg/L (around 4% of the ecotoxicity based limit) and within the range measured in previous years, remained higher downstream than upstream (Figures 5 & 6). Comparisons of ERA and SSD uranium data in Figure 5 show that the ERA monitoring program captured more elevated uranium events than the SSD dataset during and immediately after the flooding. The peak concentration of 0.46 µg/L measured by ERA during the flood period was less than 8% of the ecotoxicity based limit. For most of the season, apart from the flood event, ERA and SSD data were in good agreement (Figure 5).

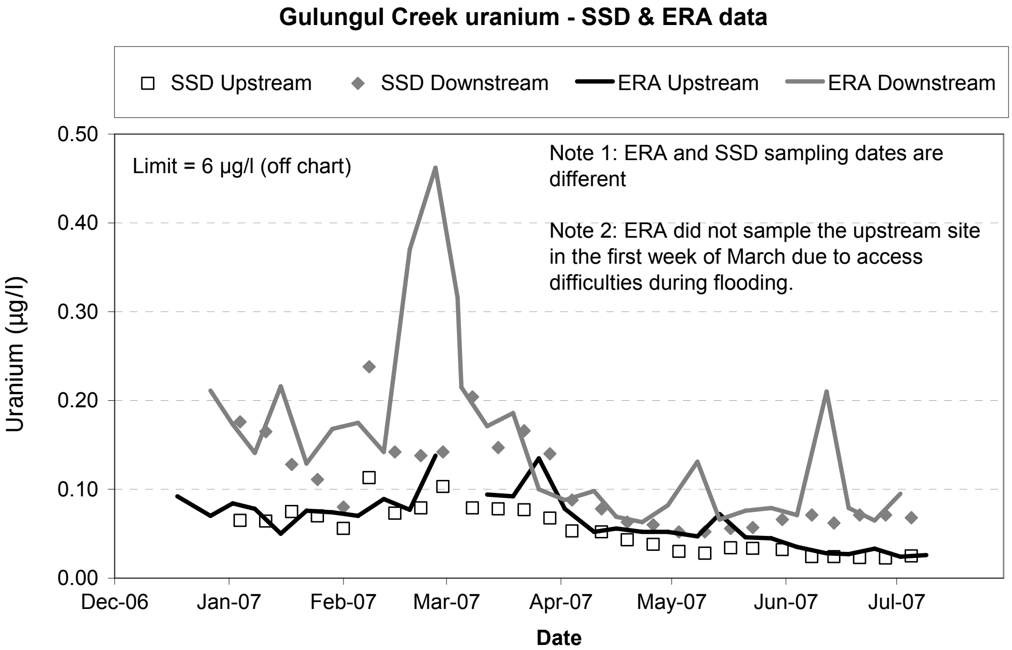

Figure 5 Uranium concentrations measured in Gulungul Creek by SSD and ERA during the 2006–07 wet season

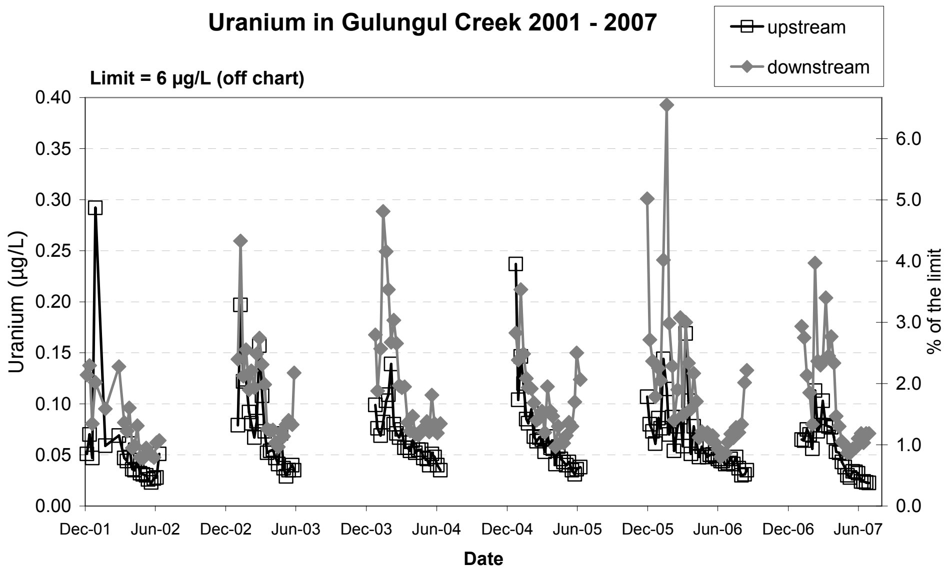

Figure 6 Uranium concentrations in Gulungul Creek between 2000 and 2007 (SSD data)

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Continuous monitoring of water quality in Magela Creek

K Turner, D Moliere, C Humphrey & D Jones

The routine water quality monitoring program carried out by SSD since 2001 involves collecting and analysing weekly grab samples to ensure that water quality indicators, including uranium, remain within guideline values derived based on the recommendations of ANZECC & ARMCANZ (2000) water quality guidelines for protection of 99% of species. However, this sampling regime could miss transient system fluctuations caused by natural or mine site inputs.

As part of its ongoing research to implement best practice technology for enhancing the effectiveness and efficiency of monitoring water quality downstream of mine sites, SSD has been evaluating the use of in situ continuous measurement of water quality parameters using sensors (Datasondes) located in Magela Creek upstream and downstream of the Ranger mine site. These sites are close to the existing statutory MCUS and G8210009 monitoring sites, respectively. In the event that this method is found to be sufficiently robust, then it may ultimately be introduced as one of the tools used for compliance monitoring of water quality in Magela Creek.

As well as providing improved capacity for early detection of mining impacts, the continuous measurement of key water quality indicators, such as electrical conductivity and turbidity, provides a more complete description of the overall water quality of the system by capturing transient fluctuations in water quality that would be undetected by the weekly grab sampling regime. Furthermore, these key water quality indicators can be used as indirect, surrogate measures of some water quality characteristics where strong correlations between the two exist. Examples of this use include the use of turbidity to measure mud (silt and clay fraction of suspended sediment) concentration, as reported by Moliere et al (2005). This type of surrogate measurement provides a basis for using the continuous EC data for deriving solute loads.

Over the 2005–06 and 2006–07 wet seasons, in situ water quality data were collected at 10–20 minutes intervals using a network of ***eriss*** and ERA in situ sensors located at key sites in Magela Creek, as well as Ranger mine site tributaries, Coonjimba Creek (including RP1) and Corridor Creek. These data have been reported previously (Turner 2007, Turner et al 2008). One upstream and two downstream continuous monitoring Datasondes were deployed in Magela Creek. The upstream sonde was located approximately 0.5 km downstream of the current Magela upstream statutory monitoring site (MCUS), but still upstream of mine surface water influence. The downstream sondes were located approximately 200 m downstream of the Magela downstream compliance site (G8210009), on either side of the western-most channel, referred to as 009E and 009W. Corresponding stream flow (discharge) data were collected from G8210009, a gauging station on Magela Creek located downstream of the Ranger mine.

Accompanying the continuous water quality monitoring, surface water samples were collected from various sites upstream and downstream of Ranger mine and analysed for solute concentrations. These data, along with concurrent EC measurements, were used to derive statistically significant relationships between magnesium (Mg) concentration and EC (Figure 1). Different EC-Mg relationships were derived for the upstream and downstream sites as a result of widely differing EC/Mg ranges at each of the sites – a very wide range of values at the downstream site as a consequence of variable mine water inputs (of $MgSO_4$), and a comparatively narrow range upstream.

The high correlations observed in these relationships (Figure 1), indicate that Mg concentrations in Magela Creek can be continuously monitored by using the primary EC data. However, it is important to note that the relationships shown in Figure 1 are preliminary at this stage, pending acquisition of data from at least one more wet season. In particular, the Mg-EC relationship lacks data for higher values in the range depicted, a deficiency that will be redressed during the 2007–08 wet season. The preliminary Mg-EC relationships were applied to the continuous EC record to predict upstream and downstream Mg concentrations over each wet season; the 2006–07 traces are shown in Figure 2.

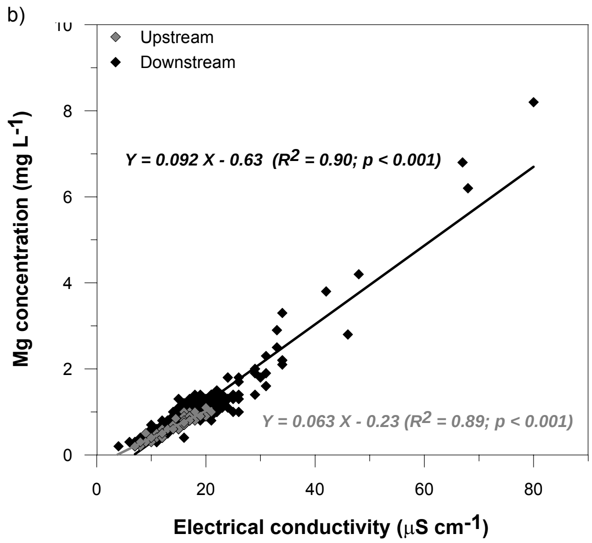


Figure 1 Relationships between Mg concentration and EC for upstream and downstream sites on Magela Creek

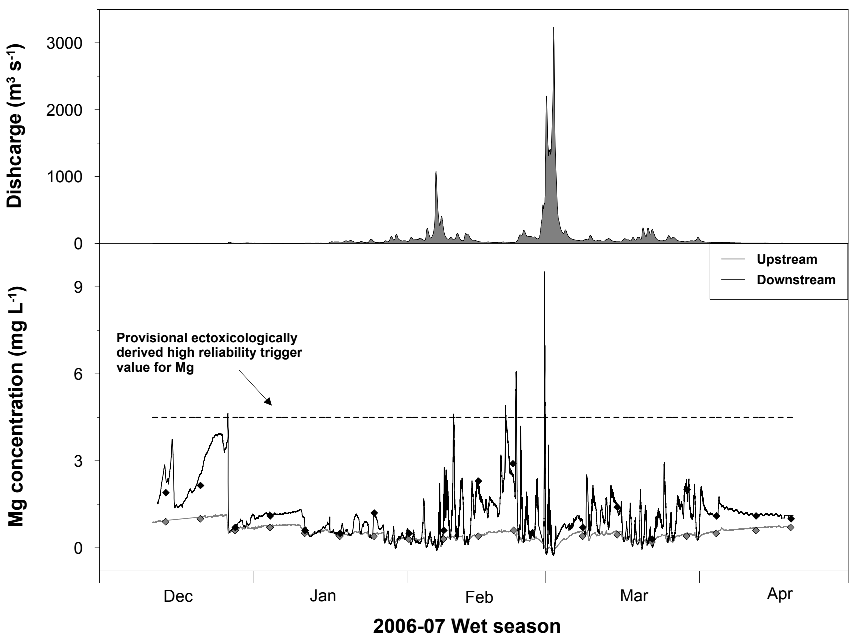


Figure 2 Predicted continuous Mg concentration traces (lines) along with grab sample values (diamond points) measured upstream and downstream of Ranger mine, together with Magela Creek discharge measured at gauging station GS8210009, for the 2006–07 wet season

The predicted Mg data illustrate transiently-elevated levels of Mg that pass undetected by the less frequent, routine water chemistry monitoring (Figure 2). On five occasions during the 2006–07 wet season (in late December and during February), the estimated Mg concentrations exceeded the current provisional ecotoxicologically-derived, high-reliability trigger value for Mg of 4.6 mg/L (van Dam et al 2006, 2007) for durations between 0.5 and 3.5 hours (Figure 2). Mg concentrations were estimated to be as high as 9.5 mg/L in Magela Creek downstream of Ranger mine at this time. Unfortunately, no biological (toxicity) monitoring tests were conducted in the periods of spiking in Mg concentrations, though the cumulative effects of these spikes, at least, were not sufficient to have adversely affected benthic macroinvertebrate communities at G8210009 (Humphrey et al 2008), a biological response measured at the end of the wet season and integrating all wet season water quality events.

The predicted Mg concentration data were used to derive the loads of Mg transported along Magela Creek during the 2005–06 and 2006–07 wet seasons. Magnesium load was 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) and Q is instantaneous discharge (L/s):

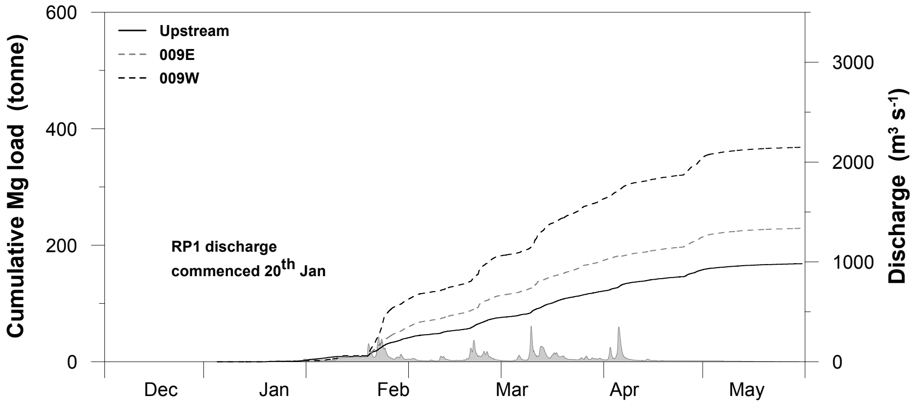
$$\text{total load} = \int_{t=0}^{t=i} [Mg]\; Q \,\mathrm{dt} \qquad (1)$$

By multiplying magnesium 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 wet season can be calculated. Figure 3 shows cumulative loads upstream and downstream of Ranger mine, along with the annual hydrograph measured at G8210009 over each of the two wet seasons. The effect of input of mine-derived waters on Mg loads is apparent in Figure 9, with the downstream loads deviating upwards from the background (upstream) load in each season, concurrently with the start of discharge from RP1.

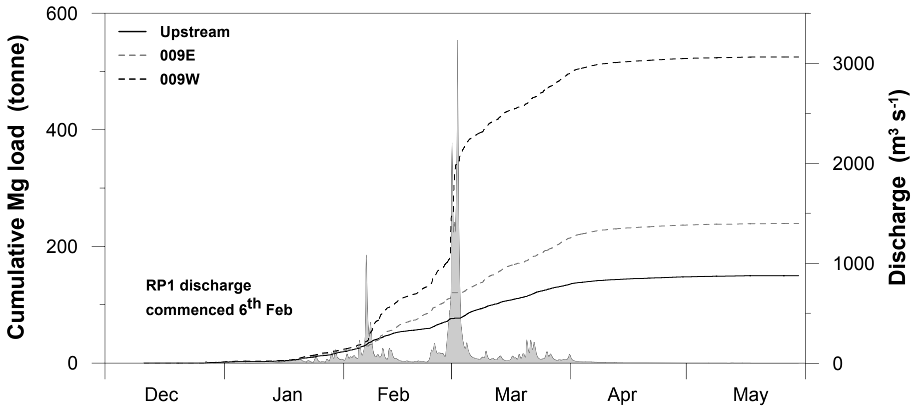
The higher load values estimated for the western side of the western channel (009W) (Figure 3) reflect the fact that mine inputs to the creek, via surface runoff from the Ranger Land Application Areas (LAAs) and water discharged from RP1, are made on the western side of Magela Creek and that there is incomplete lateral mixing by the time these waters reach the downstream site.

Between 27 February and 2 March 2007, the Magela catchment experienced an extreme rainfall event resulting in the highest flood levels in Magela Creek since recording began in 1971 (Moliere et al 2008). Table 1 shows that significant solute input to Magela Creek was associated with this event. As discussed elsewhere (above and below), differences in the loads estimated for each of the downstream, western-channel locations reflect the lack of mixing of mine waters at this creek site. The loads for the western side of this channel, in particular (009W), would overestimate total load for Magela Creek and hence should not be used for this purpose.

Table 1 also shows the estimated net Mg load for Magela Creek (downstream annual load minus upstream annual load) for both seasons, representing the mass of mine-derived Mg contributed to Magela Creek. The net load is substantially higher for the 2006–07 wet season, reflecting the significant leaching of Mg from the operational area during high rainfall events in this season.



2005-06 Wet season



2006-07 Wet season

Figure 3 Cumulative Mg loads estimated from continuous EC data measured upstream and downstream of Ranger mine. Magela Creek discharge was measured at gauging station G8210009 over the 2005–06 and 2006–07 wet seasons

Table 1 Total monthly and annual Mg load (tonne) estimated at the continuous monitoring stations on Magela Creek over the 2005–06 and 2006–07 wet seasons

	2005–06 wet season (5/12/2005 to 30/6/2007)			2006–07 wet season (11/12/2006 to 6/6/2007)		
Month	MCUS	009E	009W	MCUS	009E	009W
Dec	2.50	1.67	2.00	1.71	1.87	2.12
Jan	38.8	58.3	105	17.8	18.9	21.5
Feb	34.7	54.0	74.5	56.7	99.9	228
Mar	45.4	60.0	97.6	59.4	94.2	246
Apr	36.6	40.0	70.4	12.2	21.8	24.9
May	10.3	15.0	18.2	1.97	2.46	2.61
Jun	1.05	1.23	1.36	0.00	0.00	0.00
27 Feb – 2 Mar (flood period)				7.09	16.9	182
Total annual load	169	230	370	150	239	525
Net load		61.0	200		89.3	375

Future work and analysis of data

The total loads of mine-derived solutes estimated in Magela Creek will be compared with data collected on the Ranger site to quantify and compare the relative contributions arising from point and diffuse sources.

Solute input associated with discharge from RP1 and via GC2 in the Georgetown Creek catchment can be quantified by deriving relationships between EC and solute concentrations measured at these sites, and using ERA's continuous measurements of EC at these locations to estimate loads moving down these catchment lines through the wet season.

The solute load added to each of the land application areas from the irrigation of pond water during the dry season can be quantified using the volume of applied pond water and the concentrations of solutes contained within it.

Comparing the net mass of solutes input by irrigation of pond water, the mass of solutes being exported to Magela Creek along the RP1 and Georgetown Creek catchment lines, and the total mass of solutes being recorded downstream of the mine at 009 in Magela Creek will enable a dynamic assessment of the intra- and inter-seasonal fluxes of salts in the system.

It must be emphasised that load estimates for Mg downstream of Ranger are approximate at this stage. Factors contributing to uncertainties in the estimates include:

- the creek discharge data used to calculate the loads pertain to the total cross-section of the creek at G8210009, whereas the predicted Mg concentration data apply only to the western channel of the creek. The concentrations and loads of mine-derived solutes in the other creek channels at this location are unknown, though they will be considerably less than those estimated for the western channel that more directly receives these solutes. As a consequence, the load values reported here are over-estimates of the total solute load in Magela. Improved knowledge of the mixing characteristics of mine waters in Magela Creek across the cross-section of the creek at this site will be required to improve the accuracy of the estimate.[1]
- No EC data were recorded for a period of 12 days in early January 2006 (approximately 8% of the annual runoff occurred during this period) and, therefore, the annual Mg load for 2005–06 is underestimated.
- Unverified discharge data have been used in this report as these were all that were available at the time of reporting. The loads will be amended after any corrections are made to the data.

Further work will also entail investigating possible relationships between EC and other solutes associated with Ranger mine, including calcium. Contaminant transport by suspended sediments will also be investigated during the 2007–08 wet season by analysing water samples collected during discharge events for contaminants associated with the fine suspended sediment (mud) fraction. If robust correlations are found between the suspended sediment concentration of a given metal (eg uranium) and turbidity, then continuous turbidity data may be able to be used to derive the annual loads of the metal in the suspended sediment fraction.

With further refinement, the results from this work will be used to examine temporal and spatial variations in Mg loads and to assess changes in solute transport characteristics over

[1] Separately, providing the concentrations of Mg measured (and inferred) in the western channel are below the ecotoxicity-derived trigger values, then the aquatic ecosystem in Magela Creek will remain protected.

different flow regimes. These results may then be applied to assessment of mine site rehabilitation scenarios.

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Toxicity monitoring in Magela Creek

C Humphrey, C Davies & D Buckle

Monitoring results for 2006–07

In this form of monitoring, the effects of water discharged from the Ranger site are evaluated using responses of aquatic animals held in flow-through tanks located on the creek bank (creekside monitoring) or in the creek itself (in situ monitoring). Creekside monitoring has provided the primary means for toxicity monitoring since such testing commenced in the 1991–92 wet season and can be overviewed in the Supervising Scientist's Annual Report for 2001–02 or on the web (http://www.environment.gov.au/ssd/monitoring/magela-bio.html). The 2006–07 work program compared egg production in freshwater snails (*Amerianna cumingi*) upstream of the mine site (control site) and from the creek just below gauging station GS8210009, some 5 km downstream of the mine (test site). At each of the two sites, duplicate containers hold replicate (8) snail pairs (thus 16 pairs of snails exposed per site). In previous wet seasons, testing has also been conducted using the survival response of larval fishes (*Melanotaenia nigrans*). This larval fish test was not included in the 2006–07 wet season because of the intensive (and competing) resources required for the comparative evaluation of the snail reproduction response using creekside and in situ methods (see below).

From the commencement of testing on 1 January 2007, four comparative four-day tests were conducted, using creekside and in situ techniques, every-other-week. After the extreme late February – early March flooding, the creekside infrastructure was too badly damaged to enable this form of monitoring to continue. Testing resumed using the in situ method only, and three more tests were completed in this manner for the remainder of the 2006–07 wet season (see below).

The results of the four creekside tests (1–5 January 2007, 15–19 January 2007, 29 January – 2 February 2008 and 12–16 February 2008) and three subsequent in situ tests (12–16 March 2008, 26–30 March 2008 and 9–13 April 2008) for snail egg production are plotted as part of a continuous time series of actual and 'difference' data in Figure 1. Descriptions of the sources of creekside data and data quality issues are provided in the Supervising Scientist's Annual Report for 2001–02 and on the SSD's website (www.environment.gov.au/ssd/monitoring/magela-bio.html).

Snail egg production at the upstream and downstream sites was generally similar across all four creekside tests, as well as the three subsequent in situ tests, and resembles the pattern of egg production observed in previous wet seasons. The exception to this is the discrepancy in egg production observed between the two sites in the first creekside test for the season (Figure 1), giving rise to the largest 'difference' value ever recorded over the 16 wet-season history of creekside monitoring. Parallel creekside and in situ snail egg production results for the first four tests of 2006–07 are shown in Figure 2. While the magnitude of egg production at both sites was greater in the in situ tests, a similar pattern of egg production response was observed to that shown in the creekside results. In the first in situ test conducted for 2006–07, there was no discrepancy observed in egg production between the two test sites. Considering the similar pattern in egg production observed between the two treatment types (creekside and in situ) in 2006–07, it would appear that snail egg production at the downstream creekside station was anomalously high during the first test. While creek water levels were quite low during the first test for 2006–07, corresponding spot water chemistry data collected during this test as part of the SSD's routine monitoring program (reported above) do not indicate any significant elevation of analytes at this site nor any atypical discrepancies between the two creek sites that might explain the unusual creekside result.

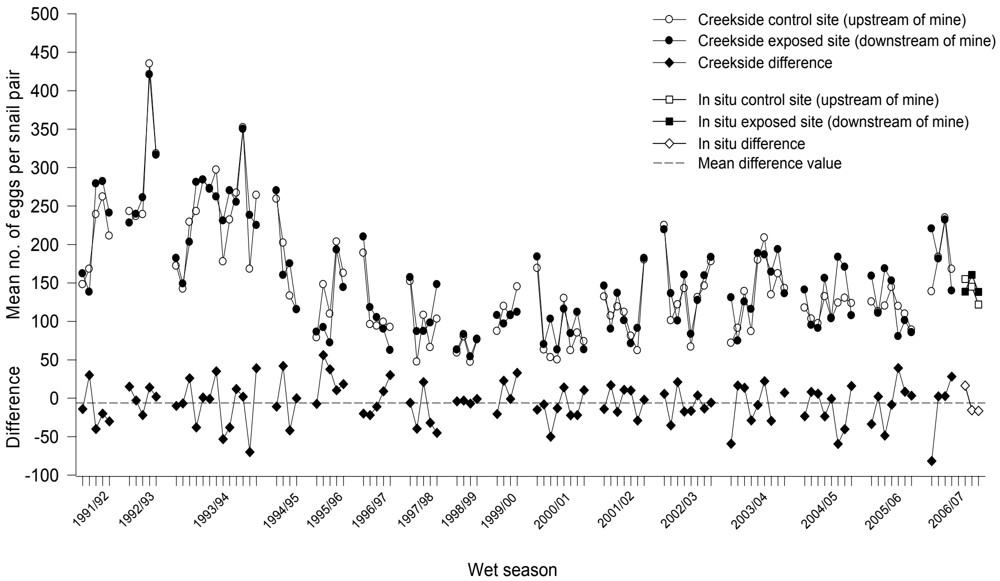

Figure 1 Creekside monitoring results for freshwater snail egg production for wet seasons between 1992 and 2007. The final 3 data points are the results from the in situ daily-feed treatment only because creekside monitoring ceased after flooding damaged infrastructure.

Thus, the increased snail egg production observed at the downstream creekside station during the first test of the 2006–07 wet season does not appear to be mine-related. From the collective creekside and in situ results conducted during the 2006–07 wet season on freshwater snails, it is concluded that no adverse effects of water discharged from the Ranger mine site to Magela Creek were observed – at least for the fortnightly periods in which tests were conducted. Unfortunately, none of the toxicity monitoring tests coincided with the periods of spiking in Mg concentrations that occurred during the 2006–07 wet season (and reported above by Turner et al 2008) that would elucidate further the ecological effects of these events.

Development of in situ methods

The mainstay of the biologically-based toxicity monitoring approach since 1991 has been creekside monitoring, described above. There are a number of practical issues with this method, including high staff resourcing demands, reliance on powered pumping systems (in an area of high electrical storm activity) and vulnerability to extreme flood events.

The problems associated with the ex situ creekside program have led to an evaluation of the viability of in situ testing – deploying floating containers in Magela Creek itself, using the same test organisms currently used for the creekside monitoring program. Potential technical advantages of this method include improved water flow-through and contact conditions for the test organisms, portability, and the ability to run an essentially continuous biological monitoring program. In addition the reduced staff resourcing needs means that more staff time will be available for other components of the monitoring program and for interpretation of the data. These advantages make the in situ method appealing for future monitoring at Ranger and, potentially, also for use at other mine sites in the Northern Territory and elsewhere.

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) clearly highlights the benefit of a method that is not reliant on complex, expensive and vulnerable infrastructure.

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). 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 baseline creekside monitoring, test data have been obtained since 1991–92 using the established creekside protocols and infrastructure. Thus, it is critical to ensure that the proposed in situ method yields comparable results before it can be phased in as the sole procedure in the future.

A three-year period was originally set aside for assessment of in situ testing, including method development. Work commenced in the 2005–06 wet season and continued in 2006–07. Concurrent creekside and in situ tests were run during 2006–07 until the early March flood severely damaged creekside infrastructure. After this, only the in situ method could be used for the rest of the wet season. Concurrent testing will be carried out again for the 2007–08 wet season after which the need for any further comparative testing will be reviewed. Should similar results between the in situ and creekside tests be obtained for a sufficient number of test runs this would provide sound technical grounds to discontinue the creekside monitoring work in favour of in situ monitoring only.

Reproductive output (egg production) in the freshwater snail, *Amerianna cumingi*, has been the main focus for in situ evaluation. Survival of black-banded rainbowfish (*Melanotaenia nigrans*) larvae has also been employed in toxicity monitoring in previous wet seasons. However as discussed above, the larval fish test was not included in the 2006–07 wet season evaluation because of the intensive (and competing) resources required for the comparative evaluation of the snail reproduction. Moreover, the fish larvae response is much less toxicologically sensitive than the snail response to uranium (van Dam et al 2007), and suffers inherent natural high mortality rates at the upstream control site. While some limited fish larvae testing has been conducted, this is confined to the upstream site only (results not reported here).

A potentially important variable for the in situ method is the nature and frequency of feeding of the deployed snails. At both the upstream and downstream locations, two feeding treatments have been 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 creekside monitoring, this would lead to a significant reduction in the resources necessary to run in situ monitoring.

For each feeding treatment and at each of the upstream and downstream sites, in common with creekside testing, there are duplicate containers each holding replicate (8) snail pairs (thus 16 pairs of snails exposed per site).

Results of the in situ feeding methods evaluation are shown in Figure 2.

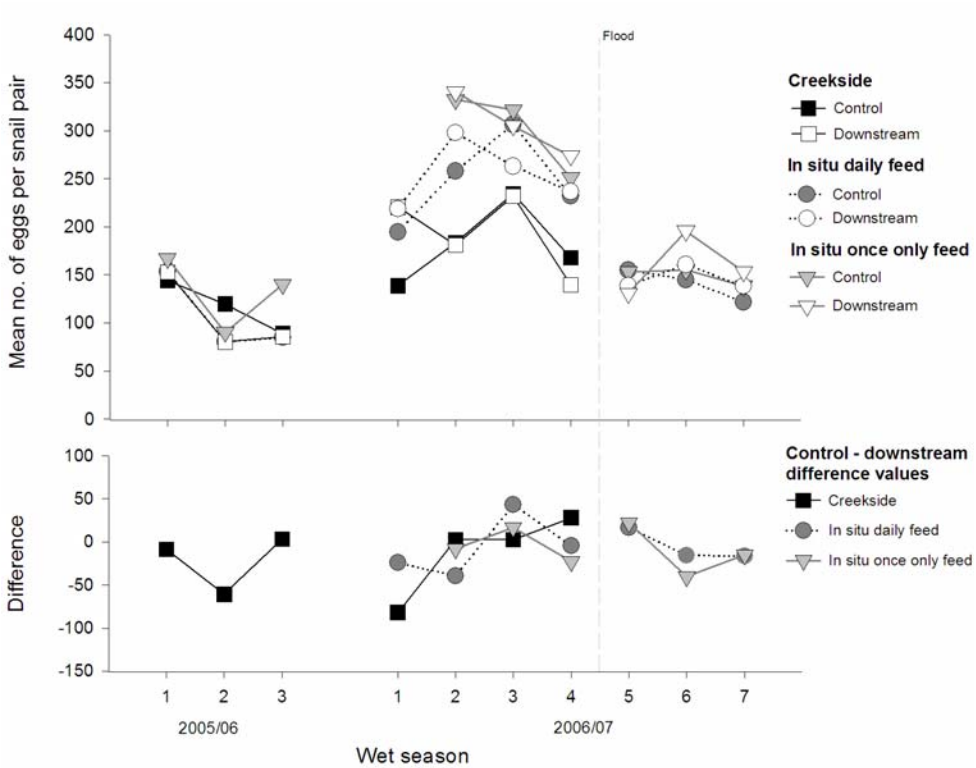

Figure 2 Comparison of snail egg production for creekside monitoring and in situ monitoring, 2005–06 and 2006–07 wet seasons (note: the once-only feeding treatment was not included in the first test of 2006–07). In 2005–06, in situ testing was confined to the control site.

While there was no statistically significant difference between the mean egg production of the two feeding treatments across the seven comparative tests (Student's *t*-test, P = 0.431), it is evident that egg production in the once-only feeding regime is consistently higher than egg production in the daily feeding regime. This may be due to the absence of any ongoing physical disturbance from workers to snails fed once only. This issue will be investigated further.

To date, results from in situ toxicity monitoring are very encouraging in terms of concordance of snail egg production between upstream and downstream sites, and the similarity of results with corresponding creekside results (Figure 1). While the magnitude in egg production may be greater in the in situ test containers, the important test end-point is the corresponding upstream-downstream difference values. To date these difference values are similar between the creekside and in situ test methods.

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Bioaccumulation in fish and freshwater mussels from Mudginberri Billabong

J Brazier, B Ryan, C Humphrey & A Bollhöfer

Background

Mudginberri Billabong is the first major, permanent waterbody on Magela Creek 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. Thus, 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 reach limits which, in principle, might harm the organisms themselves and so the bioaccumulation monitoring program serves an ecosystem protection role in addition to the human health aspect.

Mussel bioaccumulation data were obtained from Mudginberri Billabong intermittently from 1980 to 1999. Since 2000 there has been regular (annual) sampling from Mudginberri and from 2002 a control site (Sandy Billabong, Nourlangie Creek catchment). Uranium and radium data for the period since 1980 are discussed below.

Bioaccumulation of uranium and radium in freshwater mussels

Uranium concentrations in freshwater mussels from Mudginberri and Sandy (post 2000) Billabongs are shown in Figure 1, together with U concentrations in water and sediment samples collected concurrently. Data for mussels from the early to mid 1980s may suffer from QA/QC problems (specifically relating to adventitious contamination of samples by metals) arising from the outsourcing of chemical analyses or from within the laboratories of ERA that processed the samples at this time (Jones 2005). 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 or with mussel age (the latter a feature of radium concentrations in mussel soft tissues, see discussion below). Uranium in mussels has been 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 U in mussel tissues through time, with essentially constant levels observed between 1989 and 1995, and consistently low levels from 2000 to the last sample taken in May 2007, indicates absence of any mining influence. The decrease in U concentrations in water, sediment and mussels after 2000 in Mudginberri Billabong may be an artefact of changes in the sampling regime or analysis methods.

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 seen to be 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 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). The need to better characterise sediment is now recognised and a more extensive and refined sediment sampling and size fractionation protocol was used for sampling following the 2006–07 wet season (results currently being analysed).

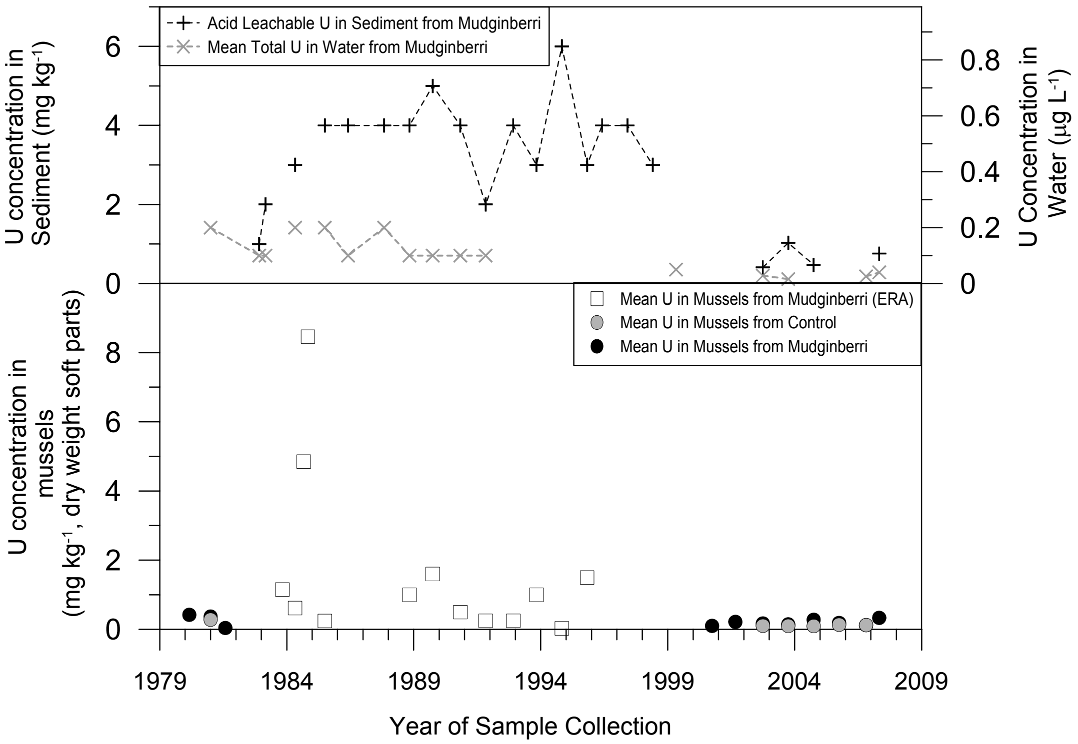

Figure 1 Mean concentrations of U measured in mussel soft-parts, sediment and water samples collected from Mudginberri Billabong and control billabongs since 1979

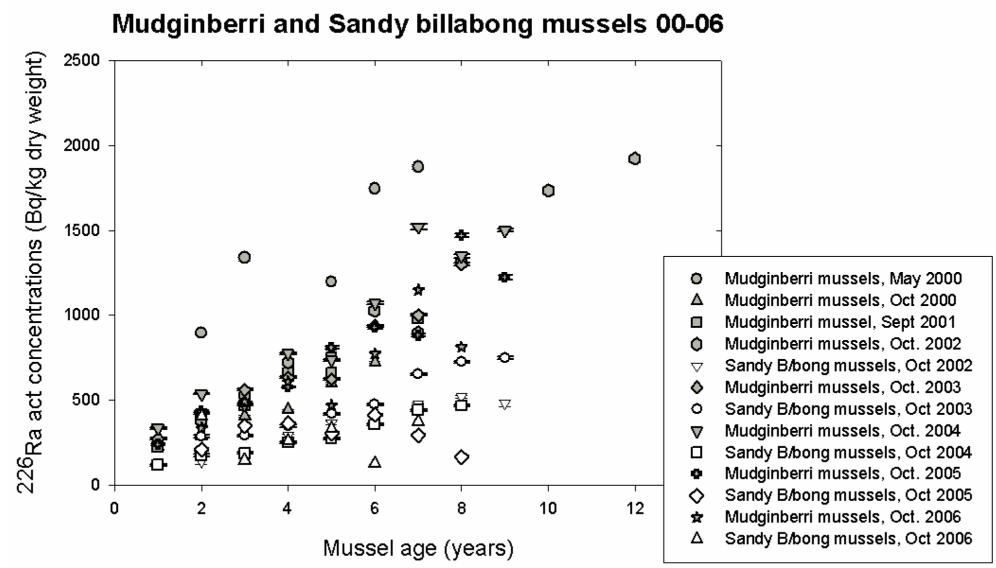

Figure 2 ^{226}Ra activity concentrations in the flesh of freshwater mussels collected from Mudginberri Billabong 2000–2006 and Sandy Billabong 2002–2006

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 statistical methods to determine differences in regression relationships (from Figure 2) will be explored as a means for quantifying any such future change.

Bioaccumulation of uranium and radium in mussels in Magela creek catchment

In October 2006, a review of the SSD biological monitoring program recommended initiating a study to determine whether the increased radium uptake in Mudginberri Billabong (Magela Creek catchment) compared to the control site, Sandy Billabong (Nourlangie catchment), was a result of mine site contribution or due to natural catchment influences.

In May 2007, mussels, sediment and water were collected from 5 sites along Magela Creek. The sites included two reference sites, Bowerbird (located approximately 20 km upstream of the mine site) and MCUS (1 km upstream of the mine), and three exposed sites, Georgetown Billabong outlet (GTC, about 1 km downstream of the mine), G0009 creekside monitoring station (009C, 5 km downstream of mine) and Mudginberri Billabong (12 km downstream).

Dried mussel flesh, dried whole and <63 µm sieved sediment and water were acid digested and measured for a suite of analytes including U, Ra and Pb isotope ratios. All analyses are complete except for Radium which is expected to be complete by March 2008.

Preliminary results for uranium (Figure 3) show no age dependency for uranium bioaccumulation, though there are higher concentrations of uranium in mussels close to the mine at GTC.

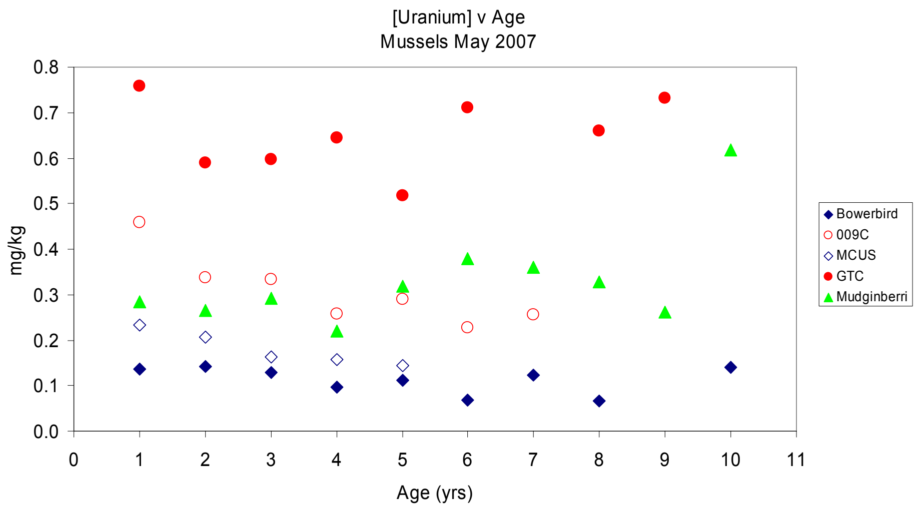

Figure 3 Uranium concentrations in mussel flesh compared to age of mussels from five sites along Magela Creek

Comparison of the average uranium concentration in mussels (1 to 5 year olds) from this collection compared to 1980 mussel uranium data collected by Allison and Simpson (1989) show similar levels at Mudginberri and Bowerbird (Figure 4). The 1980 mussel U concentration in Georgetown Billabong is much higher than the recent collection, but it should be noted that the May collection was not from the billabong but instead from the channel outlet which is influenced to a much greater extent by adjacent Magela Creek waters. Since the 1980 analyses were a baseline assessment, it suggests that the elevated uranium concentrations are more a function of natural ore body contributions rather than any mine site influence. This is an initial assessment, with overall interpretation of data scheduled for March 2008 after all analyses have been completed.

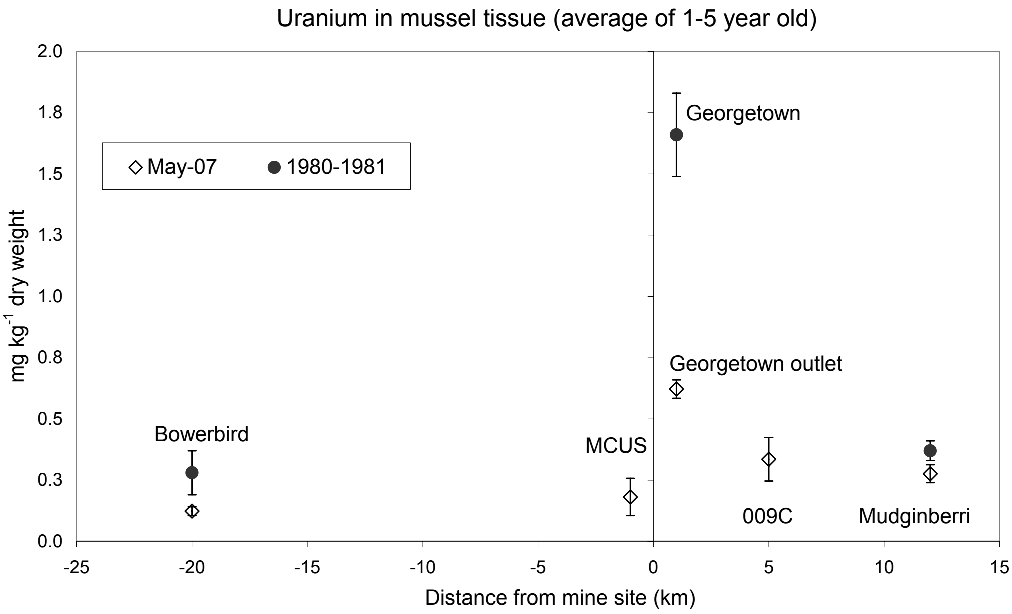

Figure 4 Comparison of May 2007 uranium concentrations in mussels along Magela creek, upstream and downstream of the mine with historical data.

Bioaccumulation of uranium in fish

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. Time series concentrations of uranium in the flesh of forktail catfish collected from Mudginberri and Sandy Billabongs are summarised in Figure 5, together with U concentrations measured in water and sediment.

The concentrations of U in the flesh of forktail catfish are low (<0.02 mg/kg) with no significant variation over time (Figure 5). The large range in the error bars indicates that sample contamination was a significant issue in 1999 and 2001. However, refinement of sample processing methods and analytical procedures reduced the influence of contamination in subsequent years.

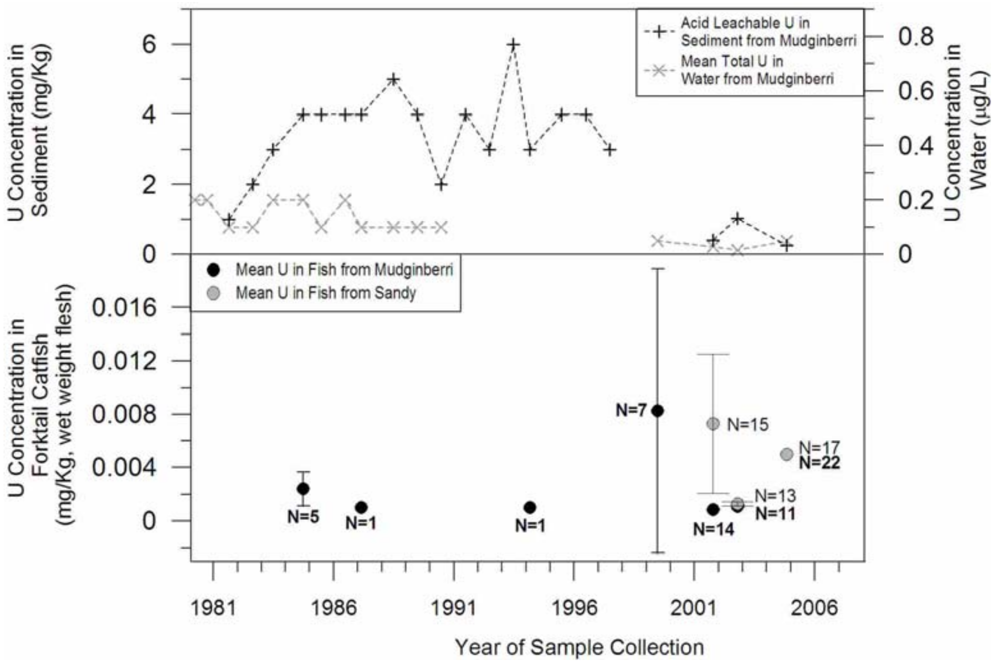

Figure 5 Mean concentrations of U measured in the flesh of forktail catfish, sediment and water samples collected from Mudginberri and Sandy Billabongs, since 1981. Error bars represent standard error.

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Monitoring of Ranger mine using macroinvertebrate community structure

C Humphrey, L Chandler, C Camilleri & 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) (Humphrey et al 2006). Samples were collected from macrophyte-edge habitat at 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, a dissimilarity index is calculated. This index is 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 (Faith et al 1995). Analysis of the full macroinvertebrate data set from 1988 to 2007 has been completed with results shown in Figure 1. This figure plots the paired-site dissimilarity values using family-level (log-transformed) data, for the two 'exposed' streams and the two 'control' streams.

Inferences that may be drawn from the data shown in Figure 1 are weakened because there are no pre-mining (pre-1980) data upon which to assess whether or not significant changes have occurred as a consequence of mining. Notwithstanding, the plots show that the mean dissimilarity value for each stream across all years is approximately the same (~0.3) and that the values are reasonably constant over time. Confirming this, in a multi-factor ANOVA (a statistical comparison) using the pooled data as depicted in Figure 1, no significant difference in the mean dissimilarities between the two treatment groups, 'control' versus 'potentially disturbed' streams was observed.

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 same macroinvertebrate data that were 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 2007), together with all other control sites sampled for the same period. Because the data-points associated with these two sites are 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.

As noted above (Brazier & Turner 2008), Magela and Gulungul catchments experienced record rainfall in the late February and early March 2007 period, with correspondingly record high flood levels reported in Magela Creek. The effect of this flooding on the abundances of macroinvertebrates sampled from the two Magela Creek sites at the end of the 2006–07 wet season is evident in Figure 3; densities are the lowest reported for any of the streams over the past 11 years, the result of stream bed scouring and (from field observations) loss of macrophytes that provide habitat for these stream organisms. The low abundances evident at both sites, upstream and downsteam of Ranger, indicate no mine-related effects on this response variable.

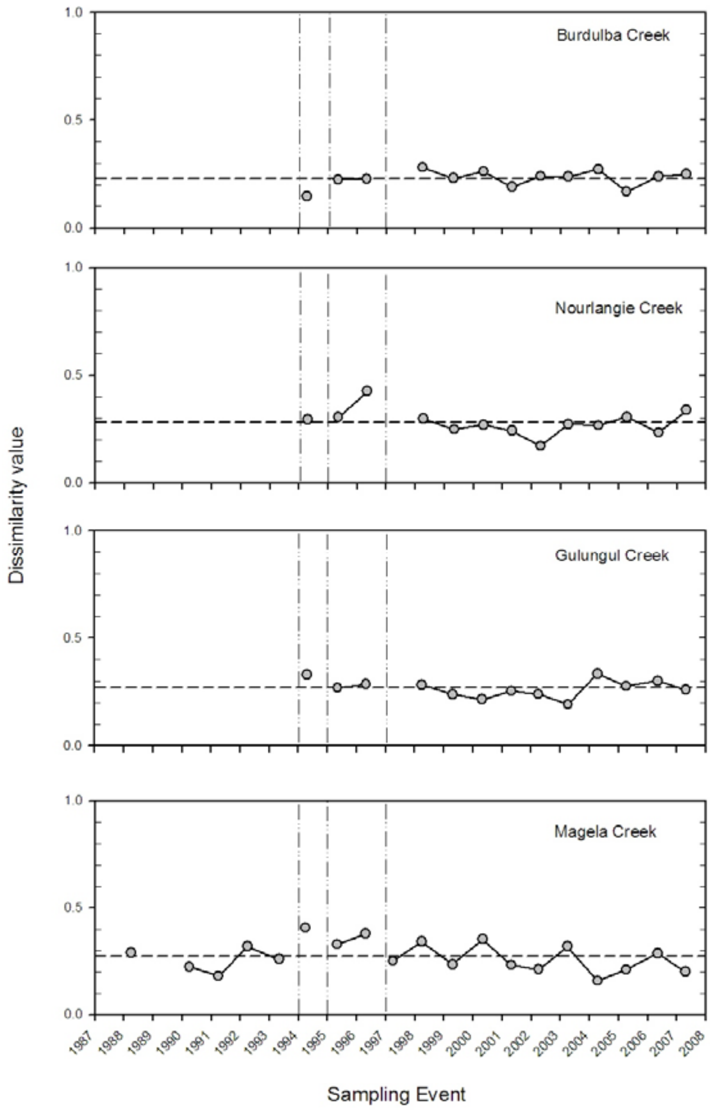

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2007. The dashed vertical lines delineate periods for which a different sampling and/or sample processing method was used. Dashed horizontal lines indicate mean dissimilarity across years.

Collectively, these multivariate and abundance results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining in the period 1994 to 2007 – including the period of high load of mine-derived salts to Magela Creek during the flood events in February and March 2007 (Turner et al 2008) – have not adversely affected macroinvertebrate communities.

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 KKN 2.2.1).

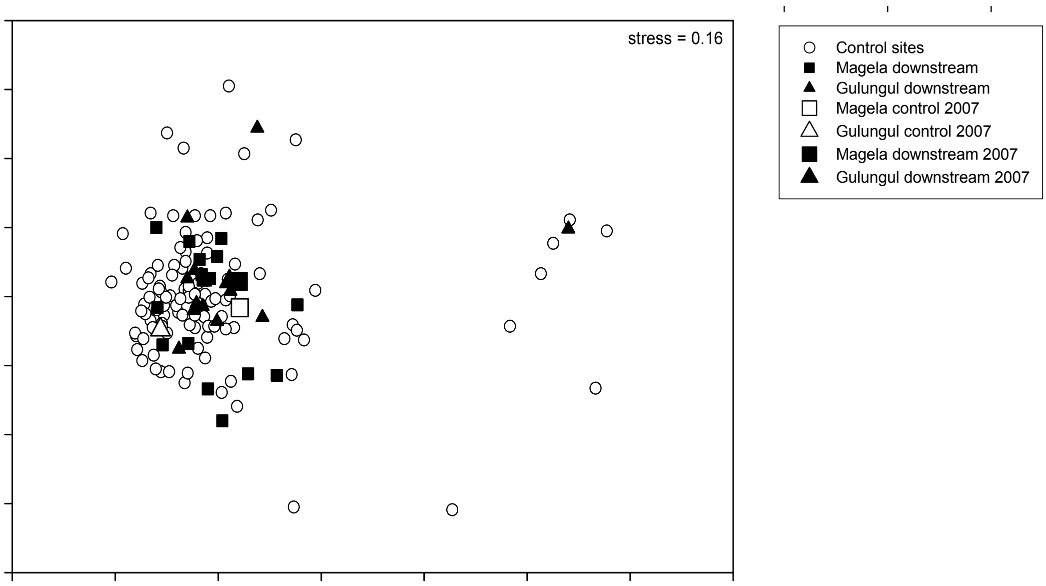

Figure 2 Ordination plot of macroinvertebrate communities sampled from sites in several streams in the vicinity of Ranger mine for the period 1988 to 2007. Data from Magela and Gulungul Creeks for 2007 are indicated by the enlarged symbols.

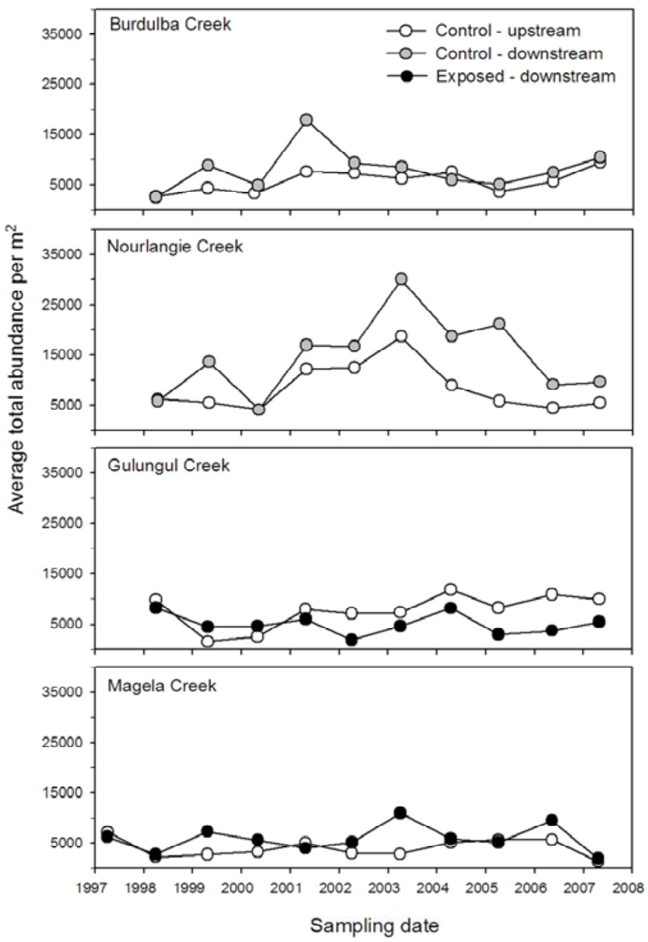

Figure 3 Macroinvertebrate total abundances recorded from paired upstream-downstream sites in several streams in the vicinity of the Ranger mine for the period 1997 to 2007

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Monitoring of Ranger mine using fish community structure

D Buckle & C Humphrey

Assessment of fish communities in billabongs is conducted between late April to the end of June of each year. Data are gathered, using non-destructive sampling methods, from 'exposed' and 'control' sites in deep channel billabongs and shallow weedy lowland billabongs. Details of the sampling methods and sites were provided in the 2003–04 Supervising Scientist Annual Report.

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 above ('Monitoring using macroinvertebrate community structure'). A significant change or trend in the dissimilarity values over time could imply mining impact.

Shallow lowland billabongs

The scope of the monitoring program for fish communities in shallow billabongs was reviewed after the 2005 sampling. An outcome of the review was a reduction to biennial sampling in six billabongs that comprise three 'control' versus 'exposed' site pairs. The reduction in sampling effort to biennial sampling was justified on the basis that an extensive baseline of data had been collected since 1994 together with evidence that changes to fish communities arising from potential mining impacts would arise over long time spans.

The similarity of fish communities in the directly exposed sites downstream of Ranger on Magela Creek (Georgetown, Coonjimba and Gulungul Billabongs) to those of the control sites (Sandy Swamp and Buba Billabongs on Nourlangie Creek and Wirnmuyurr Billabong – a Magela floodplain tributary) was determined using multivariate dissimilarity indices calculated for each sampling occasion. A plot of the dissimilarity values of the control-exposed site pairings – Coonjimba-Buba, Georgetown-Sandy Swamp and Gulungul-Wirnmuyurr Billabongs – from 1994 to the present, is shown in Figure 1. The basis for the pairing of corresponding control-exposed sites was made according to similarity in billabong size and susceptibility to seasonal drying.

The paired-site dissimilarities shown in Figure 1 average between 40 and 60% indicating fish communities in each of the billabongs comprising a site pairing are quite different from one another. Differences between the fish communities are most strongly linked to the type and density of aquatic plant communities in each billabong. For example, Coonjimba Billabong is dominated by an emergent reed (*Eleocharis* sp). Increasing density of *Eleocharis* (measured as weight) results in reduced fish abundance (Figure 2) and fish species number ($R^2 = 0.12$, $p = 0.012$; data not shown). In contrast, Buba Billabong is dominated by emergent grasses and in this habitat the number of fish species is greatest at an intermediate grass density (unimodal regression, $R^2 = 0.2$, $p = 0.0048$). Thus, fish find increasing densities of *Eleocharis* unfavourable but are favoured by emergent grasses until these become so dense that they physically prevent fish movement. For either habitat, excessive aquatic plant densities are unfavourable for fish communities.

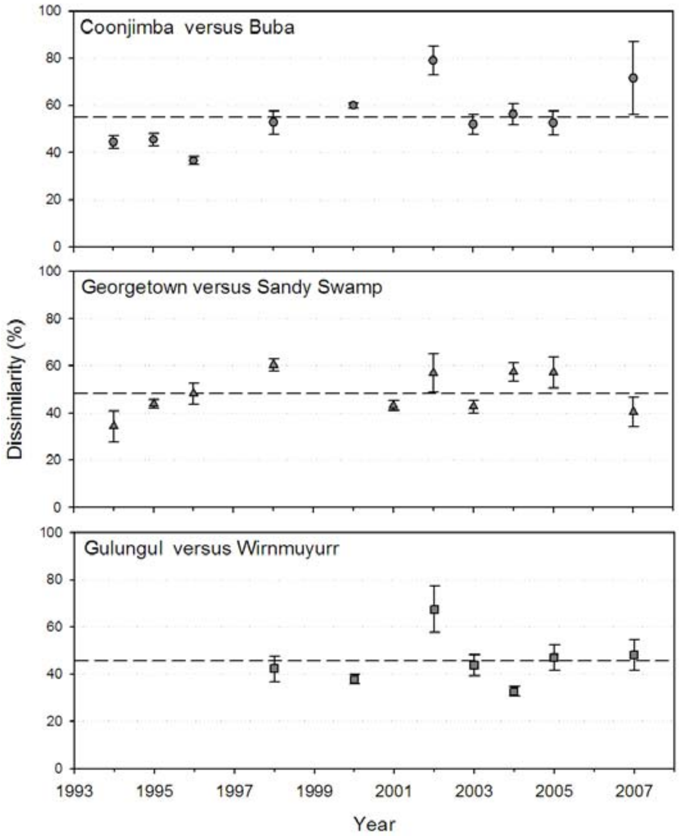

Figure 1 Paired control-exposed site dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in 'directly-exposed' Magela and 'control' Nourlangie and Magela Billabongs in the vicinity of the Ranger uranium mine over time. Values are means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of average trap enclosure data between the pairwise billabong comparisons, Coonjimba-Buba, Gulungul- Wirnmuyurr and Georgetown-Sandy Billabongs.

The particularly high dissimilarity values observed in the Coonjimba-Buba and Gulungul-Wirnmuyurr site pairings in 2002 and the Coonjimba-Buba site pairing in 2007 (Figure 1) are attributable to high plant densities in one or both of the billabongs comprising the site pairing. In 2002, there was a particularly high biomass of aquatic vegetation caused by drier (than normal) conditions at the time of sampling. This sampling was conducted after a wet season of relatively low rainfall, possibly resulting in favourable growing conditions and the rapid drying of billabongs that effectively concentrated and increased the density of aquatic plants in the available sampling areas.

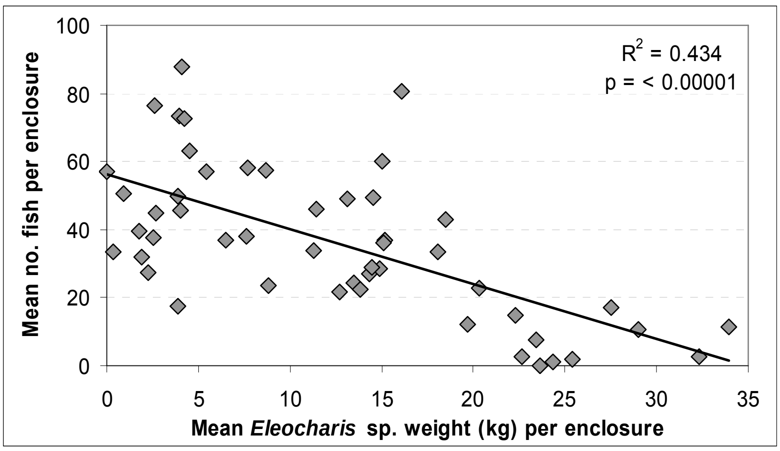

Figure 2 Regression relationship between average fish abundance and average weight of *Eleocharis* sp (kg) per trap enclosure in Coonjimba Billabong since 1994

The relatively high dissimilarity associated with the Coonjimba-Buba Billabong pairing in 2007 is attributable to unusually high *Eleocharis* densities in Coonjimba Billabong in this year. This increase in density appears to be related to natural growth cycles, whereby a reduction in density (as experienced in 2003) is followed by a recovery period. Of particular note, one of the sites randomly selected for sampling in Coonjimba Billabong comprised the only area without *Eleocharis*. As expected, the fish community sampled here differed greatly from the other sites. This caused a large variance in the data set for the fish community structure within Coonjimba, in turn reflecting the high paired-site dissimilarity value for 2007.

The dissimilarity associated with the Coonjimba-Buba site pairing has increased over time ($R^2 = 0.21$, $p = 0.0008$) (Figure 1). This increase is significant irrespective of the previously-discussed high values in 2002 and 2007 ($R^2 = 0.21$ $p = 0.0026$, 2002 and 2007 removed). ERA's water chemistry data and ***eriss*** water quality spot checks taken at the time of sampling show that inputs of salts from the mine site (based on electrical conductivity data) to Coonjimba Billabong have increased over time. However, because the levels of contaminants (including Mg and U) are relatively low in relation to the concentrations known to adversely affect fishes, it is unlikely the increasing paired-site dissimilarity values are directly related to mine site inputs. The increase in dissimilarity is more likely the result of changes to the structure of aquatic plant communities in both billabongs, though whether altered water quality or hydrological conditions in Coonjimba Billabong, as a consequence of mining at Ranger, are the cause of these changes is not known at this stage. Further analysis is required to properly infer the cause of this increase in paired-site dissimilarity over time.

Channel billabongs

In a similar manner to fish communities in shallow billabongs (discussed above), the similarity of fish communities in Mudginberri Billabong (directly exposed site downstream of Ranger in Magela Creek catchment) and Sandy Billabong (control site in the Nourlangie Creek catchment) was determined using multivariate dissimilarity indices calculated for each annual sampling occasion. A plot of the dissimilarity values from 1994 to the present is shown in Figure 3.

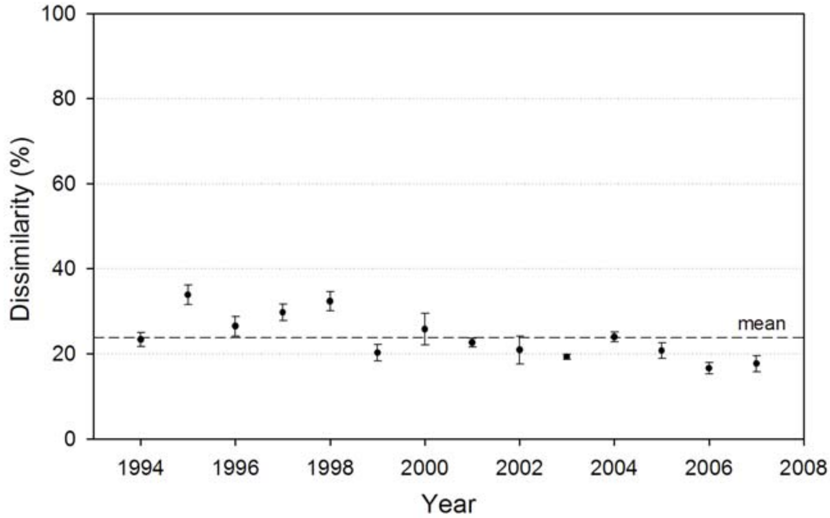

Figure 3 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of the Ranger uranium mine over time. Values are means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of transect data between the two billabongs. There has been a significant decline in paired-site dissimilarity over time but there is no evidence that this decline is mine-related (see text for further explanation).

In the Supervising Scientist Annual Report for 2003–2004, an apparent decline over time was noted in the paired-site dissimilarity measures. The value for 2007 is the second lowest on record (Figure 3).

The decline is primarily attributed to the particularly high abundances of chequered rainbowfish (*Melanotaenia splendida inornata*) and to a lesser extent glassfish (*Ambassis* spp) in Mudginberri Billabong in the early years of the study, relative to Sandy Billabong. Chequered rainbowfish have declined in Mudginberri Billabong since sampling commenced in 1989. The decline in rainbowfish numbers and, by association, the paired billabong dissimilarity value, is not related to any change in water quality over time as a consequence of water management practices at Ranger. This issue was examined in greater detail in the Supervising Scientist's 2004–05 Annual Report where the environmental correlates (1) wet season stream discharge, (2) natural, wet season stream solute concentration, (3) length of previous dry season, and (4) habitat conditions on Magela Creek floodplain, were identified as possible causes of the decline in rainbowfish.

The decline has been less influenced by chequered rainbowfish and glassfish in the latter years, suggesting that more subtle changes in community structure are also occurring.

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Monitoring support tasks

C Humphrey & D Buckle

Publication of protocols for the SSD's stream monitoring program in Magela Creek

Background and progress to date

Protocols for the SSD's stream monitoring program are being prepared for publication. Progress in preparing these protocols has been reported to ARRTC regularly. Two types of protocols are being prepared, high-level protocols and operational, detailed manuals pertaining to each of the protocols. Completion of operational manuals has rated a higher priority than high-level protocols because of the risks associated with loss of corporate knowledge should critical staff leave the SSD.

Work on the operational manuals and high-level protocols is well advanced and it is anticipated that they will be completed by the end of 2007. A biostatistician, Dr Keith McGuinness from CDU, has been assisting ***eriss*** with design and analysis aspects of the protocols, incorporating altered design and analysis features that have arisen as a consequence of reviews and refinements to some of the monitoring techniques – as described below.

Internal review of the routine biological monitoring program

Background

In early October 2006, a workshop was held to internally review the SSD's routine biological monitoring program. The review took into account:

1 Possible reduced sampling (frequencies/effort) for components of the program, considering factors such as:

   a Sensitivity of monitoring organisms to mine-related, water quality changes;

   b Adequacy of current datasets as a basis for monitoring during the operational and rehabilitation phase

   c Competing resources insofar as possible increased intensity of new monitoring approaches and rehabilitation research

2 Optimisation of existing techniques (ie similar results with similar power, but with fewer samples/data); and

3 Wishes of stakeholders, including local landowners

Summary outcomes

In summary form, the main conclusions arising from the review included:

1 Early detection techniques

(a) Toxicity monitoring (creekside and in situ)

- The highest priority task that must be conducted over the 2006–07 and 2007–08 wet seasons is the comparison of creekside and in situ snail egg production tests and development of a baseline for the in situ snail testing (see Toxicity monitoring descriptions above).
- It was noted that larval fishes were not particularly sensitive to Ranger mine waters and therefore may not be adding significant value to the toxicity monitoring program. Any additional resources that may be available after parallel snail studies (creekside-in situ) should be directed at comparing fish larvae survival between creekside and in situ test conditions at the upstream (only) site. These results may provide some insight as to the cause of the higher mortality rates of larval black-lined rainbowfishes in the upstream creekside station (compared with mortality at the downstream station). A final decision on abandonment of the fish larvae test altogether would be made after the 2007–08 wet season.

(b) Bioaccumulation in fish and mussels

- Bring Aquatic Ecosystem Protection and Environmental Radioactivity program work together, under a unified collection, sample processing and analysis regime.
- Complete a chemical risk assessment using ICPMS scans of waters from a variety of key locations to identify which metals (dilution and attenuation taken into account) need to be the focus of the future bioaccumulation analysis suite (completed and data interpretation underway).
- Freshwater mussels:
  - Initiate a sampling program of mussels resident in Magela Creek upstream and downstream of Ranger to address the issue of the extent to which inputs of Ra from the Magela Creek catchment itself (rather than the mine) may be contributing to the higher levels of Ra in Mudginberri Billabong mussels compared to Sandy Billabong mussels (conducted in May 2007).
  - Annual sampling of mussels in October 2007, possibly modified on the basis of outcomes of part a (if data are available).
- Fishes:
  - The adequacy of existing fish metals and radionuclide data needs to be assessed.
  - This assessment will determine whether (i) separate organ compositing is possible, and (ii) sufficient low-concentration data are available such that future sampling can be restricted to ‘reactive’ collections of fishes undertaken in response to significant mine events/incidents. If condition (ii) is not met, additional sampling should be conducted.

2 Biodiversity assessment techniques

(a) Macroinvertebrate communities from stream sites

- Annual sampling should continue (per current sampling design and configuration of sites)

- A level of pooling of within-site replicates should be examined that optimises sample processing costs and statistical power

(b) Fish in channel billabongs

- Annual sampling should continue (per current sampling design and configuration of sites)
- The number of observations made per site-transect should be reduced from 5 to 4

(c) Lowland billabongs adjacent to Ranger, monitoring using:

- Fishes
  - The default position is sampling every other year in a 3 exposure-reference paired site configuration (from 2007 onwards) unless it is decided that in the due year, intensive macroinvertebrate sampling should be conducted (see below). (In this circumstance, no fish sampling will be done.)
- Macroinvertebrates
  - The 3-year dataset (1995, 1996 and 2006) should be analysed to determine (i) whether there are logical exposure-reference waterbody pairs, and (ii) whether there are sufficient existing data to restrict further sampling to a smaller configuration of billabong pairs – as isnow done with fish in lowland billabongs.
  - For both water quality closure criteria development and monitoring purposes, further macroinvertebrate sampling should be conducted after wet seasons of particular ‘interest’, including seasons where water quality has been significantly poorer than in preceding seasons. This would provide the potential for closure criteria to be reviewed and enable the SSD to assess the extent, if any, of potential impacts.

Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer, P Medley & 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 assess if there have been any changes in the downstream ^{226}Ra levels. This is due to the potential risk of increased exposure to radiation via the biophysical pathway due to mining activities. Mussels, in particular, bioaccumulate ^{226}Ra (Johnston et al, 1987) which may then be incorporated into the human body upon consumption. Due to the high dose conversion factor of ^{226}Ra this radionuclide dominates the predicted dose to people living downstream of the Ranger mine from the ingestion of bushfoods (Martin et al 1998).

Water samples are collected weekly in Magela creek and monthly in Ngarradj from sites located upstream and downstream of the project areas. Weekly samples from Magela Creek were composited in 2006–07 to give monthly averages. Samples are analysed for total ^{226}Ra (ie combined filtered and particulate phase) at ***eriss***'s Environmental Radioactivity section following a method described in Medley et al (2005).

Results

Magela Creek

The ^{226}Ra activity concentration data to date from the 2006-07 wet season can be compared to previous wet seasons in Figure 1. The data show that the levels of ^{226}Ra are very low in Magela Creek, upstream as well as downstream of the Ranger mine.

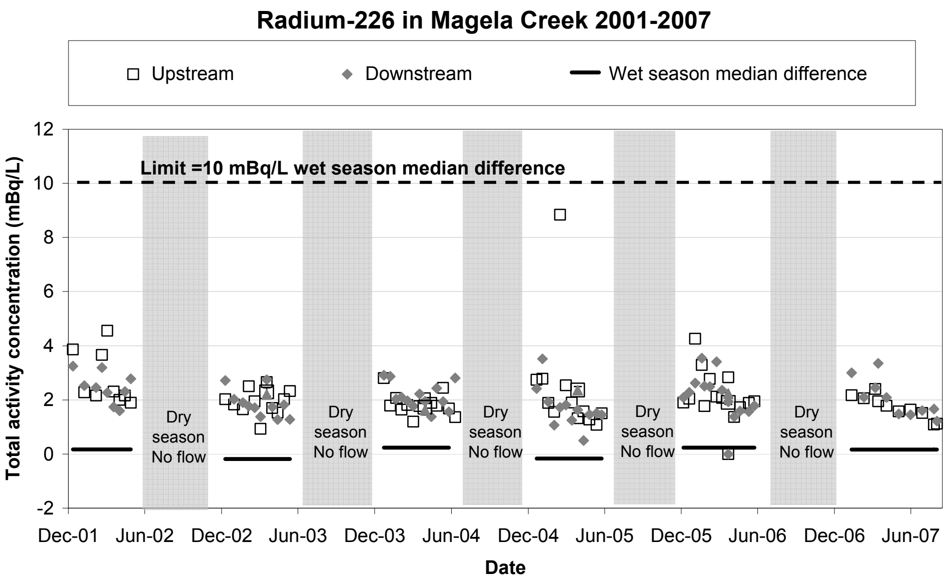

Figure 1 Radium-226 in Magela Creek for the 2001–07 wet seasons

Table 1 shows the median and standard deviations for individual wet seasons (2003–07) and for the entire study period (2001–07). The '*wet season composite samples*' (not shown) for both the upstream and downstream locations compare well to the wet season median but have a lower variability due to the compositing. In 2005–06 the '*wet season composite sample*' activities from Magela Creek upstream and downstream were 1.9 and 2.2 mBq/L, respectively, which compares well with the median of the individual measurements.

Table 1 Statistics for total ^{226}Ra activity concentrations [mBq/L]

		All years 2001–07	2002–03	2003–04	2004–05	2005–06	2006–07
Magela Creek							
Median and standard deviation	upstream	1.9 (± 1.1)	2.0 (± 0.5)	1.8 (± 0.4)	1.7 (± 2.1)	2.0 (± 0.8)	1.7 (± 0.4)
	downstream	1.9 (± 0.6)	1.8 (± 0.5)	2.0 (± 0.5)	1.6 (± 0.7)	2.3 (± 0.7)	1.9 (± 0.7)
Wet season median difference		0.1	-0.2	0.2	- 0.2	0.2	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)	1.1 (± 0.5)
	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)
Wet season median difference		0.0	- 0.3	- 0.2	- 0.3	-0.5	-0.1

Based on the potential dose received from the ingestion of ^{226}Ra in the freshwater mussel *Velesunio angasi* (Martin et al 1998), a limit of 10 mBq/L increase above natural background in total ^{226}Ra concentration in surface waters downstream of Ranger has been defined for human radiological protection purposes (Klessa 2001). The wet season median difference should not be more than this limit and is shown by the green lines in Figure 1.

The wet season median difference for the 2001–07 wet seasons is approximately zero. The available data for the six sampling seasons indicate that ^{226}Ra levels in Magela Creek are due to the natural occurance of radium in the environment and that ^{226}Ra activity concentrations in Magela Creek water are not elevated downstream of Ranger uranium mine.

Ngarradj

^{226}Ra activity concentrations in Ngarradj show that the levels of ^{226}Ra are very low in Ngarradj. and ^{226}Ra activity concentrations at the Ngarradj 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 (or smaller), indicating ^{226}Ra activity concentrations in Ngarradj are not effected by the Jabiluka project.

Steps for completion

The ^{226}Ra monitoring in Magela Creek and Ngarradj will continue in the 2007-08 wet season as for the previous season except that monitoring of the upstream site at Ngarradj will not occur.

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Part 2: Ranger – Rehabilitation

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2.1 LANDFORM DESIGN

KKN 2.1.5 Geomorphic and geochemical behaviour and evolution of the landform

KKN 2.1.6 Radiological characteristics of the final landform

2.2 ECOSYSTEM ESTABLISHMENT

KKN 2.2.1 Development and agreement of closure criteria from ecosystem establishment perspective

KKN 2.2.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites

[1] List of papers grouped by Key Knowledge Need.

Assess the geomorphic stability of the currently proposed final landform at the Ranger mine using landform evolution modelling

JBC Lowry & KG Evans

Introduction

This work is a continuation of the 2005–06 workplan where co-operative work was conducted with EWLS to assess erosion on a proposed rehabilitated landform using the Siberia landform evolution model. In previous years proposed landform designs could only be modelled for surface conditions representing best and worst case scenarios as it appeared as though Siberia was unable to perform distributed hydrology modelling. That is, it was unable to implicitly address the different hydrology characteristics associated with the different surface conditions across the landform.

Consequently, in addition to assessing landforms developed by ERA/EWLS, an important aim of the work during the 2006–07 year was to ascertain whether current problems in incorporating hydrological parameters into the Siberia landform evolution model were a result of problems within the GIS interface (ArcEvolve), or the Siberia software itself.

Progress to date

During 2006–07, no new landforms were supplied by EWLS for assessment. However, the existing proposed landform (number 785-014) was used to assess the ArcEvolve and the Siberia software used for landform modelling.

Through the assessment of these packages, it became clear that the ArcEvolve GIS interface (Boggs 2003), rather than the Siberia landform model, was unable to retain or incorporate distributed hydrological parameters. To redress this situation the Siberia code has been modified by the University of Newcastle to perform distributed hydrology simulations independently of the interface. Simulations indicate that under a 'worst case' scenario, erosion to a depth of 12 metres may occur on the most vulnerable areas of the landform. Conversely, the 'best case' scenario, which applied distributed hydrology parameters across the landform indicated erosion to a depth of 9 metres. The modified version of Siberia, in stand-alone-mode, will be used for future landform assessment to ensure that distributed hydrology parameters are used in simulations.

Results of Siberia modelling were presented at the 1st International Seminar on Mine Closure, in Perth (September 2006), and at the Uranium Conference 2007, organised by the AusIMM in Darwin (May 2007).

Steps for completion

Siberia will continue to be used to evaluate the erosion performance of future final landform designs proposed by ERA.

Recently it has been proposed by ERA that the cover of the landform will comprise at least a 2m thick layer of mixed rock and laterite (weathered material incorporating variable percentages of clay fraction material). This type of cover has not previously been assessed in modelling runs using Siberia. Erosion properties need to be estimated or measured for this material and these new parameters used for runs of both the Siberia and CAESAR landform evolution modelling packages.

It is further recognised that the Siberia model needs to be upgraded to enable simulation of both short duration extreme rainfall events, and of continuous rainfall event records. This work is currently underway

An important issue which needs to be addressed by ERA is to ensure that the source digital elevation models (DEMs) supplied to ***eriss*** are at a resolution appropriate for the scale of the landform modelling. To date, the landforms have been supplied by EWLS as DEMs with a horizontal resolution of 25 m. These have subsequently been resampled to a resolution of 12 m. However, this resolution is much poorer than required to track the evolution of erosion features, appropriate to the scale of the landform. A DEM with a resolution of 10 m is regarded as optimal for the Ranger area.

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

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

Introduction

Until recently the Siberia landform evolution model (LEM) was the only geomorphic computer modelling code being used to predict the long term behaviour of the Ranger rehabilitated landform. Siberia simulations use as input an average area-discharge relationship for a wet season and do not use the time series hydrology of a single rainfall event or series of events. Consequently, the average long-term erosion assessments conducted to date have not implicitly addressed the impact of an extreme rainfall event or a series of events comprising an 'extreme' wet season.

This project aims to assess the impact of Probable Maximum Precipitation (PMP) events on the erosional stability of rehabilitated landforms and the surrounding natural catchment using the CAESAR (**C**ellular **A**utomaton **E**volutionary **S**lope **a**nd **R**iver) (Coulthard 2001) computer model.

CAESAR uses a digital elevation model (DEM) of a catchment or river reach, water and sediment flux data, and rainfall data to simulate evolution of a landform in small timesteps. It features slope processes (soil creep, mass movement), hydrological processes, multidirectional routing of river flow and fluvial erosion and deposition over a range of different grain sizes. A tracing component allows input of contaminant data to assess transport downstream. This is especially important because it enables routing of sediment lost from the landform through a river catchment. Such functionality is not present in Siberia.

CAESAR can also be used to simulate the impact of extreme rainfall events, since time series data for actual or simulated rainfall events can be used as input. This ability is a critical attribute given the long times required to contain radioactive material, and the probability that one or more very extreme rainfall events will occur. Extreme event testing of the proposed design parameters for the constructed landform has assumed greater importance given the possibility of an increase in frequency of intense rainfall periods as a consequence of climate change. The key differences between Siberia and CAESAR are summarised in Table 1.

Progress to date

Implementation of CAESAR has been conducted in collaboration with Professor Tom Coulthard of Hull University UK, the model's author. Studies comparing CAESAR outputs with Siberia and natural erosion rates are being conducted in co-operation with Dr Greg

[1] University of Hull, UK.
[2] The University of Newcastle, NSW, Australia.

Hancock of the University of Newcastle, NSW. A hydrologically corrected, and pit-filled digital elevation model (DEM) for the proposed final landform and surrounding undisturbed areas (Fig. 1) is available. Unlike Siberia, CAESAR can not presently simulate the effect of different areas of surface treatment on a landform so intial simulations used a sub-catchment which flows into Magela Creek (Fig.1). The study catchment, referred to as the tailings dam catchment (TDC), is the largest sub-catchment on the landform and drains the tailings dam through a channel between the Pit1 cap and Pit 3 cap.

Table 1 Differences between the Siberia and CAESAR models

Siberia	CAESAR
Sediment transport and hydrology equations optimised for average annual data therefore, cannot simulate individual events	Uses hourly rainfall data so can output data on hourly basis – potential application to extreme events
Relatively quick to run	Slower to run for simulated periods of tens to hundreds of years
Able to model different surface treatments across DEM representing different areas eg the waste rock dump, tailings pond etc	Presently cannot vary treatments across landform
Cannot simulate transport from the boundary of the landform into a receiving river	Can route sediment from the landform through a river catchment and simulate changes in river channel dimensions and morphology in response to sediment delivery

Sensitivity studies were conducted using observed rainfall to calibrate input parameter values to previously measured erosion rates for both mine disturbed areas and natural areas. Once parameter values were estimated, simulations were run using the Jabiru rainfall record comprising 22 years of complete hourly data; surface material size distributions for Ranger waste rock (Evans & Loch 1996); Magela Creek discharge (Q) data at gauging station G8210009 and estimated sediment discharge (Qs) for Magela Creek based on observed continuous turbidity data; and stream sediment size distributions of Roberts (1991). The Magela Q and Qs were used as inputs into Magela Creek at the upstream boundary of the DTM to simulate the effects of throughflow in the Magela reach of the TDC and to investigate how backflow effects of Magela Creek may effect transport and deposition of sediment leaving the TDC. The simulations were conducted using a vegetation growth rate of 0.6 years, so early rains fell on bare waste rock but once vegetation was mature it remained that way for the rest of the simulated period.

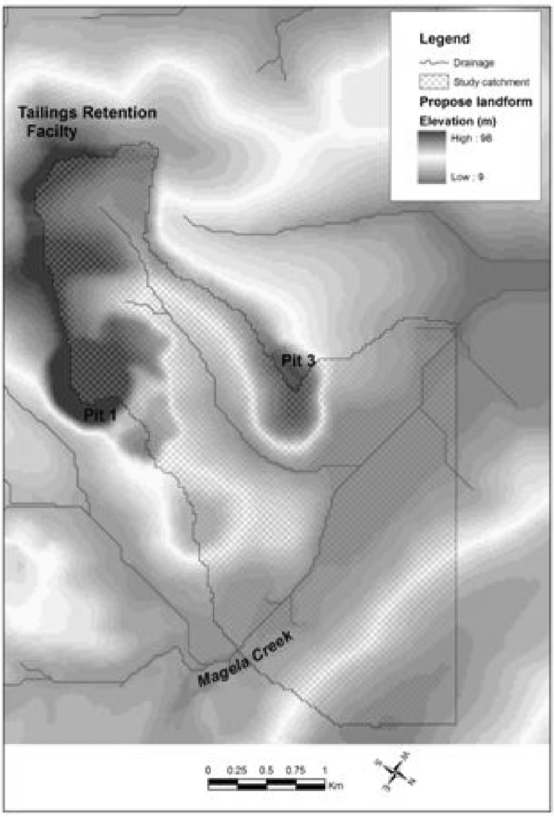

Figure 1 The Tailings dam catchment showing its relationship to Magela Creek and other mine site sub-catchments

Three simulations were run:

1 A 22 year period using the observed hourly rainfall record and daily Magela Q and Qs for complete years between 1972 and 2006;

2 A 22 year period as above with the 2006-07 time series for Q and Qs data inserted after 10years to assess the impact of an extreme event after the landform has reached equilibrium. There was extraordinarily high rainfall and discharge in 2006–07 where rainfall events between 6 and 72h exceeded a 1 in 100y event and the resulting flood exceeded a 100y event; and

3 A 22 year period using the the 2006/07 rainfall year to replace the first year of the 1972–2006 data used in 1 above. This was to simulate the effects of an extreme event on the landform in the wet season immediately after rehabilitation.

Results

Annual denudation rate for each simulation are shown in Figure 2. The simulations show that there is a high level of sediment loss 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 and vegetation grows, leaving coarser material in the thalweg[3] of drainage lines.

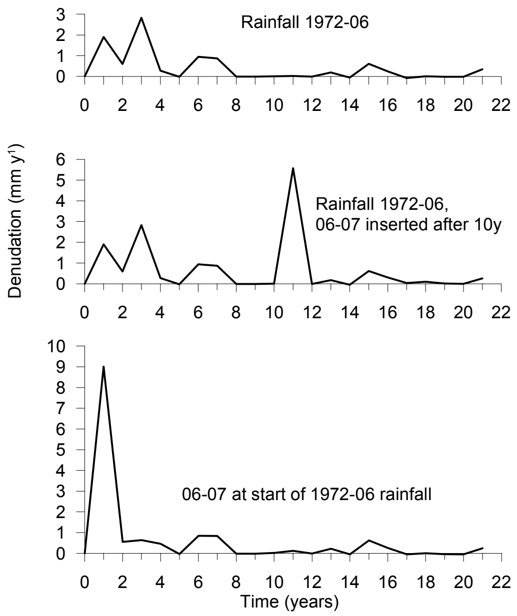

Figure 2 Denudation rates of the TDC and Magela reach sumulated using CAESAR with various rainfall inputs

Simulations show that the catchment is conditioned in 4 to 8 years. 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). After an initial phase of 5 years, denudation rates oscillate between positive (erosion) and negative (deposition) with a

3 The line connecting the lowest points along the valley floor, bed of a stream or drainage line.

median of 0.01 mmy^{-1} to 0.07 mmy^{-1} and an average of 0.17 – 0.21 mmy^{-1}, within a range of -0.05 mmy^{-1} to 0.94 mmy^{-1}. Negative rates indicate that Magela sediment input is greater than that exported and that deposition has occurred within the catchment.

Previous studies in the area give a range of denudation rates for waste rock of -2 mmy^{-1} to 7 mmy^{-1} with a median of 0.04 mmy^{-1} and for natural areas, denudation rates of -1.25 mmy^{-1} to 1.25 mmy^{-1} with a median of 0.018 mmy^{-1} (Erskine & Saynor 2000). The simulated erosion losses compare well with published. Since input parameter values were calibrated to published erosion rates, this result was expected and indicates adequate simulation of physical processes by the model. The initial large oscillations are also to be expected as any system in disequilibrium needs to find the steady state condition.

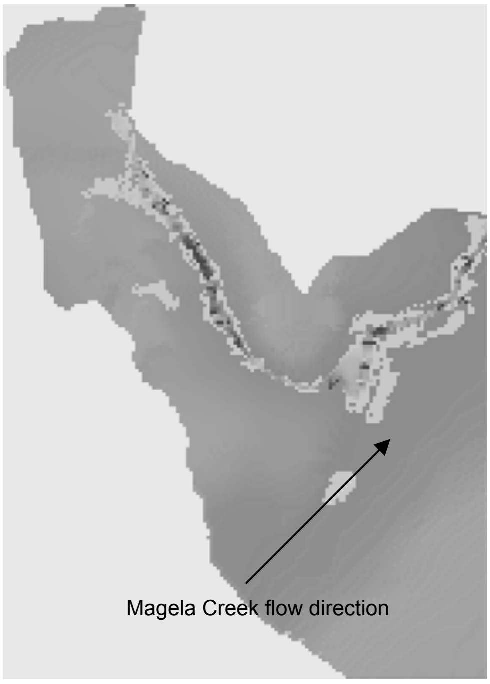

Figure 3 Screen dump of CAESAR output showing a plume of suspended sediment moving from the TDC into Magela Creek and leaving the study catchment

Initial application of the the 2006–07 rainfall time series (simulation 3 above) is sufficient to flush all of the mobile sediment from the catchment in the first year giving a denudation rate of 9 mmy^{-1}, with little further conditioning taking place in the following years. If, instead, this extraordinary event occurs after 10 years (simulation 2) a large pulse of sediment is exported with a denudation rate of 5.58 mmy^{-1} with erosion rates returning to 'normal' after a year.

This is an initial assessment only of the effects of the impacts of an extraordinary ranfall event on the TDC. It should be noted that although this study includes the Magela Creek reach and surrounds, the model as it is currently able to be run treats all areas of the catchment as being covered with waste rock material. It does not account for the different morphologies, sand bed channel or surface condition and vegetation abundance of the riparian zone and adjacent woodlands. However, the results to date indicate that CAESAR has excellent potential for simulating the effects of extreme events on landform stability and sediment export from the mine site to natural areas of the catchment (Fig 3).

Steps for completion

- Validation of input parameters values using field data and comparison with other models,
- Develop model capability to incorporate spatial variability in surface material types and to include vegetation growth and differential distribution (a collaborative project to do this is currently being developed with Prof Coulthard),
- Determine what is an 'extreme event' and obtain/suitable input data,
- Test the effects of DTM resolution,
- Run the model the with laterite and waste rock mix being proposed by ERA for the Rangerfinal landform,

- Compare long-term erosion rates between CAESAR and Siberia models (to be conducted collaboratively with Dr Hancock), and
- Specifically evaluate the importance of vegetation on landform stability.

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

A Frostick[1], A Bollhöfer, D Parry[1], N Munksgaard[1] & KG Evans

Introduction

This project aims to develop an innovative, sensitive and cost-effective methodology to assess and monitor impacts of past, present and future uranium mining activities in and near Kakadu National Park, and is funded through the ARC Linkage-Projects scheme. A combination of stable lead isotopes, trace metals and radionuclide techniques is used. Concentrations are measured in both surface scrapes and sediment cores to determine the sediment deposition history and extent of erosion and deposition of mining-related contaminants in potentially mining influenced catchments. A pilot study conducted in the Ngarradj catchment, influenced by developments on the Jabiluka mineral lease, outlines the general approach (Bollhöfer & Martin 2003).

Lead has four stable isotopes three of which, ^{206}Pb, ^{207}Pb and ^{208}Pb, are endmembers of the ^{238}U ($t_{1/2}$= 4.5·10^{9} yrs), ^{235}U ($t_{1/2}$= 0.7·10^{9} yrs) and ^{232}Th ($t_{1/2}$= 14·10^{9} yrs) decay chains, respectively. Primordial lead was formed at an isotope abundance of 2% (^{204}Pb), 18.6% (^{206}Pb), 20.6% (^{207}Pb) and 58.9% (^{208}Pb) (Tatsumoto et al 1973). The radioactive decay of uranium and thorium in the Earth's crust results in radiogenic lead being formed, and hence changes in crustal lead isotope ratios with time. Present day average crustal (PDAC) ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb ratios are approximately 1.20 and 2.47, respectively.

In uranium and thorium rich minerals more radiogenic lead has formed over time. For example, monazites with high Th/U exhibit ^{208}Pb/^{207}Pb and ^{206}Pb/^{207}Pb ratios much higher than 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) measured ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb ratios of 9.690 and 0.0494, respectively, in particulates from uranium tailings. Consequently, the ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb ratios of any mixtures between natural PDAC material and uranium rich sources fall within the grey area indicated in Figure 1.

Historically, lead isotopes have mainly been used as a source tracer in urban and developed areas (eg. Chow & Johnstone, 1965). However, conditions in the ARR provide an opportunity to develop the method as a monitoring tool for mining impacts in remote areas. As a relatively unpolluted environment is more susceptible to detect changes in the sources of lead via lead isotope ratios in various environmental compartments (aerosol, water, biota or soil), and due to the uniqueness of lead isotope ratios of the sources in the region (Gulson et al 1992, Munksgaard et al 2003, Bollhöfer et al 2006), this work provides a unique opportunity to study impacts of uranium mining on the environment in the Top End.

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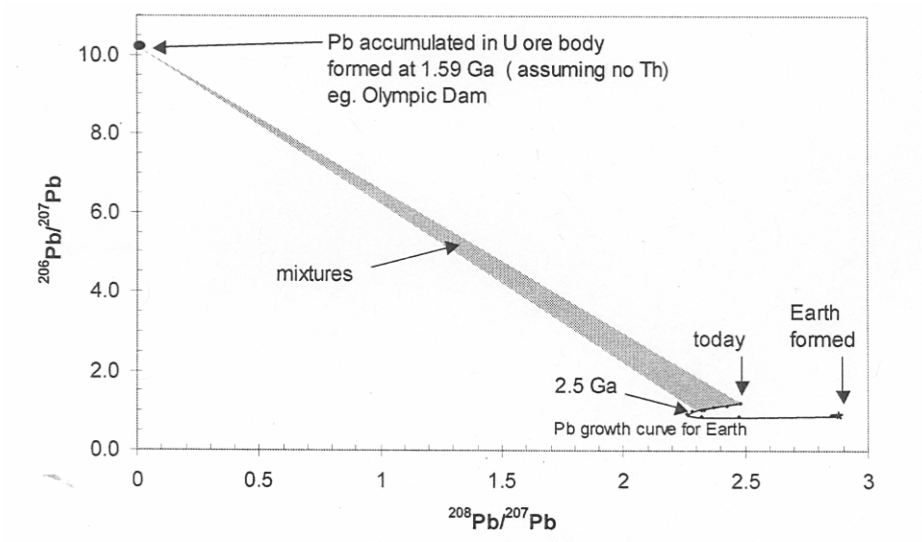

Figure 1 Origin of the isotopic composition of mixtures of radiogenic lead from uranium ores with common lead (from Bollhöfer et al 2005)

Methods

To determine the extent of erosion and deposition at the rehabilitated Nabarlek mine, surface scrape samples and 8 sediment cores were collected in 2005, in addition to samples that had been collected in 2003. The cores were taken in backflow deposition zones along Cooper Creek, upstream and downstream of Cooper Creek West, in the Kadjirrikamarnda catchment and downstream of its confluence with Cooper Creek (Figure 2). Cores were sliced into 1 cm increments along their lengths and each section prepared for measurement of metal content (ICPMS) and lead isotope ratios. Samples were then combined for gamma analyses for U and Th-series elements, ^{137}Cs and ^{40}K in an attempt to date the cores. Further samples for analysis were collected from the Gulungul and Magela catchments in the dry seasons of 2005 and 2006.

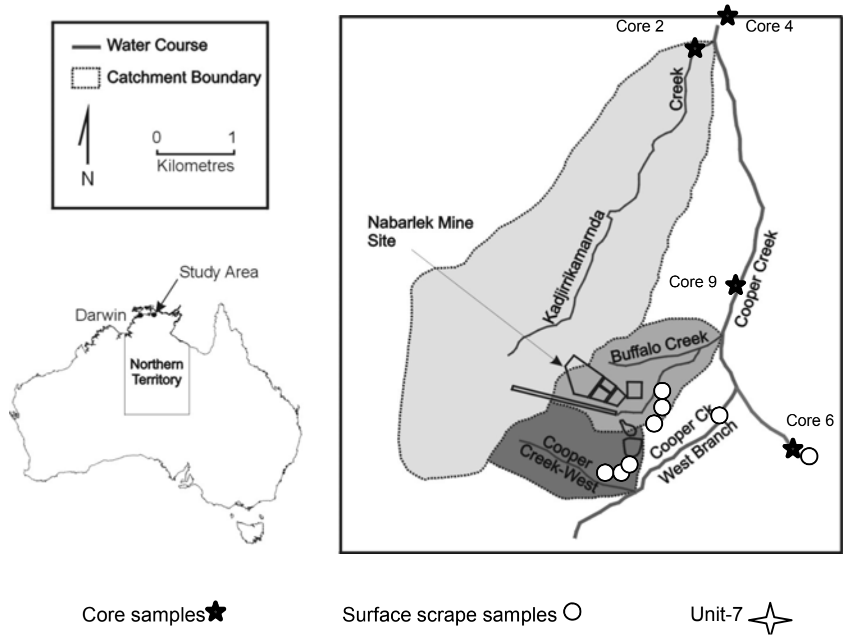

Figure 2 Location of the Nabarlek mineral lease and the sampling sites (modified from Hancock et al 2006)

Pb isotope ratios and trace metal concentrations in the sediments were measured by ICPMS at Charles Darwin University (CDU). Samples were digested in a nitric/perchloric acid mixture and heated in a block digester at 180˚C. An aliquot was dried down and redissolved in 10% nitric acid, before being injected into the plasma torch of a Perkin Elmer Elan 6000 ICPMS. Details of the operating system and the sample preparation can be found in Munksgaard et al. (1998) and Munksgaard and Parry (2000). Typical uncertainties for the Pb isotope ratios are 0.5–1% relative standard deviation.

Sediments were analysed for elements of the natural uranium and thorium decay series, and potassium-40, using high resolution High Purity Germanium (HPGe) gamma detectors at the Environmental Research Institute of the Supervising Scientist (***eriss***). Procedures for sample collection, preparation and measurements of radionuclide activities via gamma spectrometry at ***eriss*** are described in Marten (1992). The counting system has been calibrated for the respective sample geometry, using certified uranium and thorium standards.

Results

Unit-7, a bare area on the rehabilitated Nabarlek mine, that has previously been identified as the source of three quarters of the uranium eroding off site (Hancock et al 2006), exhibits highly radiogenic $^{206}Pb/^{207}Pb$ ($^{208}Pb/^{207}Pb$) isotope ratios of 13.14 (0.448). This isotope ratio is more radiogenic, and is distinctively different from, lead isotope ratios previously measured in Ranger uranium ore. The ratio can be used as a source tracer for material eroding off site.

Some soil scrapes collected on site and from areas adjacent to the Nabarlek mine site exhibit elevated levels of trace metals, radionuclides and distinct lead isotope ratios indicating the presence of uranium rich material (Figure 3). $^{206}Pb/^{207}Pb$ ($^{208}Pb/^{207}Pb$) isotope ratios of the scrapes taken along major drainage lines on-site indicate that material is actively eroding and depositing in overflow areas in the vicinity of the mine.

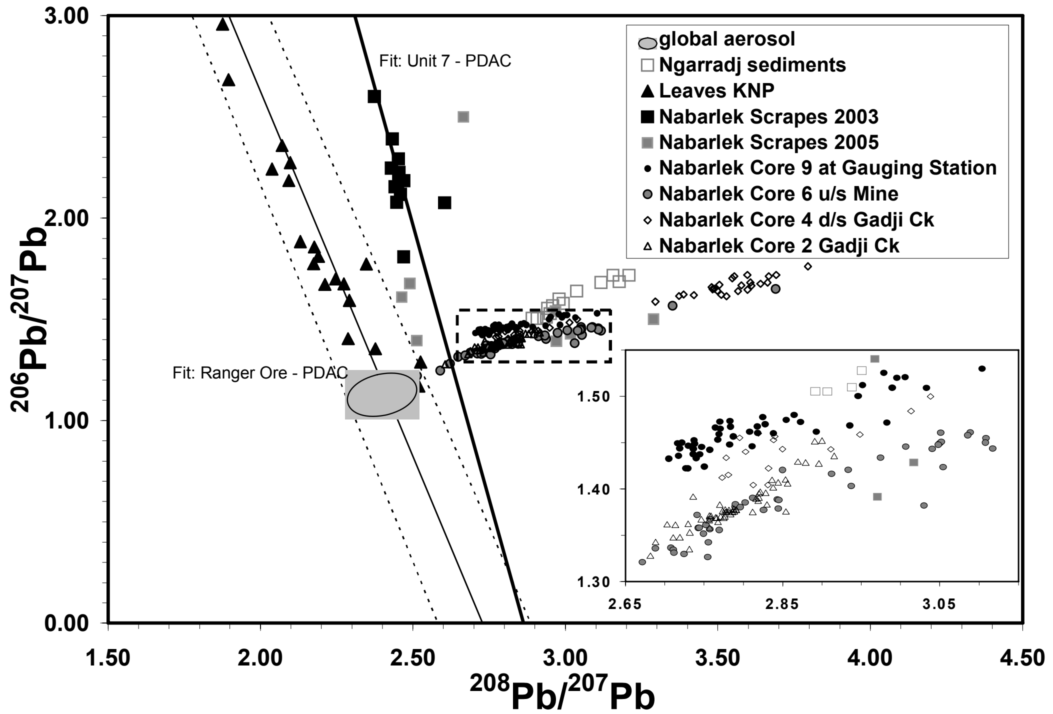

Figure 3 3-isotope plot of soil scrape and sediment samples taken at the rehabilitated Nabarlek mine

Although some of the radiogenic erosion products deposit outside the fenced area in a sediment settling pond and, since the breach of the settling pond wall, to the south of the rehabilitated waste rock dump retention pond, dispersion of this material is limited at this stage.

Cores 4 and 9 exhibit more radiogenic $^{206}Pb/^{207}Pb$ ratios as compared to the upstream core 6 and core 2 from the Kadjirrikamarnda catchment (see inset in Figure 3). This is most likely due to the higher radiogenic contribution from the catchment – the orebody (pre-mining) and the mine and rehabilitated site (post-mining) – upstream of these sites. The Kadjirrikamarnda catchment only delivers 3.3 % of the total ^{238}U flux from the Nabarlek mine site (Hancock et al 2006) and thus core 2 is relatively little influenced by radiogenic runoff.

Steps for completion

Sediment cores have been taken from Georgetown and Mudginberri Billabongs, from the Magela floodplain and the Gulungul catchment, to investigate pre-rehabilitation radionuclide and heavy metal concentration, sedimentation fluxes and lead isotope composition of the sediments in the Magela catchment. These sediments, and sediments from the Cooper Creek catchment at Nabarlek, are currently being investigated for their radionuclide activity concentrations and will be dated using a combination of ^{210}Pb, ^{137}Cs and Optically Stimulated Luminescence (OSL) dating techniques to investigate the history of regional sediment deposition. Additional scrape samples will be collected in the Magela and Gulungul Creek catchments, at Ngarradj and in the Nourlangie catchment in the dry seasons of 2006 and 2007. It is envisaged that Nourlangie Creek will act a a natural analogue for the study.

Acknowledgments

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

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

C Humphrey, D Jones, J Hanley, L Chandler & C Camilleri

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 mine site (Map 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 sample processing and 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 mine site 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 are still comparable to reference waterbodies in the region.

Results are reported here of the comparison between Georgetown Billabong macroinvertebrate communities from macrophyte and benthic habitats, and 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.

Progress to date

Sampling

Macroinvertebrates were sampled in May 2006 from Coonjimba, Georgetown and Gulungul Billabongs and Ranger Retention Pond 1 (RP1) and Retention Pond 2 (RP2) (mine water exposed sites) and Baralil, Corndorl, Wirnmuyurr, Malabanjbanjdju, Anbangbang, Buba and Sandy Billabongs and Jabiru Lake (reference sites, not exposed to Ranger mine waters). See Maps 2 and 3 for locations of these waterbodies.

In each waterbody, samples were collected from five sites and at each site, separate samples were taken from the macrophyte zone and littoral (near shore) sediment (benthic) habitats (thus 10 samples per waterbody).

Results

Macroinvertebrate community structure data (taxa and their relative abundances) from replicate sites of the different waterbodies were analysed using multivariate procedures. One of these techniques, non-metric, multi-dimensional scaling ordination (NM-MDS), depicts the community structure of samples graphically, in a reduced (typically two or three) dimensional space. The closer samples (in this case replicate sites) are together in ordination space, the more similar is their community structure. Possible mine-related effects upon community structure would be inferred if mine-water-exposed sites were separated from those of reference sites, while interspersion of replicates of the different exposure types would imply no significant difference in community structure.

Figure 1 illustrates this concept for the separate macrophyte and benthic habitats, as well as for the combination of habitats (for comparison with previous years). For each analysis, ordinations were constructed after replicate sites from the Ranger RP2 water body (mine site pond water) were removed from the datasets. This was done because the macroinvertebrate communities were so impoverished in RP2, the most contaminated waterbody studied (see Figure 2), that the inclusion of data from this site would have severely skewed the resulting ordinations. To assist in interpreting the patterns shown in the ordination, community summary results were also plotted for each waterbody as the mean total abundance of organisms and mean taxa number, in Figure 2.

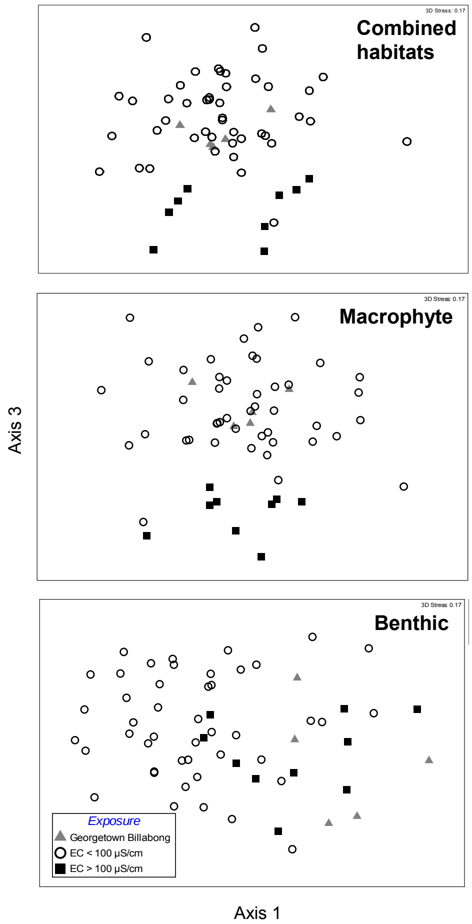

Figure 1 Ordination plots of macroinvertebrate communities from different sites and habitats in waterbodies on or near the Ranger mine site. Waterbodies are classified according to different degrees of exposure to mine waters indicated by electrical conductivity (EC): high for two waterbodies and low for all other waterbodies including Georgetown Billabong.

To simulate 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. As seen in the resulting ordination (Figure 1), the collective macroinvertebrate communities of waterbodies most influenced by mine site inputs (RP1 and Coonjimba Billabong, as indicated by higher electrical conductivity and solid black squares in Figure 1) were quite well separated from the low-EC waterbodies, including Georgetown Billabong. The extreme low-EC replicate that clusters among the high-EC points in the ordination represents a shallow, turbid and feral-animal disturbed location. The interspersion of Georgetown Billabong replicates among the reference waterbody replicates is a similar result to that observed in 1995 and 1996. That is, the composite habitat dataset for Georgetown Billabong indicates that the billabong has not been significantly impacted by mine inputs.

When the macrophyte habitat data were analysed separately, the same ordination pattern arose as was observed for the combined habitat ordination (Figure 1). An additional extreme low-EC replicate was observed to cluster near the high-EC points in the ordination, representing a non-mine location with unexplained low abundance and taxa number.

Apart from RP2, the mean total abundance and mean taxa number for the macrophyte habitat type do not vary markedly between the waterbodies (Figure 2). A similar result was found for combined habitat community summaries (data not presented here). Based on these results, the macroinvertebrate communities from macrophyte habitat (and composited habitats) in Georgetown Billabong are very similar to those sampled from the same habitat in reference waterbodies.

Overall, the diversity of organisms in the benthic habitat for all waterbodies (including unimpacted reference waterbodies), is much lower compared with the macrophyte habitat (Figure 2).

The data for the benthic habitat type were much more scattered with much less separation of higher EC mine-exposed waterbodies from other waterbodies, and with considerable overlap among replicate sites from the different exposure types in ordination space. This was accompanied by greater within-waterbody scatter of replicate sites reflecting higher biological variability (Figure 1).

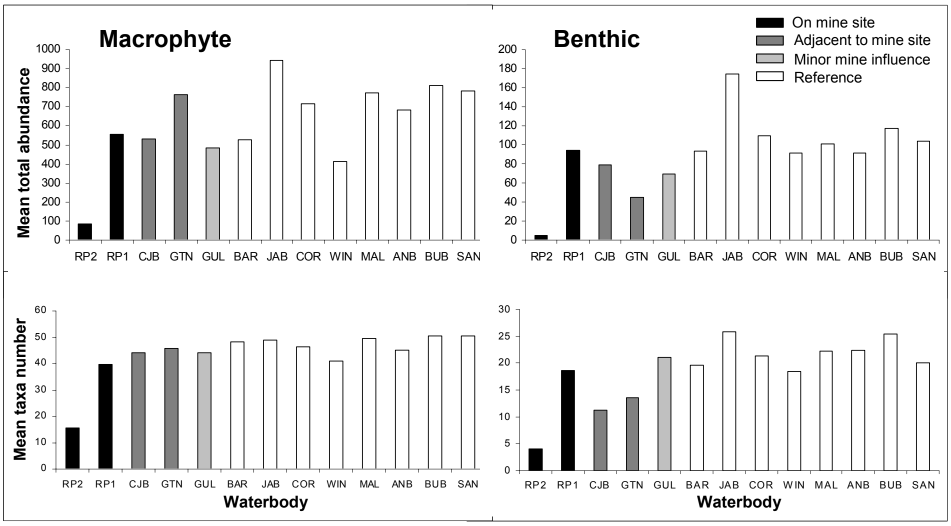

Figure 2 Histograms of mean macroinvertebrate abundance and taxa number among waterbodies on or near the Ranger uranium mine site. Histogram shading depicts the gradient of exposure to mine waters, from most contaminated ('on mine site') to no exposure ('reference'). Site codes are Ranger Retention Pond 2 (RP2) and Retention Pond 1 (RP1), Coonjimba (CJB), Georgetown (GTN), Gulungul (GUL), Baralil (BAR), Corndorl (COR), Wirnmuyurr (WIN), Malabanjbanjdju (MAL), Anbangbang (ANB), Buba (BUB) and Sandy (SAN) Billabongs and Jabiru Lake (JAB).

Unlike the results for the macrophyte and combined-habitat data, the benthic communities from Georgetown Billabong cluster among those sampled from the benthic habitat from more impacted waterbodies, most notably Coonjimba Billabong. Apart from RP2, the mean total abundance and mean taxa number for Georgetown benthic communities were among the lowest recorded from the waterbodies (Figure 2).

In summary the data indicate 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 higher EC waterbodies. There could be a number of reasons for why the benthic data for Georgetown Billabong appear to indicate a greater degree of disturbance compared to the reference billabongs, including (i) contamination of sediments from mine-derived constituents, and/or (ii) physical and chemical attributes of the sediments that are unrelated to mining, as follows.

In relation to possible contamination of sediments from mine-derived constituents, concentrations of uranium (U) in the sediments of Georgetown Billabong prior to open-cut mining at Ranger (1978) were higher than values reported in adjacent billabongs (Noller & Hart 1993), presumably a consequence of the very close proximity of the billabong to surface expressions of the original orebody. Up to 1995 at least, U concentrations in Georgetown sediments were still similar to the background values (Jones & Eames 1997). Whether the concentration of U in the sediments of Georgetown Billabong has increased since 1995 will not be known until results of more recent sediment quality sampling become available. Whether the background concentrations are sufficiently high to elicit toxicity is also not known.

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 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, clay sediments that characterise the littoral benthos of Georgetown and Coonjimba Billabongs provide less habitat 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 (Figure 2). The cracking clays present in the former two billabongs in particular, are both very low in moisture content in the dry season and form a compacted substrate even months after wet season inundation, inhibiting dry and wet season residence by invertebrate life stages respectively. 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 recent leaf-fall from *Melaleuca* trees that closely abut the water's edge in this billabong (ie more so than for the other waterbodies). Leaf-fall to wetlands can adversely affect the benthos through production of toxic tannins or other compounds, deoxygenation of waters or suppression of other important ecosystem functions (eg Bailey et al 2003). The possibility of similar inhibitory effects upon benthic organisms of Georgetown Billabongs is an aspect that requires further study.

Future investigations will focus on better quantifying and describing the physical and chemical nature of sediments from the various waterbodies as an aid in interpreting the patterns observed in the present study. The concentrations of U in the macroinvertebrates may also be measured. U content could provide an indication of the extent to which the organisms are accumulating U from their surroundings, and the levels of U may provide an indication of whether this could be a cause for the lower abundance and diversity in the benthic macroinvertebrates from Georgetown Billabong. The outcome from these more detailed assessments will indicate if closure criteria may be needed for sediments as well as water.

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

C Humphrey, I Hollingsworth[1], M Gardener[2] & G Fox

Background

Characterisation of plant communities from appropriate natural analogue sites will assist in selection of species for revegetation of the Ranger mine landform following rehabilitation of the site. The characteristics of these communities also assist in developing performance measure targets against which the success of revegetation can be tracked by the post-rehabilitation, vegetation monitoring program. 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) need also 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 studies are located on the ancient weathered Koolpinyah landscape surface, with differences 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 adjacent to and south-east of the Ranger mine. A deliberately broad range of vegetation environments was covered by this 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 Ranger operational area.

The ***eriss*** program also included areas adjacent to the Ranger mine (on 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. The underlying geology of these hill sites encompassed quartzite, sandstone and schist mineralogies (Brennan 2005).

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Progress to date

Work to date has involved (1) classification of analogue vegetation communities 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.

1 Analogue vegetation analysis

Data for trees and shrubs from 38 combined EWLS and ***eriss*** sites have been analysed (Humphrey et al 2006, 2007). Pattern in the plant communities was examined using multivariate classification techniques with the resulting dendrogram shown in Figure 1. Of the five natural groupings arising from the classification (C1–C5, Figure 1), three of the dominant groups (C1–C3) were consistent with those from existing classifications of the region's vegetation units, published by Schodde et al (1987). The three groups are described as dry mixed woodland, mixed woodland and *Melaleuca* woodland, according with Schodde and co-workers'(1987) open forest, woodland and myrtle-pandanus savannah vegetation units respectively – see Table 1. The dominant, mixed eucalypt communities from the Georgetown analogue sites surveyed by Hollingsworth et al (2003) were represented in the two main classification groups (mixed eucalypt communities) identified by multivariate analysis of the broader regional group of sites. This indicated that the Georgetown analogue sites adequately described the eucalypt community types found in the region.

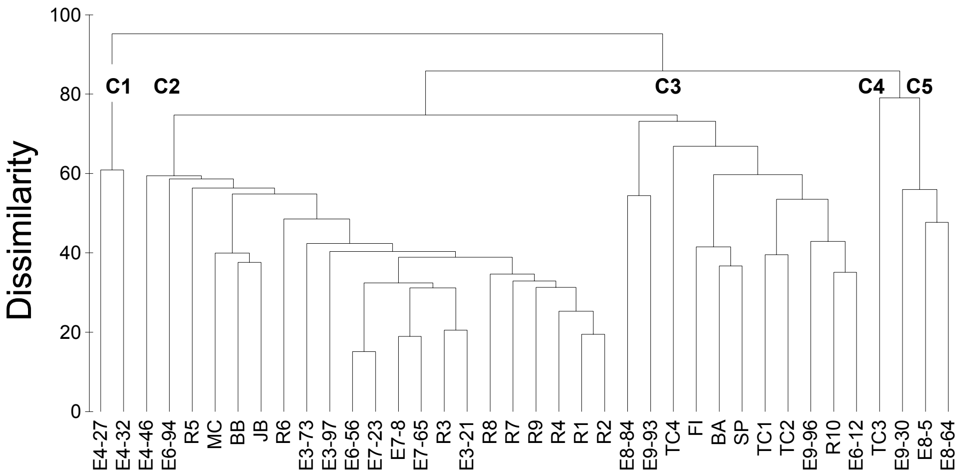

Figure 1 Cluster analysis (group average linkage) of combined ERISS and EWLS 'trees and shrubs' data from different ARR landforms (data log transformed density/hectare units)
Key to site codes:
'E' sites = EWLS Georgetown analogue site,
'R' sites = ***eriss*** lowland Koolpinyah sites around Ranger
'TC' (Tin Camp Creek) and 'F1' (Fisher) = schist hills
JB, BB, BA, SP = sandstone hills
MC = quartzite hill

The analysis described above showed that the composite dataset from ***eriss*** and EWLS encompassed the diversity of both plant community types and environmental conditions that may potentially arise in the final Ranger landform, and hence provides a good basis for species selection. *Melaleuca* woodland sites, however, were not well represented in the dataset which was analysed and hence in the subsequent classification – see Table 1.

Hollingsworth et al (2006) have subsequently described ecological attributes of each of the three dominant community groups described above using species phenology, including growth form, life history, time to maturity, response to fire, type of re-sprouting and deciduousness. In general, all three broad vegetation community types identified in Table 1 have similar attributes, ie an even mix of tree and shrub species, comprising mostly long-lived perennials able to re-sprout after fire. The only attribute that differed was deciduousness, with the drier community having a greater proportion of deciduous species.

Table 1 Descriptions of three broad analogue communities and the matching vegetation units according to Schodde et al (1987)

Broad vegetation community	Dominant tree species	Classification unit from this study (Figure 1)	Vegetation units used by Schodde et al (1987)
Dry mixed Eucalypt woodland	*Xanthostemon paradoxus* *Erythrophleum chlorostachys* *Corymbia foelscheana* *Eucalyptus tectifica*	C3	Open forest
Mixed Eucalypt woodland	*Eucalyptus tetrodonta* *Xanthostemon paradoxus* *Corymbia porrecta* *Eucalyptus miniata*	C2	Woodland
Melaleuca woodland	*Melaleuca viridiflora*	C1	Myrtle-Pandanus Savannah

A plant location database of all known records of vascular plants on the Ranger project area has also been developed by EWLS (unpublished database 2007). At least 500 species have been recorded on the lease, of which 140 species are shrub or tree species with the remainder being grasses, herb, sub-shrubs and climbers. This database will facilitate collaboration between stakeholders and inform revegetation planning.

2 Gathering of environmental and additional plant survey data

Phase 1 of the project described above revealed two knowledge gaps: (i) additional vegetation plots were needed to characterise the *Melaleuca* community group (only three plots were represented in the original survey and subsequent classification); and (ii) the need for a soil characterisation survey to define the range of environmentally relevant (physical and chemical) characteristics of the soils at all of the analogue locations.

The soil characterisation component of this phase of the project was completed in September 2006. Soil physical properties and in situ water infiltration rates at various profile depths were measured. Soil profile samples were collected for analysis of metals and nutrients. Two additional *Melaleuca* community sites were also surveyed in the Georgetown analogue area in June 2007. These additional plant and environmental data are currently being compiled for data analysis.

It is proposed that the original plant survey data be re-modelled by incorporating the new plant, soils and additional environmental data to provide enhanced information about the relationship between vegetation type and soil profile condition. These data have the potential to be used for both design of the surface growth medium of the landform and well as being used to specify target closure attributes for the landform.

3 Modelling plant and environmental data

Considerable progress has been made towards identifying the major 'framework' plant species that should (based on occurrence in the analogue areas) be used for revegetation of the final Ranger landform. For this analysis, an array of environmental data gathered from field surveys or from (desk-top) terrain analysis of digital elevation models were selected that were relevant to climate and water balance, local topography, parent material and soil morphology, as well as fire disturbance. Multivariate gradient analysis and modelling were applied to derive quantitative models for important framework species and highly correlated environmental predictors. Some major findings from this analysis (Hollingsworth et al 2006) include:

- identification of several framework species useful for early establishment because of their broad environmental tolerances in woodland communities, or occurrence in wet areas;
- identification of framework species associated with stabilisation of areas of higher erosion risk in the natural landscape, and hence likely to be suitable for use in those areas of the rehabilitated landform at greater risk of erosion;
- the importance of relief, slope and curvature in determining the occurrence of plant community types and hence an appreciation of how the landform will need to be designed to facilitate establishment of the required vegetation types.

As discussed above, these and complementary data analysis procedures will need to be re-run to include the newly-acquired plant and environmental (soils) data. The analyses also need 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. The results of these and related analyses will enable the development of a detailed revegetation and adaptive management plan based on: (i) accepted revegetation criteria; (ii) capacity to manage species presence and diversity using fire; (iii) capacity to control erosion; and (iv) capacity to manage weeds.

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Hydrochemical and ecological processes of constructed sentinel and reconstructed wetlands in the Magela Creek catchment

P Bayliss & DR Jones

Introduction

The aim of this project is to close existing knowledge gaps in hydrochemical and ecological processes of perennial and ephemeral wetlands, in order to specify design and management criteria of constructed wetland filters and, if required, to reconstruct Djalkmara Billabong. A key closure objective is that surface water arising or discharging from the Ranger Project Area (RPA) following rehabilitation must not compromise primary environmental objectives.

The proposed project addresses three critical knowledge gaps with respect to the design and management of wetland filters: (i) hydrochemical – what is the composition of runoff and the flow duration of seepage?; (2) ecophysiological – what is the maximum load of solutes that can effectively be treated during the wet season by ephemeral and perennial wetlands?; and (3) ecological – how can we rapidly establish sustainable perennial and ephemeral wetland ecosystems?

The work will be staged into the following three phases:

i Develop conceptual models for sentinel and reconstructed wetlands and, undertake a literature review of existing knowledge (including the grey literature/data) and a gap analysis. Compile existing chemical limnological and ecological knowledge for a selection of ephemeral and perennial analogue wetlands in the Magela catchment;

ii To close knowledge gaps use results from (i) above to characterise selected wetland analogue sites (eg Mt Brockman Bund, Corridor Ck, RP1/Coonjimba Billabong) and which encompasses riparian feeder creek systems. Closely monitor key hydrochemical and ecological processes, particularly the trophic transport of contaminants, and develop predictive ecosystem models; and

iii If necessary, design and commence experimental constructed wetland work that integrates with, and runs in tandem to, the experimental terrestrial landform plots proposed by eriss perviously and now managed by EWLS-ERA.

Progress to date

Since the last ARRTC progress report this project has been put on hold because the rehabilitation timeline for Ranger has been extended to 2020 from 2011. A delay of up to 5 years can be expected before the project commences phase ii. In the interim phase i will be completed.

Key aspects of native flora seed biology that support Ranger mine rehabilitation

P Bayliss & S Bellairs[1]

Introduction

The aim of this project is to determine seed viability, dormancy, longevity and germination characteristics of native plant species local to the Ranger mine site to ensure reliable germination and promote establishment of diverse local vegetation on rehabilitated landforms at Ranger. ERA/EWLS and SSD are collaborating with Charles Darwin University (CDU) and Traditional Owners (principally through Kakadu Native Plant Suppliers, KNPS) to investigate the seed biology of Kakadu species to identify those with the most potential for rehabilitation use. A broad range of species will be investigated to ascertain which species can be effectively established from seed. Therefore effective techniques will be required to germinate the seeds, whether for direct seeding or for propagation of tube stock.

Progress to date

Seeds for research are being supplied by KNPS, Top End Seeds and Greening Australia. Most seed lots have been obtained from KNPS, but factors such as unusual rainfall patterns and cyclones have prevented KNPS from supplying seeds during some seasons and alternate suppliers have been required.

Since seed responses are highly susceptible to variation in temperature and moisture conditions, controlled conditions in laboratory incubators are necessary to standardise the many factors impacting on seed germination. 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 (MSB) 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.

A pilot investigation of the viability, seed coat permeability and germination of three native grass species, *Heteropogon triticeus*, *Cymbopogon* sp and *Dichanthium fecundum*, was carried out prior to the start of the project. Removal of the covering structures (glumes, palea, and lemma) of the seeds increased germination of the *Cymbopogon* sp and *Heteropogon triticeus* seeds from approximately 50% to 90% and 85% respectively. Removal of the covering structures from the caryopsis did not affect the germination level of the *Dichanthium fecundum* which was approximately 85%.

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The main seed biology project commenced on 3 July 2006. A meeting was held at Jabiru with ERA, ***eriss***, EWLS, KNPS and project staff. Fifty priority species were chosen based on their abundance in the analogue sites (data provided by EWLS & ***eriss***) and their difficulty in propagation (information provided by KNPS). Species were chosen that tended to be perennial and were likely to minimise fire risk when established on the rehabilitation areas.

A seminar and report on the viability, germination and dormancy mechanisms present in eleven species were provided to project sponsors on 23 February 2007. The species in the report included *Alloteropsis semialata, Brachychiton diversifolius, Buchanania obovata, Denhamia obscura, Livistona humilis, Owenia vernicosa, Persoonia falcata, 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. Additionally, a detailed review was completed on tropical seed biology with reference to minesite revegetation (Bellairs 2007).

The research work will continue to test new species and new seed lots of existing species. Considerable literature has been collected on the taxa (although often on southern species of the genera in the priority list) and this will be collated in a project report to be produced in August 2007.

Figure 1 Application of treatments to seeds to determine dormancy mechanisms that limit germination. Once the dormancy mechanism is identified then alternate treatments are identified that can be applied to batches of seeds by Kakadu Native Plant Suppliers (photo: Julie Crawford).

Figure 2 Two common fruit eating birds, the Figbird (*Sphecotheres viridis*) and Pied Imperial Pigeon (*Ducula bicolour*) that were used to assess the affects of bird consumption on seed germination of two local *Ficus* species (photos: Richard Noske)

In addition to the above, a student research project supported by ***eriss*** and supervised by staff from CDU is investigating how frugivorous (fruit eating) birds affect seed germination of local *Ficus* species. *Ficus* or figs are a keystone resource for frugivorous animals in tropical regions worldwide, and are thus potentially useful in rehabilitation programmes for both attracting birds that will disperse more seeds into a site and as a food resource for fauna.

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. Seed retention time was recorded to determine the likely dispersal distance of consumed seeds. Seeds were also treated with concentrated sulfuric acid (H_2SO_4) to ascertain if artificial procedures could mimic the action of gut passage on seeds. The experimental work for this project has just been completed and the results will be presented at the next ARRTC meeting.

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Bioaccumulation of radionuclides in terrestrial plants on rehabilitated landforms

B Ryan, P Medley & A Bollhöfer

Introduction

The aim of this project is to develop a radiation dose model for radiation exposure through consumption of terrestrial foodstuffs growing on the rehabilitated landform. Up to early 2000, work focused on aquatic animal and plant species due to the identified importance of the downstream aquatic transport pathway during the operational phase of uranium mining operations (Martin et al 1998, Martin & Ryan 2004). With the rehabilitation of the Ranger site there will be radiological protection issues associated with the land use by local Aboriginal people and a shift towards near field terrestrial food sources. Consequently, there has been an emphasis over the past few years on collection of food items such as fruits and yams and terrestrial animals and investigation of radionuclide uptake by them. A study by Ryan et al (2005) has shown that the majority of the contribution to committed effective dose from the ingestion of long lived members of the uranium decay series in fruits and yams is from ^{226}Ra and ^{210}Po. This is primarily due to the relatively high dose conversion factors of these two radionuclides. Consequently, research efforts focus on determining concentration factors in terrestrial plants for ^{226}Ra and ^{210}Po.

Highest dose rates to humans will be received from the consumption of foods from the vicinity of the contamination source. Therefore, the dose assessment needs to be site specific, and the radiation dose model has to include local habits and human land use, and land use expectations, for the site. Apart from information on the uptake of radionuclides in terrestrial plants, estimates of terrestrial bush food consumption by local Aboriginal people and site occupancy estimates are needed to enable a radiological risk assessment to be carried out for the rehabilitated site at Ranger.

Results

Radium uptake in terrestrial plants

A study was initiated to determine radium sources and radium uptake via the ^{226}Ra/^{228}Ra activity ratios measured in terrestrial (bushtucker) flora samples and associated soils. Sequential soil leach extractions were used to identify the geochical phase that contained the most bioavailable fraction – identified by comparing the ^{226}Ra/^{228}Ra activity ratio in the bushtucker with that in the different soil leach fractions. The study takes advantage of a new method that has been developed for the determination of ^{228}Ra activity concentration in environmental samples to address problems encountered with an earlier chemical separation procedure. This new method utilises the radium-containing material produced via the Sills method (Medley et al 2005), with ^{228}Ra being measured via either the ^{228}Ac gamma lines in the sample at highest efficiency or, after digestion of the filter, via the measurement of the alpha decay of ^{228}Th (Medley 2007) if ^{228}Ac activity is insufficient.

The uptake of radium from different soil fractions and concentration factors related to available radium in soils for a species of bush passionfruit (*Passiflora foetida*) was determined by a pilot study in 2006–07. This species was chosen for a number of factors: it is consumed by local indigenous groups, is commonly eaten by children (which is important because dose conversion factors are significantly higher for children than for adults (ICRP 1990) and it exhibits high concentrations of radium relative to other foods. Passionfruit grows well in many different contaminated sites in the Alligator Rivers Region, and since the plant is shallow rooted it is more likely to have a higher uptake of radium when growing in areas where contaminated material is retained in the surface layer (for example, land application areas).

The following sequential leaching steps were applied for the study:

Step 1: 1:5 – Soil:Water

Removes loosely adsorbed radium ions that can be easily desorbed from the soil matrix (eg $RaCO_3$). This can be termed 'readily available fraction' to plants, and may be related to the portion, which can be readily mobilised by freshwater from rainfall events.

Step 2: 1:5 – Soil:1 M $CaCl_2$

Similar to step 1. May be related to the fraction that can be mobilised by water retained in the soil/water matrix for a period of time after rainfall events, and often termed the 'exchangeable' fraction.

Step 3: 1:1 – Soil:1 M HCl

This will dissolve the acid-soluble forms of radium co-precipitated with iron and manganese oxides. Dissolution of ferric hydroxide can be taken as a criterion of the completeness of acid soluble radium extraction (Benes et al 1981) in this step. A method by Allen (1993) utilising a dilute1 M HCl step is also recommended in the Australian Water Quality Guidelines to provide a measure of the fraction of sediment-bound metal that may be available to aquatic organisms

Step 4+5: 1:2 – Soil:0.1 M EDTA with 1.7 M Ammonium hydroxide

This fraction represents that portion that can be mobilised through the activity of complexing agents and is assumed to bring into solution the remaining radium in the soil that is not bound in the more labile phases attacked by Steps 1 to 3. Radium is believed to be primarily $RaSO_4$ in this fraction due to strong co-precipitation of radium with $BaSO_4$, the high sulfate content of many soils in the region and the high levels of sulfate in irrigation waters from mining operations (Akber & Marten 1990, Willett & Bond 1995).

Figure 1 shows the results of the leach experiments and associated $^{226}Ra/^{228}Ra$ isotope ratios measured in the various leach fractions and the *passiflora foetida* samples, collected at the Rockhole tailings residue area in the South Alligator River Valley and at the Nabarlek mine, respectively.

The activity ratios measured in these fruit samples closely resemble the ratios measured in the water and/or $CaCl_2$ leach. It may therefore be more appropriate to determine concentration factors based on the radium activity concentration in the water/$CaCl_2$ leach fractions, which appear to ultimately be the source of radium in the fruit.

Investigations are ongoing and more samples and associated leachates are being analysed from the Ranger land application areas, soils in the Gulungul catchment and an undisturbed sites downstream Magela Creek on the Ranger lease. However, ^{228}Ra activities in some of the leachates are low and the ^{228}Th ingrowth method and subsequent measurement via alpha spectrometry needs to be used for the determination of ^{228}Ra (Medley 2007).

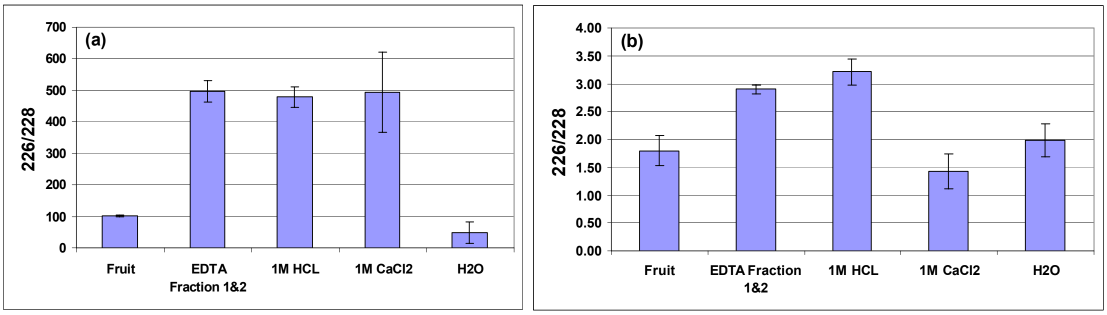

Figure 1 Cumulative $^{226}Ra/^{228}Ra$ activity ratios in various leach fractions – for example, the $^{226}Ra/^{228}Ra$ activity ratio given for the 1 M $CaCl_2$ fraction is based on the sum of activities from both the 1 M $CaCl_2$ and H_2O fractions. Samples were investigated from (a) the Rockhole residue site and (b) the rehabilitated Nabarlek mine.

This is a much slower method of analysis than can be used for the higher activity samples. It is likely that final results will be available by March 2008. However preliminary data indicate that the variability in site specific concentration factors for the uptake of radium in *passiflora* on the Ranger lease is reduced by 1 order of magnitude when calculating the uptake of radium relative to the water and $CaCl_2$ leach fraction rather than by using the activity measured in the whole soil.

Estimate of the diet of local Aboriginal people

A questionnaire has been developed and was distributed to local Aboriginal people. Dietary information was collected, with the view to integrate the information in an updated dose assessment model for the rehabilitated Ranger mine. In addition, information was received from a local supplier of meats to Aboriginal outstations, on buffalo, wallaby and geese consumption. In addition, knowledge gained after many years of observation and discussion with local Aboriginal people has been incorporated in the model of the diet. The information was then collated in tabular form and fed back to local people for comment. Based on this information the diet of local Aboriginal people has been approximated and is shown in Table 1.

The protein intake due to the consumption of traditional foods is significantly lower than the 281 kg previously published by Koperski & Bywater (1985). This is in particular due to the reduced consumption of buffalo meet in the region, following the buffalo eradication program in the 1980s. Johnston (1987) has estimated an annual intake of 220 kg of buffalo flesh, significantly lower than the estimate of Koperski and Bywater (1985).

A recent estimate of the diet of people living in the Gunlom Land Trust area (Bollhöfer et al 2002) estimated an annual bushfood intake of approximately 380 kg. In this assessment the intake of buffalo flesh was estimated much lower at 50 kg, whereas it was assumed that the consumption of wallaby and kangaroo is higher at 110 kg per year. The overall higher intake of traditional bushfoods compared to the people living in northern Kakadu is thought to be due to the distance of the Gunlom Land Trust group from major shopping centres and thus a lower consumption of shop bought foods.

Table 1 Estimate of the annual intake of bushfood of local Aboriginal people in northern Kakadu, shop bought food not included. Previous estimates for the region in the vicinity of Ranger mine (Koperski and Bywater, 1985) in brackets.

Food item	Flesh	Organs eaten	Kg/yr per person
Buffalo flesh	✓		146 (281)
Buffalo kidney		✓	18
Buffalo liver		✓	18 (20)
Wallaby	✓	✓	20 (no data)
Pig	✓		25 (25)
Magpie goose	✓	✓	20 (25)
Fish group 1	✓	✓	10 (60, all fish)
Fish group 2	✓	✓	20
Mussels	✓		4 (4)
Turtle flesh	✓		5 (10, all reptiles)
Turtle liver		✓	0.5
Filesnake	✓		3
Crocodile flesh	✓		2
Goanna	✓	✓	2
Yams	✓		20
Fruit	✓		3 (10, red apple)
Water lily	✓		3 (15)
Food total			320 (450)

Steps for completion

It is important to note that the diet shown here represents a ‘model’ diet, which may be profoundly different for some individuals in the northern Kakadu Aboriginal population (some people will favour certain food items over others). However, this diet is used as a best estimate of the average annual intake of bushfoods, and will continuously be updated as more information becomes available. Using this model diet, and concentration factors estimated for various key food items and radionuclides, the dose received from the ingestion of traditional Abriginal foods will be determined.

The technique to determine radionuclide uptake in terrestrial flora needs to further be refined and tested. In particular, radionuclide uptake in terrestrial fauna needs to be quantified, to allow reliable estimates to be made of the radionuclide intake due to the consumption of traditional foods, during and after rehabilitation of the mine.

Acknowledgments

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

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Groundwater investigations at Ranger

B Ryan & A Bollhöfer

Introduction

The groundwater monitoring program aims at investigating and understanding the sources and dispersion of contaminants at Ranger via the groundwater pathway. This is important from both a hydrogeological and radiological perspective. Once the aquifers have been characterized, in terms of distribution and transport of contaminants and aquifer interactions, models can be developed that predict future groundwater quality, solute transport and its impact onto surface water before and after rehabilitation, respectively. ***eriss***'s radionuclide data can be used to validate hydrogeological transport models. Radiological data, in particular isotopic ratios of uranium (Osmond & Cowart 1982) and/or radium isotopes (Martin & Akber 1999) can be used to characterise sources and transport mechanisms, or water-host rock interactions. With the new issues of tailings repositories emerging, and a further lift of the tailings dam wall imminent this year, the focus is on the analysis of data from monitoring bores in the vicinity of pit 1 and 3, without neglecting important bores in the vicinity of the tailings dam.

Solute balance work was reported to ARRTC16 by David Klessa from Earth Water Life Sciences (EWLS) Darwin. His work indicated that there is a significant discrepancy between the estimated loads of some solutes that are leaving the site, and the loads that are being measured at 009, downstream of the Ranger mine. Alluvial flow of solutes is one process that could cause such a discrepancy. This also means that a significant component of minesite inputs to Magela Creek may currently be undetected, and the consequences could potentially manifest further downstream where the Creek enters the floodplain. Consequently this year, apart from the analyses of routine groundwater monitoring samples collected by the Northern Territory Government Department of Primary Industry, Fisheries and Mines (DPIFM), a study was instigated that aimed at identifying alluvial flow into Magela Creek from the Ranger land irrigation areas, using an EM31 electrical conductivity meter. The EM31 maps groundwater contaminants and salt loads via the measurement of changes in soil electromagnetic inductivity, with an effective exploration depth of approximately 6 metres. This initial investigation was undertaken to profile alluvial water along the Magela channel and find out whether source inputs of solutes from the Magela and Djalkmara land application areas may be entering the alluvial flow system as a density plume.

In order to be able to independently monitor specific bores, ***eriss*** has also acquired bore water sampling equipment, which has successfully been tested at the South Alligator River Valley groundwater monitoring bores in the vicinity of the planned containment near the former El Sherana airstrip.

Results

Interim investigations to profile alluvial water chemistry at points along the Magela channel were conducted in November 2006, using two EM31 conductivity meters (Figure 1). Three grids were studied.

Grid 1: between the Jabiru East application area and Conjiimba Billabong, Grid 2: covering the Djalkmara and Djalkmara Extension land application areas and

Grid 3: between the Magela land application area and the creek channel.

In addition, two transects were surveyed, transect 1 across the Magela Channel downstream of the Ranger mine, and a second transect in the Magela channel itself, upstream of the Georgetown Billabong confluence (Figure 2).

Figure 1 Surveying using the EM31 conductivity meter

Figure 2 Ranger Land Application Areas (black) and EM31 survey grids (grey)

Due to discrepancies between the calibration of the two instruments, quantitative data characterisation is difficult, and further data analysis may be required. A localised area of high conductivity was identified within grid 2 of the survey, between the Djalkmara and Djalkmara Extension land application areas. Further investigation is needed, however, to identify this area as a true plume, a cultural feature or a geological fault line.

Groundwater samples have continued to be collected by the Northern Territory Department of Primary Industry, Fisheries and Mines (DPIFM) this financial year, and aliquots have been supplied to the Environmental Radioactivity group of ***eriss*** for analyses of radionuclides and metals. Heavy metal concentrations are determined via ICPMS-OES. Radionuclides of interest are radiochemically separated from the sample, and measured via alpha spectrometry. The list for U-series radionuclides potentially contaminating the groundwater is: ^{238}U, ^{234}U and ^{226}Ra. The ^{234}U/^{238}U acitivty ratio can be used to identify mine related sources, as they are are likely to exhibit ^{234}U/^{238}U acitivty ratios of ~1 in contrast to most natural groundwaters with activity ratios > 1 (Ivanovich & Harmon 1982).

Time series of >15 years for some bores have been obtained and data have now been collated up to the September 2005. For many of the bores a time series record is not available for this whole period. Some bores were covered by the expanding mine footprint or were abandoned. and other new bores were installed thorugh time.

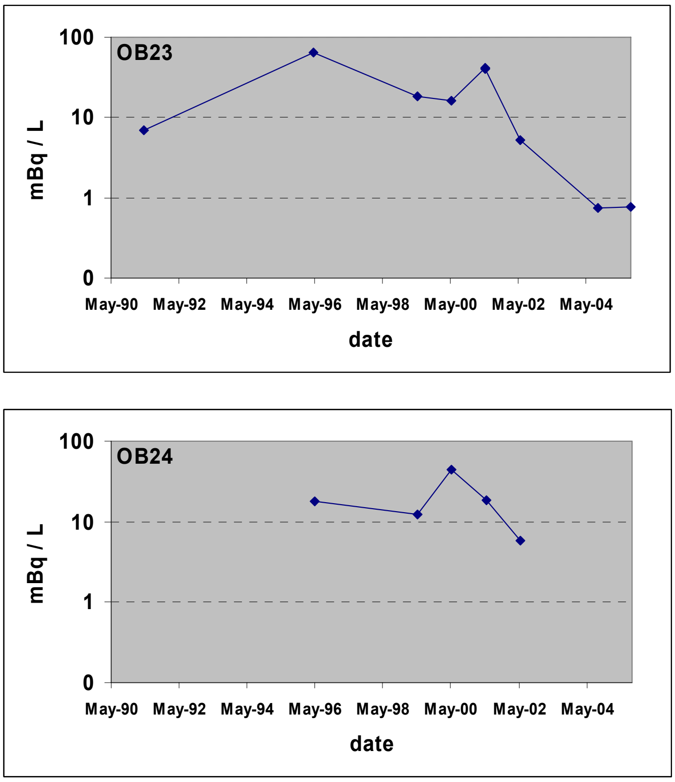

Figure 3 Uranium activity concentration in bores OB23 and 24, north of the tailings dam

Bores close to the northern and north-eastern walls of the tailings dam, have shown increasing uranium concentrations, with bores immediately north of the tailings dam wall showing a dramatic increase following the raising of the wall in 1990. In addition, uranium concentration in observation bores 4A, 6A, 7A, 9A and 10A south and southeast of the tailings dam increased in 1998. The reason for this is unknown but may be due to major works conducted at Ranger, when tailings were transferred from the tailings dam to Pit 1 in 1997-98. In contrast, OB23 and 24, approximately 500 m north of the tailings dam, have shown decreasing uranium concentrations since 2000–2001. Analyses and interpretation of the data is ongoing. However this work was allocated a lower priority in 2006–07 due to installation of monitoring bores at the South Alligator River Valley, testing of the groundwater equipment and other external priorities.

Results of a study of groundwater dispersion at Nabarlek have now been published (Ryan & Bollhöfer 2007). The lessons learned from the study will aid in analysing, interpreting and modelling Ranger groundwater data.

Steps for completion

It is intended to use ***eriss***'s groundwater sampling equipment to monitor targeted bores, after discussion with EWLS, so that monitoring can be continued independently during operations and following rehabilitation. In this context it is important especially to monitor bores north of Pit 3 (in addition to statutory Ranger bore 83/1) to provide a baseline of the alluvial system which is required for the development of water quality trigger values for future monitoring of water quality in bores between Magela Creek and the backfilled and capped pit#3.

More immediately, with the imminent lift of the tailings dam wall, bores in the vicinity should be targeted and sampling of bores 23 and 24 should continue, to investigate whether an increase in bore water activity concentration similar to that observed in bores north of the tailings dam wall in 1990 will occur.

Acknowledgments

The Northern Territory Department of Primary Industry, Fishery and Mining is acknowledged for collection of the bore water samples and providing the aliquots for analysis.

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Development of a spectral library for minesite rehabilitation assessment

K Pfitzner, Bollhöfer & G Carr

Introduction

The aim of this project is to develop a spectral library of land cover components to make recommendations for acquisition of appropriate remotely sensed data for minesite rehabilitation assessment. 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 land cover spectra relevant to the tropical mine environment can be developed.

Because the spectral signatures of land cover components represent complex physical and biophysical relationships, an initial focus in this project was to review and conceptualise the factors that influence field-based spectra (Pfitzner 2005, Pfitzner et al 2005). The concept of collecting consistent and accurate spectral data while minimising the influence of potential extraneous variation is relevant to field spectrometry in all environments, but these methods are rarely reported (Pfitzner & Carr 2006). Consequently, SSD then designed and implemented standards for the collection, documentation and storage of spectral data and metadata. The standards were developed to enable a consistent and repeatable method that minimises the influence of extraneous factors in spectral measurement, and are reported in Pfitzner et al (2006, 2007).

To populate the spectral library, reflectance characteristics over the visible to shortwave infrared (350-2500 nm) using a FieldSpec-FR (ASD Inc 2004) of weed and native ground covers are sampled fortnightly from permanent plots around the greater Darwin region. A challenge in the early project design phase was to locate sites with homogenous dense cover that were unlikely to be disturbed from threats such as fire, development or mowing. Replicate plots were established with support from Commonwealth and Northern Territory Government Departments and private industry, and priority species were identified with stakeholders. Measurements are now fortnightly taken at Crocodylus Park, CSIRO and Berrimah Farm in Darwin. The measured spectral data are supported by metadata describing the viewing and illumination geometries, environmental conditions and state of the target measured.

Results

In order to investigate the spectral responses of ground cover vegetative species over the year, the ***eriss*** database now contains over 500 spectral sampling sets from around 30 native and introduced vegetation species which are sampled every fortnight. It has been shown that while spectra from one species may have a similar overall shape and position of absorption features, the depth and width of absorption features and the magnitude of reflectance change as the sample senesces over time. In addition, the intensity of water absorption features also change over time. Whether or not these changes are a result of biophysical changes of the target or attributable to the illumination conditions can only be assessed by an increased length of the sampling record. It is thus only with accurate spectral and metadata collection that both averaged 'reference' spectra and any significant temporal change in spectral response can be identified.

The Research Scientist of this project has been absent on maternity leave, and as such the project continues to gather data and make progress, but will move to the next stage of analysing the data and producing 'reference' spectra in 2007–08.

Field collection methods

Basic field techniques remain the same as reported at earlier ARRTC meetings. Minor changes include two 20mW powered lasers mounted either side of the foreoptic, allowing the operator to more accuarately pinpoint the sampling area. Scaled close-up photographs of the sampling area are now captured allowing further analysis of the spectral signature.

Challenges

Time and effort has had to be put in to prevent grazing animals from consuming the vegetation within the sampling plots. As the areas surrounding the plots are grazed, the healthy vegetation within the plots becomes more enticing for stock. Fences have had to be reinforced numerous times in order to prevent intrusions. Some plots, particularly those at Crocodylus Park, had to be abandoned due to extensive earthworks on site.

The dynamic nature of weeds has also provided challenges. Thick growth of one species of weed with ample seed distribution within the plot does not guarantee that the same species will proliferate during the next season. The variance of build-up conditions, eg early or sporadic rains, can favour a particular species over another, and can out-compete the species growing within the plot.

Spectral Database

The number of spectral samples now exceeds 500. An additional field in the meta-database named: 'Visibility' will be added, allowing the atmospheric conditions to be quantifiably measured. Darwin airport, which is within a few kilometres of our sites, uses radar to measure the amount of particulate matter in the atmosphere. Visibility is measured in terms of kilometres, with a clear day giving 30 kilometers, and an extremely smokey day being sometimes less than one. These data are available on the Australian Weather News website at: http://www.australianweathernews.com/pub/SR0CAP.HTM. Daily records are archived and accessible and will be added to the database.

Steps for completion

The database to store and retrieve ground-based spectral information and metadata needs to be established. Once this database is established, the data needs 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 on the most appropriate datasets for minesite rehabilitation assessment made.

Acknowledgments

Thanks to Dr Grahame Webb, Charlie Manolis and John Pomeroy (Crocodylus Park), Dr Gary Cook and Rob Eager (CSIRO) and Rob Kelley and Arthur Cameron (Berrimah Farm) for continued support and access to vegetation plots suitable for spectral sampling. Thanks also to Peter Bayliss, Mark Gardener, Jane Addison, Sean Bellairs, Bronwyn Bidoli, Dave Walden and James Boyden for initial discussion of the project.

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Incorporation of disturbance effects in predictive vegetation succession models

P Bayliss, D Walden, J Boyden, M Gardener[1] & A Petty[2]

Introduction

The aim of this project is to develop a predictive vegetation succession model that incorporates the effects of fire and weed disturbances at the landscape scale, to provide the basis for monitoring and assessing revegetation performance within a risk management framework. The project is in two parts. Part 1 will draw on existing knowledge and data to: (i) characterise fire and weed disturbances with respect to frequency, intensity, duration, spatial extent and possible interactions between these parameters; (ii) develop a conceptual model for vegetation succession that incorporates disturbance effects; and (iii) use the conceptual model to commence development of a spatially explicit vegetation succession model. The model will incorporate current knowledge on landscape-vegetation relationships and vegetation dynamics, the starting conditions with respect to agreed establishment species and methods, the desired vegetation end point, and undesirable states that deviate from the desired trajectory pathway. Part 2 will: (i) examine the resilience of vegetation along the successional trajectory by simulating temporal and spatial responses to disturbances using combined individual-based and cellular automaton modelling approaches; (ii) couple the spatially explicit vegetation dynamics-disturbance model with the CAESAR landform evolution model (see KKN 2.1.5) to assess the critical role of changes in vegetation cover and composition over time in stabilising the geomorphic behaviour of the final landform, especially in the first few years; (iii) identify gaps in knowledge that may require focused monitoring studies and/or data from landform-vegetation trials; and (iv) develop an ecological risk assessment and adaptive management framework to reduce risk from ecological disturbances during the rehabilitation phase. Preliminary results characterising fire and weed disturbances on the Ranger Project Area (RPA) and surrounds are reported here.

Progress to date

Progress on characterisation of fire and weeds (Part 1 above) on the RPA and surrounds has been reported to ARRTC previously (ARRTC18) and this task will be completed by November 2007. Part 2 (spatial modelling) will commence in early 2008.

Fire

An initial comparison of fire history attributes based on their mean occurrences on 250 m grid cells within each location (RPA – Jabiluka & Ranger leases; WAL – western Arnhem Land; KNP – Kakadu National Park) between 1995 and 2004 (n=10 y) has been undertaken. Results indicate that the RPA had similar fire histories to the surrounding KNP, and appeared to burn a little less frequently (proportion of years burnt 0.21 cf 0.23, respectively). The greatest contrast was between WAL and the lowlands (0.11 cf 0.18 - 0.24, respectively), being burnt

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about half as frequently. However, that difference has reduced with the inclusion of 2005 and 2006 fire history data encompassing two extensive fires over WAL. Nevertheless, these contrasts can only provide a general picture of fire history across the landscape because the analysis does not account for spatial autocorrelations and scale effects (see previous ARRTC summary). A more appropriate spatial analysis of fire history in relation to landscape attributes was undertaken using a Classification and Regression Tree (CART) approach. The following four explanatory variables were used in a Regression Tree model to explain spatial variations in Landsat-derived fire histories across the study area: distance from road (m); distance from waterways (m); vegetation type ((n=27 or n=6 classes); landform or landsystem type (n=6, table 1a); and land tenure type (n=4: RPA – Jabiluka & Ranger leases; WAL; KNP). Vegetation type was examined using a fine-scale classification system (n=27 classes, Shodde et al. 1987), which was compared to a much broader set of amalgamated classes (n=6, Table 1b, Wilson et al. 1990) used by EWLS in revegetation planning. The following fire history metrics were examined across years and for early and late dry season periods within years: the proportion or percentage burnt; the frequency of burns; time since last burnt (yrs); and mean, minimum and maximum intervals between burns (yrs).

Table 1a & b Landform types and broad vegetation classes used in the CART analysis

Code	(a) Landform type	Code	(b) Broad vegetation class
1	Coastal floodplains	1	Woodland
2	Koolpinyah surface - deeply weathered & eroded	2	Open forest
3	Arnhem Land plateau	3	Low open woodland
4	Alluvial floodplain	4	Floodplain
5	Dissected foothills of igneous origin	5	Melaleuca forest/woodland
6	Koolpinyah surface - sedimentary plains	6	Closed forest

Recursive partitioning (ie the 'regression tree') to each fire history attribute was used to determine which explanatory variables accounted for most of the spatial variation in fire history. Models were generated from a 10% random sample of all available100 m pixels including the total buffer zone area, which were then cross-validated against the remaining 90% of pixels within the dataset (ie a bootstrapping technique). A subset of model results are reported here only (Table 2) because all fire history attributes are highly correlated.

Table 2 Models tested and a summary of overall CART results (total variance explained and model complexity as indicated by the number of classification leaves or nodes)

Model tested	Total explained variance (%)	Number of leaves or nodes
1. Pf = V_6 + LF + LT +DR + DW	64	6
2. Pf = V_{27} + LF + LT + DR + DW	69	6
3. %LDSf = V_6 + LF + LT +DR + DW	19	4
4. %EDSf = V_6 + LF + LT +DR + DW	16	5
5. Pf = V_6 + LF + LT +DR + DW + %EDSf + %LDSf	65	6
6. Pf = V_{27} + LF + LT +DR + DW + %EDSf + %LDSf	70	6

Pf = proportion of years burnt; LF = landform type (see table 1a); V_{27} = Shodde et al. (1987) vegetation classes; V_6 = Wilson et al. (1990) broad vegetation classes (see table 1b); LT = Land tenure or location (RPA-Jabiluka/Ranger leases, WAL, KNP); DR = distance to roads (m); and DW=distance to waterways (m); %LDSf=percentage of total burns in the late dry season; %EDSf=percentage of total burns in the early dry season.

Results (Table 2) show that a large proportion of spatial variation in fire history, as indexed by the proportion of years burnt between 1995 and 2004, was explained by the models combining all predictive landscape variables. Model 2 has about five times more vegetation classes than model 1, yet model performance only improved marginally (65% cf 69%). Although model 1 with only 6 vegetation classes is obviously far easier to interpret across the region, if predictions need to be made at more local scales and/or for specific vegetation communities then the more complex vegetation model (model 2) may be the more appropriate choice. Surprisingly, the seasonal effects of fire, as indexed by the percentage of all fires that occurred in either the early or late dry season, explained little spatial variation in fire history, and this was true whether it was used as a predictive variable or a dependent variable in its own right (Table 2). Classification results for model 1 (Table 3, Figs 1 & solid line in Fig 2) show that landform type explained most spatial variation in fire history, followed by land tenure type, vegetation class and distance to roads. The importance of vegetation class and distance to roads depended on which landform type and which land tenure type. Model 1 certainty is high because validation on the remaining 90% of pixels was high, as demonstrated by the close correlation between relative model error and validation test error (Table 3 and see Figure 2).

Table 3 Summary of CART results for model 1

Split number	Complex parameter (prop. explained variance)	Relative error	Validation test error
1	0.540	1.000	1.000
2	0.037	0.460	0.454
3	0.017	0.422	0.418
4	0.016	0.406	0.401
5	0.010	0.390	0.386
6	0.010	0.380	0.377

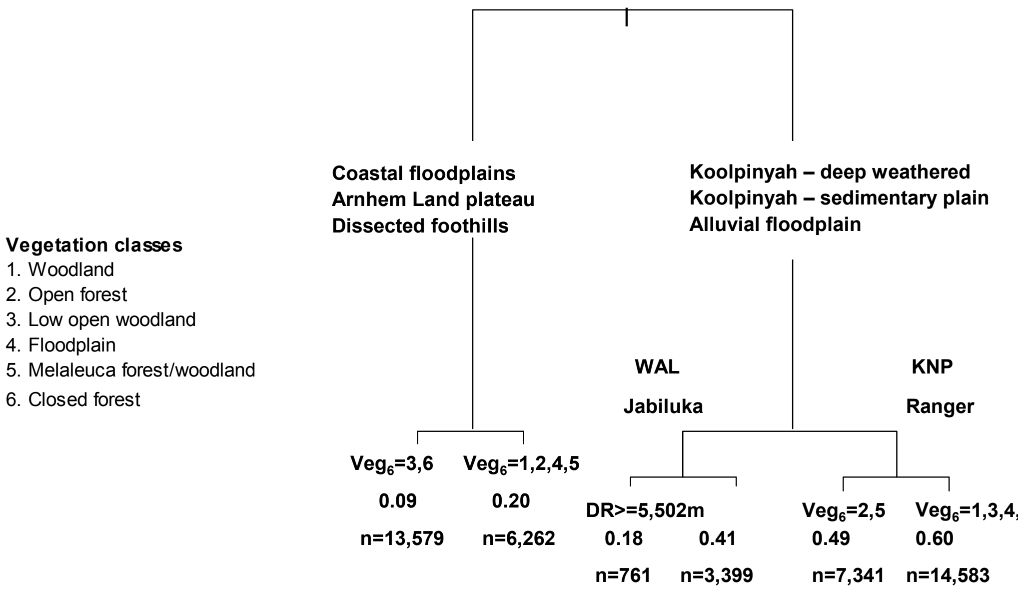

Figure 1 Regression classification tree for model 1. n=sample size (10% of 459,250 pixels or 45,925), DR = distance to roads (m), WAL=western Arnhem Land, KNP= Kakadu National Park, Jabiluka and Ranger mineral leases, Veg_6 = 6 broad vegetation classes (see codes above, from Wilson et al 1990).

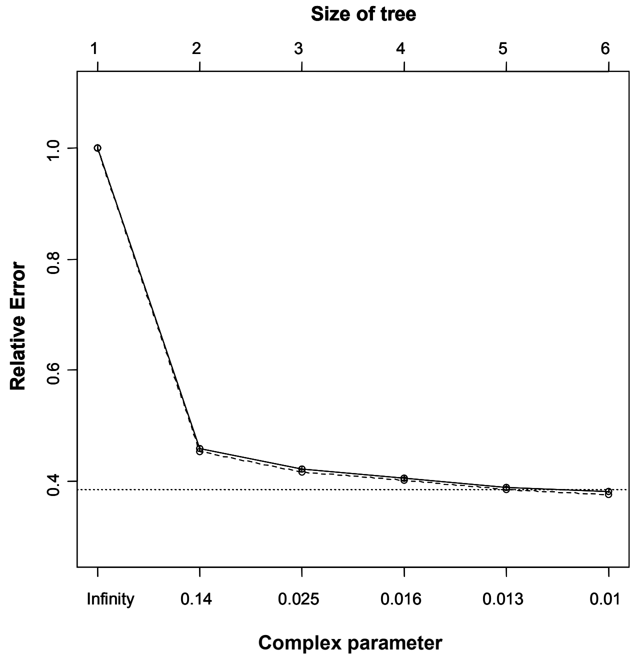

Figure 2 Scree diagram for model 1. Relative model error vs. complex parameter. Size of tree is the number of leaves or splits. Solid line is the model fit based on a 10% sample of pixels (100m grid cells), dotted line (just visible) is the validation or test fit to the remaining 90% of pixels.

Whilst spatial variables performed poorly in explaining fire seasonality, analysis of decadal trends in the extent (km^2) and sequence of early versus late dry season fires across KNP suggests that decadal trends in climate are major determinants of fire seasonality. The influence of climate still manifests despite the fact that extensive prescribed burns are undertaken by park management in the early dry season (EDS) to reduce the occurrence and severity of late dry season (LDS) fires. Cusum (cumulative sum of the mean deviations) trends in the extent of LDS fires (Fig 3a) are negatively correlated to the cusum trends in the extent of EDS fires (Fig 3b), demonstrating that fire management strategies on KNP are working. Cusum trends in the extent of LDS fires are negatively correlated also with cusum trends in rainfall (Darwin), and nonlinearly (concave quadratic) to cusum trends in the Interdecadal Pacific Oscillation (via the Pacific Decadal Oscillation or PDO) and ENSO (via the SOI). A combined multiple regression equation (Table 3) explains 88% of variations in observed cusum trends of the extent of LDS fires on KNP using PDO only (the SOI is a redundant regression variable because it is negatively correlated with PDO). Four regression models were tested before arriving at the final model, hence a Bonferonni correction was applied to the regression P-value to protect against Type I error.

It should be noted, however, that cusum values are smoothed trends about an average (ie stripped of 'white noise' or fine-scale data variability) and, hence, explained variance is expected to be high. Nevertheless, results suggest that, in general, trends in the extent of LDS fires decrease with increasing trends in the extent of EDS fires from prescribed burns or otherwise (ie reduced fuel load), rainfall (eg wetness) and PDO value (ie climatic) conditions other than instantaneous rainfall conditions).

Needless to say, biophysical interactions between extent of seasonal fires, rainfall conditions (which when lagged indexes vegetation growth & fuel load) and climatic conditions other than rainfall affected by the PDO phase, without doubt involve complex nonlinear relationships, hysteresis effects and multiple time lags. Hence the multiple regression model, although highly predictive, should be treated with caution until validated by other forms of modeling that incorporates non-stationary values in variables with multiple time lags. The final phase of Part 1 of this project will develop a methodology to combine both spatial and temporal models in order to characterise, and better predict, the seasonal occurrence of fires across the landscape.

Table 3 Summary of multiple regression results predicting cusum time trends in the extent (km^2) of late dry season burns from cusum trends in the extent (km^2) of early dry season burns (CS EDS), rainfall (CS RF Darwin, mm), and the quadratic polynomial of the Pacific Decadal Oscillation (CS PDO^2). Kakadu National Park, 1980–2004.

R= 0.9380, adjusted R^2 = 88%, n= 25, P Bonferonni < 0.001, SE regression = 1050

Variable	Beta	SE Beta	B	SE B	P
Intercept			980.2	479.1	0.053
CS PDO2^2	-0.48	0.09	-42.7	8.2	<0.001
CS RF	-0.71	0.10	-5174.7	743.7	<0.001
CS EDS	-0.57	0.09	-0.7	0.1	<0.001

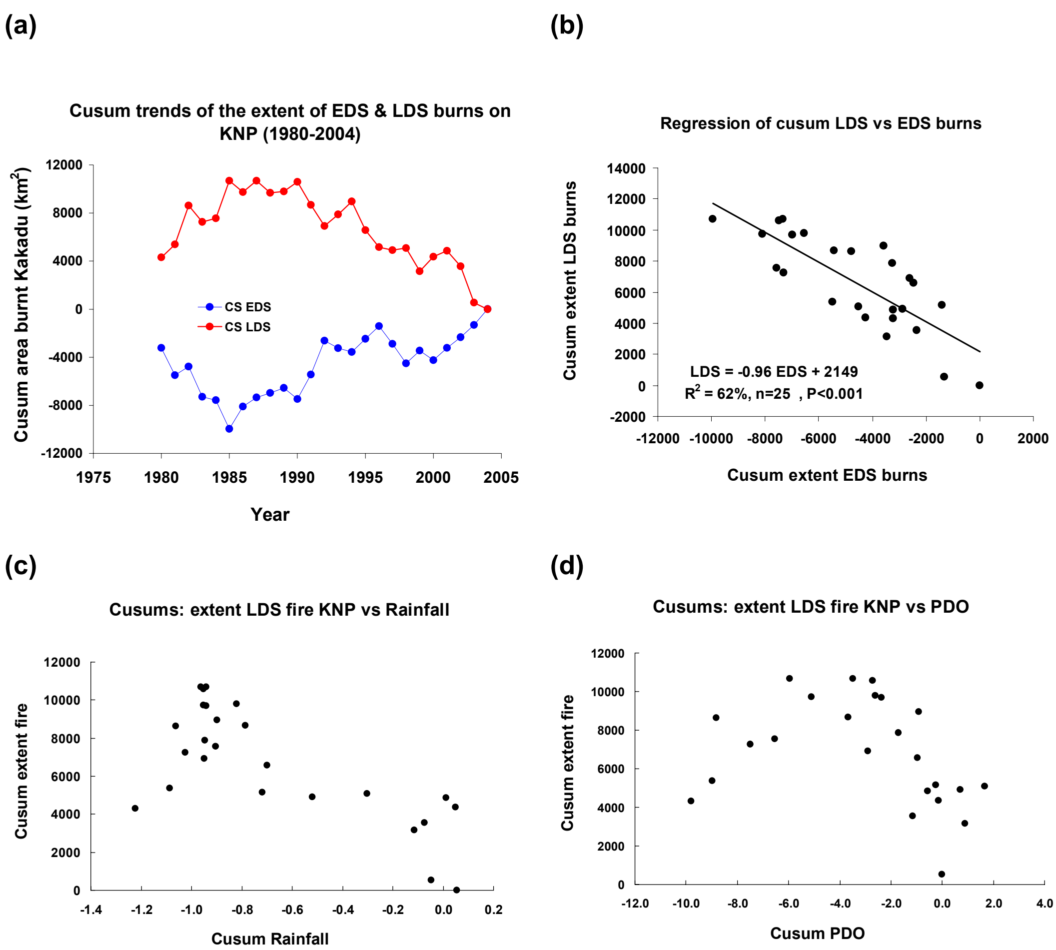

Figure 3a–d Trends in the extent of late dry season (LDS) fires on KNP between 1995 and 2004 in relation to trends in climate using cusum values. (a) Trends in the extent of LDS (top line) and early dry season (EDS) (bottom line) fires over time, (b) negative correlation between the extent of LDS fires and EDS fires, (c) general negative relationship between the extent of LDS fires and rainfall (slightly nonlinear), and (d) strong nonlinear relationship between the extent of LDS fires and the Pacific Decadal Oscillation (PDO).

The availability of finer spatial resolution fire history data derived from Landsat captures allows previous multivariate analyses of all analogue data (see KKN 2.2.2) to be re-examined with fire included as a major ecological factor or component. This work is currently being undertaken by a consultant in collaboration with PAN-Kakadu, EWLS and NT Bush Fires Council. An EWLS collaborative project that involves Gundjeihmi Aboriginal Corporation will attempt to align fire management on the Ranger and Jabiluka leases with surrounding KNP by re-introducing traditional burning regimes (Gardener et al 2007).

Weeds

The GIS mapping and monitoring of weeds on the RPA commenced by EWLS in 2005 has been continued and improved over time. Additionally, the GIS weeds database for Kakadu is now continually updated by PAN staff. Characterisation of the potential disturbance effects of weeds to successful revegetation on Ranger is currently being undertaken using available spatial data for the RPA and surrounding Kakadu. However, it is recognised that more systematic weed surveys need to be undertaken on KNP to avoid bias associated with current survey effort being concentrated at control sites, especially with respect to perennial and annual mission grass (*Pennisetum polystachion & P. pedicellatum*, respectively), identified at the Weeds Closure Workshop in November 2005 as being one of the major risks to successful revegetation of Ranger. Potential interactions between weeds and fire will be investigated through a collaborative project between Kakadu-PAN and ***eriss*** using all available temporal and spatial data. A KNP Weeds Experts Panel has been formed to review weeds management issues (including Ranger rehabilitation), strategies, species priorities, operational procedures and control costs and budgeting. The panel includes specialists from ***eriss***, ERA-EWLS, CDU, NTG and Indigenous organisations, and will adopt a risk assessment and management approach.

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Quantitative comparison of remotely sensed data for minesite assessment

K Pfitzner

Introduction

The aim of this project is to compare a variety of remotely sensed data for minesite rehabilitation assessment. The challenge in monitoring minesite environments using remotely sensed data is to differentiate cover types with spectral variation across an inherently variable land surface, and over different capture times. The differentiation of introduced weeds, native ground and tree canopy cover and exposed soil is required as these index local environmental conditions in addition to rehabilitation success.

This project will use the analysed ground-based spectral library database to make recommendations on suitable remotely-sensed data for minesite rehabilitation assessment; acquire simultaneously a variety of spatial, spectral and costed datasets, as seen suitable by the ground-based database; analyse and quantitatively compare such data and, perform a cost-benefit analysis, and publish results.

A variety of remotely sensed data have been acquired by SSD. Examples include: Compact Airborne Imaging Spectrometer (CASI) over Nabarlek, DeBeers Hyperspectral Mapper (Ranger, Nabarlek and Rum Jungle), temporal QuickBird™ data (Nabarlek) and airborne gamma-ray surveys (Nabarlek, upper South Alligator River valley and Sleisbeck) (Pfitzner 2005). High resolution data is required because of the variability and short range variation in surface cover, which is typical of the disturbed environment. While high spatial resolution satellite data has been used contextually to temporally identify changes in vegetation cover, such data is limited for species identification due to poor spectral separability.

Results

It has been shown that revegetation at Nabarlek is under pressure from threats such as weeds and fire. While ground-based measurements provide necessary information on species abundance, the method is limited due to small sample sizes and the effort required for monitoring changes in abundance due to disturbance (Bayliss et al 2004a,b). In addition, the qualitative nature of many traditional ecological-based methods may also cause problems with consistency when used by different assessors (Corbett 1999).

Remotely sensed data provides a source of information to monitor broad landscape changes. Due to the short range variation in surface covers of the Nabarlek minesite, however, very high resolution (VHR) data is required to resolve the land cover features required for revegetation assessment. Four band data is limited for spectral separation of land cover features. Until further developments of remote sensing technologies, a contextual approach for separating cover components that index local environmental condition using VHR satellite data has been implemented at Nabarlek and shows promise as a monitoring tool for rehabilitated mine sites in general.

VHR QuickBird™ data, covering 14 km x 45 km was acquired in 2006 after cyclone Monica impacted on the region. Additional QuickBird™ data encompassing the entire northern end of the Magela floodplain was also captured in August 2006. This data will be reported under KKN 5.1.2. Because fire disturbed the Nabarlek landscape prior to the capture of Quickbird data, multi-temporal Landsat TM5 data was acquired to separate the cyclone effects from fire. Landsat TM5 data was acquired pre-fire and cyclone, post cyclone (pre-fire) and post cyclone/fire. Vegetation recovery from the effects of the cyclone and fire in 2006 will be assessed with a late wet season capture in 2007.

Airborne CASI data flown in 2002 at the Rum Jungle mine and gamma survey data from 1996 were integrated to characterise the mine site. Reflectance field spectra were measured in 2002 and in August 2006, to identify different mineral assemblages on site. The results of this study have been published (Pfitzner & Clifton, 2006).

For 2006–07, it was planned to analyse and publish the DeBeers airborne hyperspectral data obtained over Ranger in 2004, using data reduction techniques to prioritise field sampling locations for the spatio-temporal sampling outlined in the Project ‘Development of a spectral library for minesite rehabilitation assessment’. The DeBeers data and EWLS' weed mapping data will contribute in the selection of suitable homogenous dense stands of cover. Due to maternity leave arrangements the major part of this project has been put on hold and publishing of the results from the DeBeers hyperspectral data will be conducted in 2007–08.

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Monitoring of sediment movement in Gulungul Creek

D Moliere, KG Evans & M Saynor

Background

Two gauging stations were installed in Gulungul Creek upstream (GCUS) and downstream (GCDS) of the Ranger mine in November 2003 and February/March 2005, respectively (Moliere et al 2005). These stations supplemented the existing mid-reach station G8210012. Several years of continuous turbidity data, collected using field-calibrated turbidimeters from GCUS and GCDS, will be used to establish baseline catchment condition and assess mine impact using The Australian and New Zealand water quality guidelines (WQG) (ANZECC & ARMCANZ 2000) techniques.

Results

February–March 2007 flood

During the 2006–07 wet season, an extraordinary rainfall event occurred over a 3-day period between 27 February and 2 March 2007 resulting in the highest flood levels recorded within the Gulungul Creek catchment since 1971. The recorded maximum rainfall intensity exceeded a 1:100 y storm event for durations between 6 and 72 hours. The flood peak occurred at all three stations during the afternoon of 1 March 2007 with peak discharge at G8210012 corresponding to a 1:100 y event.

Equipment at GCUS and GCDS were submerged by floodwaters, causing data loss for the flood peak and recession flows. The flood-damaged equipment at both stations is being upgraded prior to the 2007–08 wet season to withstand a high magnitude flood in the future. In addition, redundancy is being built into the systems to ensure that data will continue to be collected in the event of failure of the primary system.

Impact assessment

Mud moves through stream systems in pulses or waves generated by discharge events. Reliable mine impact assessment requires an understanding of the mud loads transported during these pulses. Trigger levels for event-mud-loads based on current, pre-rehabilitation conditions (which can be used for future impact assessment) were derived using two techniques:

1. Before-After-Control-Impact, paired difference design (BACIP) (Stewart-Oaten et al 1986, 1992), and
2. A relationship between event-mud-load and corresponding event discharge characteristics.

BACIP

Event-mud-load data collected during 2005–06 and 2006–07 were used for BACIP analysis. GCUS and GCDS were treated as paired sites. Two methods were used. Firstly, the differences in event-mud-loads at the two locations (Fig 1), and secondly the differences of log-transformed event-mud-loads (Fig 2). The log-transformation is effectively an assessment of the ratio between GCDS and GCUS as shown in the following equation:

$$\log(\mathrm{GCDS}) - \log(\mathrm{GCUS}) = \log\left(\frac{\mathrm{GCDS}}{\mathrm{GCUS}}\right). \tag{1}$$

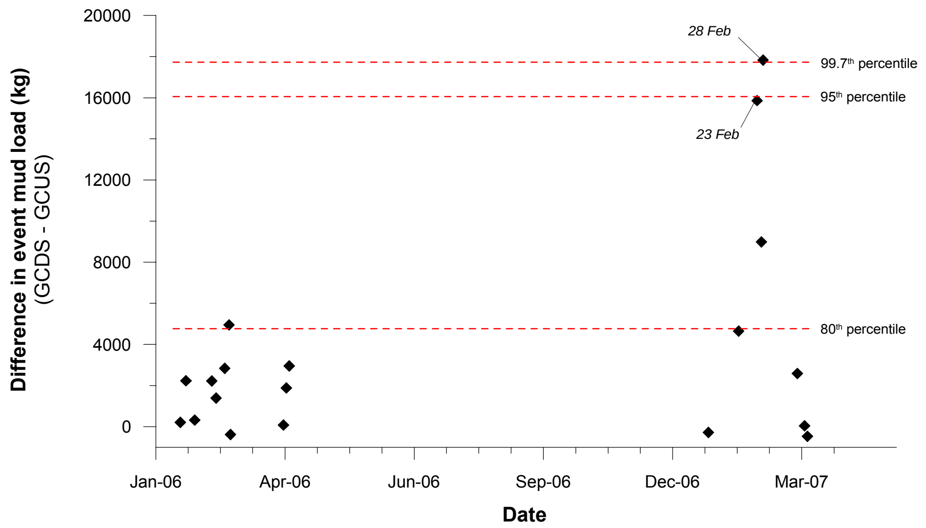

Figure 1 Temporal variation of the difference in event-mud-loads measured at GCDS and GCUS during 2005–06 and 2006–07 (indicated as ◆). The 80th, 95th and 99.7th percentiles of the difference in event-mud-loads are shown as dashed lines for reference.

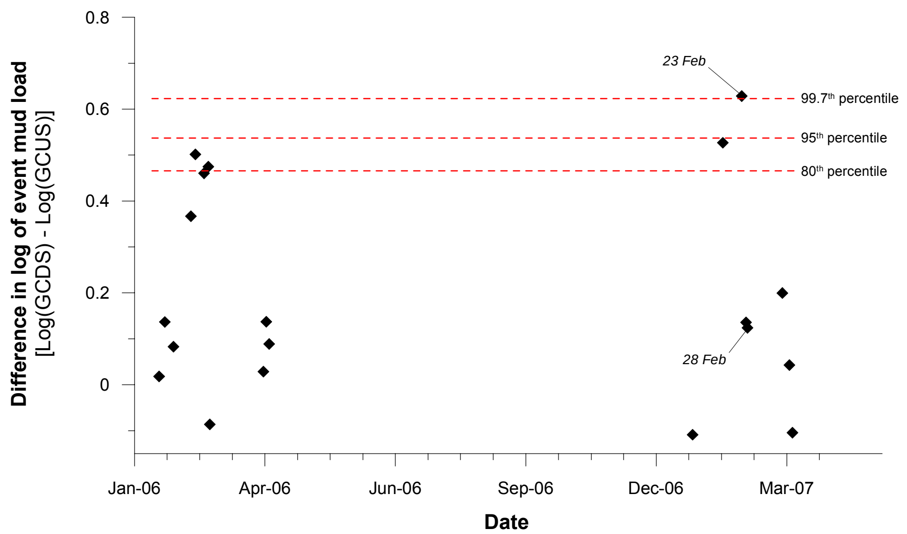

Figure 2 Temporal variation of the difference in the logarithms of the event-mud-loads measured at GCDS and GCUS during 2005-06 and 2006-07 (indicated as ◆). The 80th, 95th and 99.7th percentiles of the difference in the log-transformed event-mud-loads are also shown for reference.

The 80th, 95th and 99.7th percentiles (trigger levels) of the BACIP results can potentially be used to specifiy increasing levels of interventions by supervising authorities and the mining company to control impact. This is analgous to the focus, action, and limit framework establish for management of water quality in Magela Creek.

Relationship between mud load and discharge characteristics

An alternative to the BACIP approach is the use of a relationship between event-mud-load and corresponding event-discharge characteristics. Significant relationships (Eqns 2 & 3) were fitted for each station using all event data collected for a site i.e. a four-year monitoring period at GCUS (2003–07) and a two-year monitoring period at GCDS (2005–07).

$$T_{GCUS} = 88.7 Q_T^{0.37} Ri^{0.82} \quad (R^2 = 0.93;\ n = 50;\ p<0.001) \qquad (2)$$

$$T_{GCDS} = 56.6 Q_T^{0.38} Ri^{0.60} \quad (R^2 = 0.87;\ n = 22;\ p<0.001) \qquad (3)$$

where T is total mud load, Q_T is total discharge during the rising stage of the hydrograph and Ri is maximum periodic rise in discharge over 6-minutes.

Since observed loads are normally distributed around the best-fit line (predicted loads) +1 SD, +2 SD and +3 SD from the 1:1 line were used to derive trigger levels.

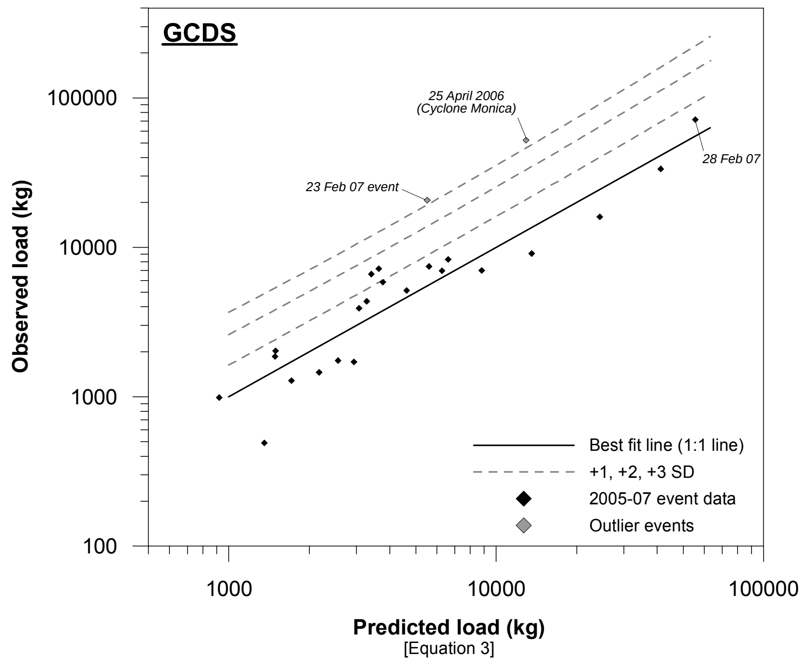

Figure 3 Event-based mud load relationship for GCDS. Event data collected between 2005 and 2007 used to fit the relationships are also shown. The dashed lines indicate +1 SD, +2 SD and +3 SD from the 1:1 line (SD = standard deviation).

Discussion and conclusions

The BACIP analysis using simple comparison indicated that the event on 28 February 2007 (the largest monitored event) had a significantly elevated mud load at GCDS compared to that measured at GCUS (Fig 1). However, the log-transformed difference method indicates that the 28 February 2007 event plots well below the 80th percentile line (Fig 2). The mud load–discharge relationship fitted for GCDS (and GCUS) also indicated that the mud load on 28

February 2007 was not significantly elevated compared to the load predicted using the event discharge (ie the event plots within +2 SD of the 1:1 line) (Fig 3).

This analysis indicates that the BACIP approach using a direct difference comparison between event loads at the two sites is biased towards events with large mud loads. For example, the mud load at GCDS and GCUS during the event on 28 February 2007 was 72 t and 54 t respectively. This corresponds to a difference of 18 t, the largest difference in mud load between the two stations for the two-year monitoring period. However, the mud load at GCDS during this event is only 1.3 times that measured at GCUS and the log-transformed difference value plots well below the 80th percentile for this method of data analysis.

Consider the event on 23 February 2007 where the mud load measured at GCDS of 21 t is more than four times the 5 t measured at GCUS, but the difference is 16 t. This event exceeds the 99.7th percentile for the log-transformed difference method (Fig 2) but is within the 95th percentile for the simple difference analysis (Fig 1). Using the log-transformed difference method, a number of events exceed trigger levels where GCDS loads are relatively much greater than GCUS loads. These same events are below triggers levels using simple differences because they may be relatively small discharge events where total loads are less and subsequently the difference in loads between sites will be small. Alternatively, a large mud load event associated with a high magnitude flood may plot above the simple difference 95th percentile line despite it being a natural, non-impacted event.

Both the log-transformed difference method (Fig 2) and the mud load-discharge relationship fitted for GCDS (Fig 3) identified the event on 23 February 2007 as an outlier. That is, this event had a significantly elevated mud load at GCDS relative to (1) the mud load measured at GCUS, and (2) the predicted load using event discharge characteristics at GCDS, respectively. The mud pulse for this event at GCDS peaked at about the same time as the hydrograph. The sedigraph generally peaks before the hydrograph. As a result of this 'shift' in the timing of the mud *C* peak at GCDS, the observed mud load is elevated compared to that at GCUS, and to the the predicted load for that event discharge at GCDS.

The rainfall data indicates that the storm centre may have been over the mine-site tributaries between GCUS and GCDS. Therefore, it is possible that the contribution of mud load at GCDS from the ungauged, mine-impacted tributaries downstream of GCUS may have been relatively high during this runoff event and subsequently affected the timing of the sedigraph peak. However, this is difficult to clarify given the fact that flow and mud *C* are not monitored in these tributaries. It is recommended that turbidity data be collected at G8210012 to better assess inputs from small streams draining the western side of the footprint of the tailings dam.

The mud load recorded at GCDS on 25 April 2006 during runoff associated with Cyclone Monica plots as an outlier using Equation 3 (Fig 3). Treefall along the channel banks was a differentiating physical factor (compared with the 'normal' riparian conditions) that would have been expected to have resulted in increased stream mud concentrations during the event. Unfortunately, turbidity/mud concentration data were not recorded at GCUS during this event owing to damage to the gauging station (Moliere et al 2007). Since cyclonic treefall was catchment-wide, the mud load at GCUS *may* also have been elevated relative to event discharge characteristics. This shows the value of assessment using the mud load-discharge relationships for a given gauging station. Once a pre-impact relationship is developed for a site, upstream data are not required to assess post-impact conditions. This is a 'before-after' assessment in a temporal rather than a spatial sense. Such an approach is predicated on having obtained a sufficient record of downstream data before the impacting activity is established in the catchment. It is recommended that upstream data are collected during the period of establishing the relationship to ensure that catchment characteristics are fully understood. This approach also does not replace the need for monitoring of water quality upstream and downstream as part of an ongoing water quality compliance monitoring program.

Streams are highly variable systems where the magnitude of event-mud-loads is driven by hydrology. This analysis indicates the log-transformed difference method (Fig 2) and the mud load-discharge relationship fitted for GCDS (Fig 3) may be more robust for assessing impact than using event-mud-load simple differences between GCDS and GCUS.

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Assessment of continuous Magela Creek turbidity data upstream and downstream of Ranger

D Moliere, KG Evans & K Turner

Background

Hydrolab probes and telemetry for the continuous measurement of water quality have been installed in Magela Creek upstream (MCUS) and downstream (MCDS) of the Ranger Mine. Two water quality stations were installed at MCDS – one on the western side (Ranger Mine side of the creek) and one on the eastern side of the main channel. During the 2006–07 wet season, water samples were collected by an automatic pump sampler installed at the upstream station and one of the two downstream stations (western station), and analysed for mud concentration. Mud is defined as the silt and clay component of sediment transported in the water column (ie the portion < 63 µm and > 0.45 µm diameter). Along with in situ continuous measurement of turbidity, these data were used to derive site-specific turbidity/mud concentration relationships for both upstream and downstream Magela Creek sites. The mud concentration data, derived from continuous turbidity measured over several years, will be used to assess the degree and extent of erosion arising from the Ranger mine site, particularly that arising during and after construction of the final landform. For this, turbidity trigger values will be derived in accordance with methods recommended in the Australian and New Zealand water quality guidelines (WQG) (ANZECC & ARMCANZ 2000), and monitored using a BACIP approach (see below for explanation of the BACIP design).

Results

Turbidity-suspended sediment concentration relationships

During the 2006–07 wet season, water samples were collected on a fixed-time basis (every 12 hours) by an automatic pump sampler installed at the upstream and downstream stations. The mud concentrations in each sample were determined by sieving, filtering and oven drying techniques (Erskine et al 2001). The derived mud concentration and corresponding turbidity values measured at both the upstream and downstream sites were plotted and used to calculate a single linear regression relationship (Figure 1). The pooled-site regression is justified because there is currently no significant input of mine-derived suspended sediment in Magela Creek. Thus a Student *t*-test comparing the population means of mud concentration (of the water samples) between the two sites showed no significant difference over the 2 year period. Furthermore, a Student *t*-test also showed that there was no significant difference between the individual regression equations fitted for MCUS and MCDS. The high correlation observed between mud concentration and turbidity for the pooled regression ($R^2 = 0.84$) indicates that turbidimeters can be used to reliably monitor mud concentration within the catchment.

During late February and early March 2007, an exceptionally heavy period of rainfall occurred throughout the entire Magela Creek catchment as a result of a monsoon trough which extended across the Top End. Total rainfall over a three-day period between 1700h on 27 February and 1700h on 2 March 2007 at Jabiru airport was 785 mm, the highest three-day rainfall that has been recorded anywhere in the NT's Top End (Moliere et al 2008). Figure 2 shows that rainfall was almost continuous during these 72 hours and that within this period there were four particularly intensive rainfall episodes, the most significant of these being the period of rainfall that occurred during the morning of 1 March 2007. Maximum rainfall intensities during this period exceeded a 1:100 y storm event for durations between six and 72-hours.

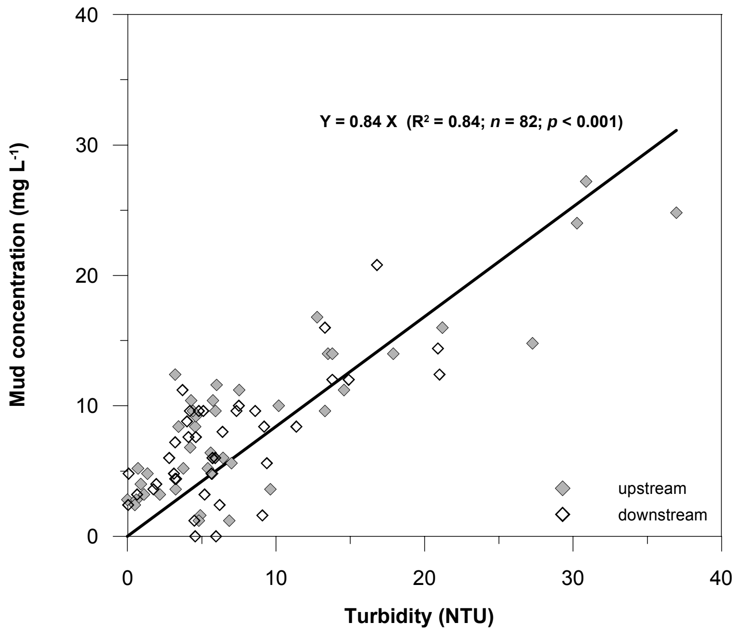

Figure 1 Relationship between turbidity and mud concentration for Magela Creek

The late February and early March 2007 flood event

This event resulted in the highest flood levels recorded within the Magela Creek catchment since recording began in 1971. Several gauging stations within the catchment were submerged by floodwaters. Total runoff in the main Magela Creek channel during a three-day period between 1200h 28 February and 1200h 3 March 2007 was greater than the mean annual runoff for the catchment. Peak flood height along the main channel exceeded the previous maximum flood height recorded on 4 February 1980 by more than 20% (Figure 2).

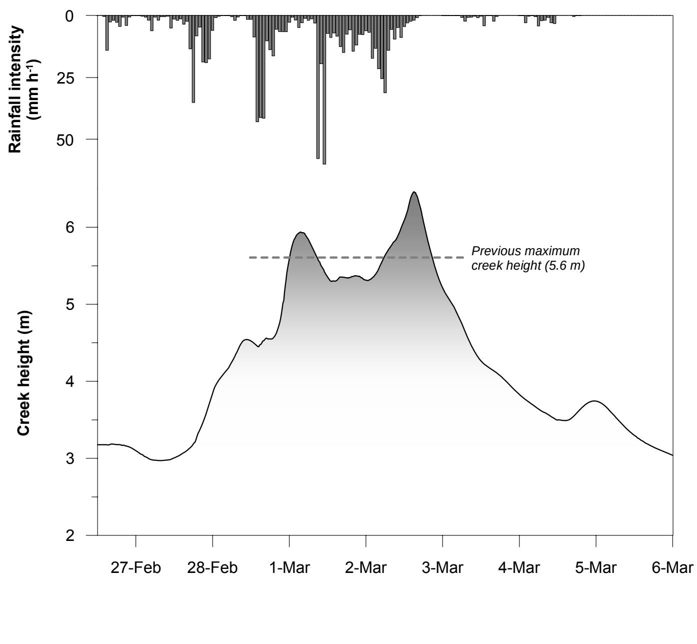

Figure 2 Hourly rainfall data collected at Jabiru airport during the flood event in relation to the flood height at G8210009, a station along Magela Creek downstream of the mine

The corresponding peak discharges along the main channel were approximately eight times the mean annual flood discharge and more than double the 1:100 y flood event discharge. The rainfall and discharge data recorded during this event will provide an extreme-event data set to use for simulating the performance of the proposed post-mining rehabilitated Ranger landform using geomorphic computer modelling.

Flood sedigraph

Fine suspended sediment moves through stream systems, such as Magela Creek, in pulses or waves generated by discharge events. In Magela Creek, peak mud concentration of a mud pulse event typically occurs at the upstream station several hours before the corresponding peak occurs downstream (Moliere et al 2008). For a particular event, peak mud concentration at the upstream station is also generally higher than that observed downstream and mud concentrations at the two downstream sites are almost identical (Moliere et al 2008). However, the sedigraph for the March 2007 flood (Figure 3; mud concentration estimated from turbidity-mud regression relationship) shows that mud movement in the catchment is very different to that typically observed during runoff events in that (1) mud concentration downstream of Ranger is much higher than that recorded upstream, and (2) mud concentration is higher in the western side of the channel than that observed in the eastern side of the channel at the downstream stations.

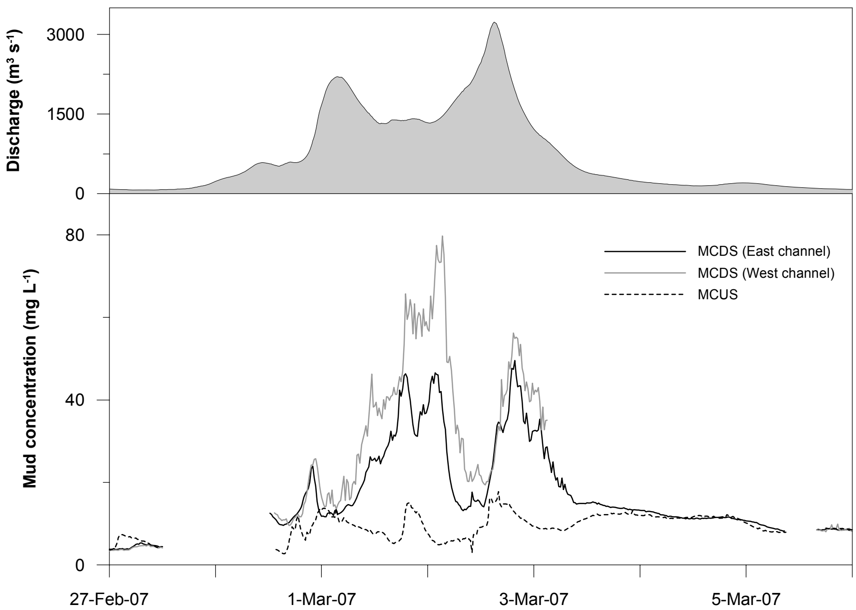

Figure 3 Mud sedigraph at the upstream and downstream stations along Magela Creek during the flood event. The corresponding hydrograph at G8210009 is also shown.

The mud load associated with this flood event (ie the area underneath the sedigraph between 1500h 28 February and 0900h 3 March) was 3400 t and 10700 t at the upstream and downstream stations respectively, corresponding to approximately 50% and 80% of the total annual mud load at the upstream and downstream stations respectively. (It should be noted that the downstream load is the average load estimated for the two downstream sites.)

Impact assessment of the flood

It is important to determine whether the discrepancy in mud loads at the two stations measured during the flood is statistically significant. In other words, is the mud load measured downstream during the flood significantly elevated compared to both the upstream load and to the corresponding event discharge characteristics? To assess this, trigger levels for event mud loads based on current conditions were derived using:

1 an event-based Before-After-Control-Impact, Paired difference design (BACIP) (Stewart-Oaten et al 1986, 1992), where MCUS and MCDS are treated as paired sites, and

2 relationships between event mud load and corresponding event runoff characteristics.

Both approaches are described in detail in kkn2.5.2 – Monitoring of sediment movement in Gulungul Creek and Moliere et al (2008). For BACIP analysis, when mud load data from the upstream (C) site are subtracted from those at the downstream (I) site, a set of paired-site (P) 'difference' values are derived. These difference values may be compared statistically for different parts of the time-series (eg before (B) and after (A) intervention) or else, as in this paper, statistical percentiles ('trigger' values) of the difference data arising in a 'reference' period are derived and to which additional data of interest may be compared. Approach 2, uses the mud-discharge relationship derived in a 'reference' period as the basis for comparing and assessing corresponding observed and predicted mud loads derived for a period of interest.

BACIP analysis

Event mud load data collected during 2005–06 and 2006–07 were used to calculate trigger values for the event-based BACIP analysis. During the two year monitoring period there were 28 events (defined in Moliere et al 2008) where complete event load data collected at the two stations (not including the March 2007 flood event) were available. Figure 4 shows the mud difference data associated with the flood event and, when plotted against the derived trigger levels, lie well above the 99.7th percentile. This indicates that the mud load measured downstream during the flood is significantly elevated compared to both that measured upstream at the same time, and when compared with that measured previously.

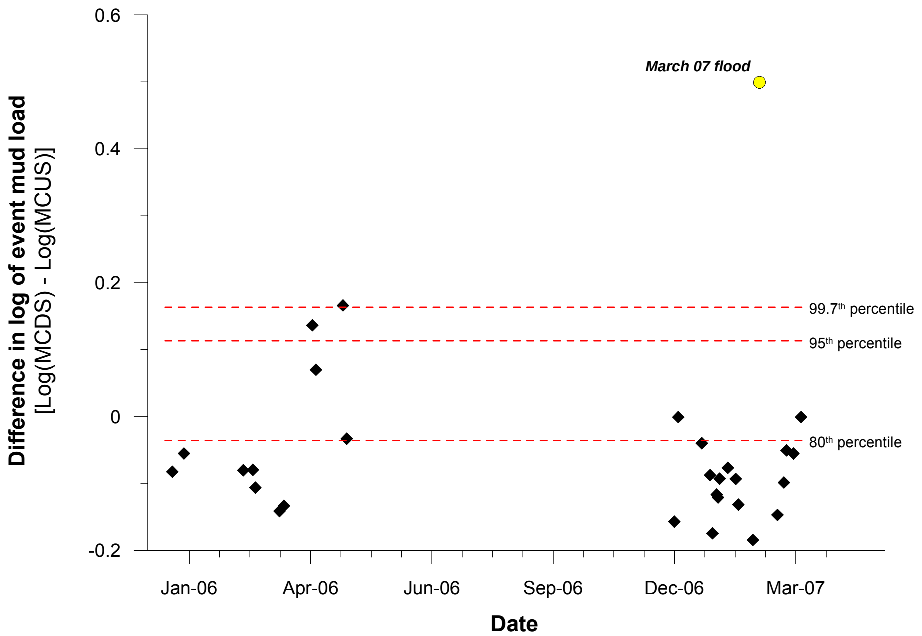

Figure 4 Temporal variation of the difference in the logarithms of the event mud loads measured at MCDS and MCUS during 2005–06 and 2006–07 (indicated as ◆). The 80th, 95th and 99.7th percentiles of the difference in the log-transformed event mud loads are shown as dashed lines.

Relationship between mud load and discharge characteristics

Event mud load data collected during 2005-06 and 2006-07 (except for that measured during the flood) were used to derive significant relationships between event mud load and corresponding event discharge characteristics (Equations 1 & 2) for each station.

$$T_{MCUS} = 0.015 Q_T^{0.96} Ri^{0.15} \quad (R^2 = 0.90;\ n = 31;\ p<0.001) \tag{1}$$

$$T_{MCDS} = 0.038 Q_T^{0.89} Ri^{0.18} \quad (R^2 = 0.91;\ n = 28;\ p<0.001) \tag{2}$$

where T is total mud load, Q_T is total discharge observed during the mud pulse and Ri is maximum periodic rise in discharge over 10-minutes.

Figure 5 shows that the mud load observed during the flood event at the downstream station, when plotted against the fitted relationship, is significantly elevated compared to the corresponding event discharge characteristics as it falls outside 2 standard deviations (+ 2 SD) of the fitted relationship (Figure 5). The mud load observed at the upstream station, however, falls within +2 SD of the fitted relationship and lies around the 1:1 line. This result indicates, unlike the mud load at the upstream site, that the mud load at the downstream site is significantly elevated compared to the discharge characteristics at this station.

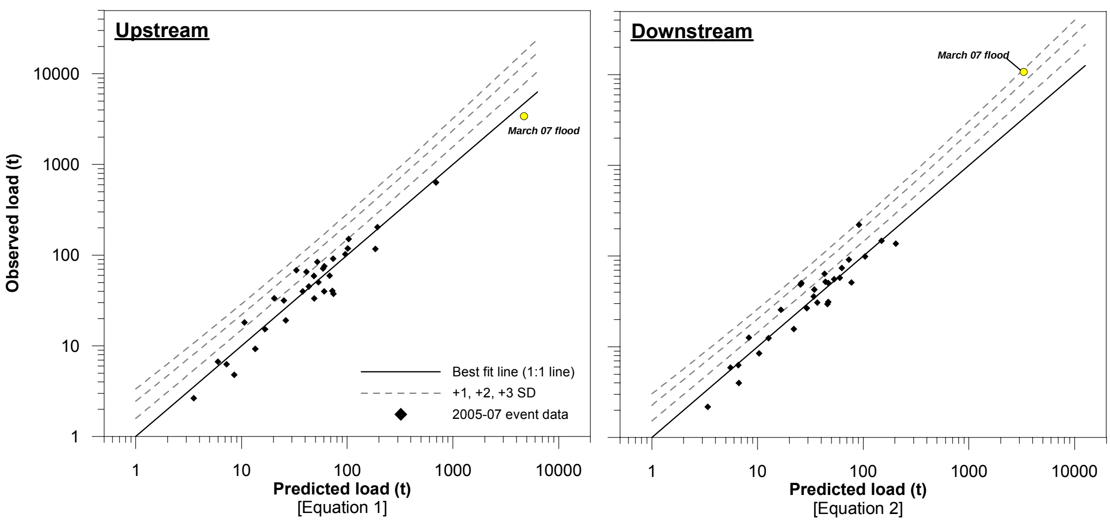

Figure 5 Event-based mud load relationships for MCUS (Left) and MCDS (Right). Event data collected between 2005 and 2007 used to fit the relationships and flood event data are also shown.

Discussion

This analysis has shown that the mud load measured during the flood event at the downstream station exceeded trigger levels associated with both the BACIP analysis (Figure 5) and the mud load-discharge relationship (Figure 6). In other words, the mud load measured downstream of Ranger Mine during this event is significantly elevated compared to that measured both upstream and historically, and to the predicted according to the corresponding discharge characteristics of the event.

According to the 2006-07 ERA Ranger Mine wet season report (Klessa 2007), the extraordinary rainfall which occurred between 27 February and 2 March 2007 resulted in the failure of several sumps and bunded structures on the mine site. As a consequence, poor quality surface water and seepage from the ore stockpiles, which are normally directed to retention ponds on site, entered Coonjimba Creek and Corridor Creek during the flood event, which then enter the main Magela

Creek channel between the upstream and downstream stations (Figure 1). It is considered likely that fine suspended sediment concentration in Coonjimba Creek and Corridor Creek would have been elevated well above typical levels as a result of the input of poor quality surface water runoff from the stockpiles entering these creeks. In addition, prior to the 2006–07 wet season, exploration drilling commenced east of Pit 3 (Figure 1). Drill pads and tracks, which were constructed from waste stockpile material, were completely submerged by floodwaters. Consequently, it is considered that this drilling area may also have been a source of fine suspended sediment into Magela Creek during the flood.

The input of poor quality water from the ore stockpiles into the minesite tributaries and the input of fine suspended sediment into Magela Creek from the exploration drilling area are likely to have contributed to the elevated mud concentrations recorded downstream of Ranger Mine. It may also explain why mud concentrations were higher in the western side of the channel (Ranger Mine side) at MCDS compared to that observed in the eastern side of the channel (because it is known that mine waste waters at MCDS are still unmixed and preferentially adhere to the western side (mine-side) of Magela Creek (Turner et al 2008)). The standard sediment controls implemented on the minesite and the drilling area were simply overwhelmed during this extraordinary rainfall-runoff event. This result has important implications for control procedures that will need to be in place on the minesite during the decommissioning and rehabilitation period, when the site will be vulnerable to high rainfall events.

The analysis of continuous turbidity-mud concentration data over the 2005–06 and 2006–07 wet seasons has also shown that one other event occurred at the downstream station (described in Moliere et al (2008)) with an elevated mud load compared to both that measured upstream and the corresponding event discharge characteristics (Figures 5 & 6). It is considered that, given the generally good correlation between the BACIP and mud load–discharge relationship approaches, both are useful techniques to assess impacts on mud concentration in Magela Creek.

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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

G Staben, M Saynor, D Moliere, KG Evans, G Hancock[1], J Lowry, G Fox & G Calvert[2]

Introduction.

During April 2006, the very destructive core of severe tropical cyclone Monica (category 5) crossed the Northern Territory coastline approximately 35 km west of Maningrida and continued in a south-westerly direction, rapidly weakening in intensity as it moved across land. The eye passed almost directly over the former Nabarlek mine, tracking close to the Jabiluka project area and the Ngarradj catchment, and continuing through to Jabiru (Figure 1) where it had weakened to a category 2 cyclone. Monica then continued to track westerly, and weakened to below cyclone intensity (Australian Bureau of Meteorology 2007).

Cyclone Monica was a small cyclone in spatial extent with extremely destructive core winds. There was a large amount of treethrow in the catchments surrounding Ranger mine, the Jabiluka project area and Nabarlek mine. Treethrow in the Gulungul Creek catchment adjacent to the Ranger mine, although less than at Nabarlek and in the Ngarradj catchment at Jabiluka, was also substantial.

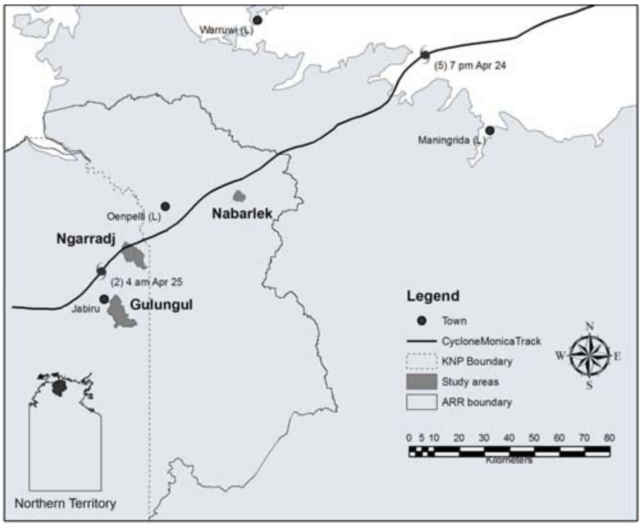

Figure 1 Estimated track of cyclone Monica across the ARR and the location of the three focus catchments

This event provided an opportunity to gather information on the increased availability of sediment for erosion and possible impacts on stream sediment transport resulting from the effect of treethrow in creek catchments.

Remotely sensed (Landsat TM5, QuickBird™) and field data are being used to assess the level of treethrow across the catchments. Although Cyclone Monica was an example of an extreme event, in the Gulungul Creek catchment (as elsewhere) it did not have the accompanying prolonged heavy rainfall and subsequent runoff that is usually associated with

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[2] Maunsells Australia Pty Ltd

a cyclone or associated rain depression. Most of the immediate damage was caused by high wind velocities knocking over trees that were growing in areas with saturated soils (viz riparian zones along creek lines).

Figure 2 Example of a rootball and crater as a result of treefall in the Gulungul Creek Catchment

When a tree has been blown over by wind, there is usually a large amount of sediment trapped within the roots of the tree rootball (Figure 2). This sediment may be removed/eroded in the future and transported into sediment sinks and drainage depressions. In order to be able to distinguish erosion events arising from mining activities from those that are a consequence of natural events, such as cyclone damage, it is important to estimate the contribution to catchment erosion arising from such natural events. To this end, information is required on the percentage of trees that have fallen within the catchment, species composition of the trees, the landform trees are growing on (Koolpinyah surface, bedrock, swamps, drainage depressions etc), and the amount of sediment that has been disturbed and its fate, eg depressions, swamps, channels. The aims of this project are;

- To estimate the impacts of Cyclone Monica on stream sediment loads in the Gulungul Creek catchment, and
- To assess utility of remotely sensed data for estimating effects of treethrow from extreme wind events on catchment response.

An investigation was undertaken to quantity these aspects of treethrow on natural lowlands, floodplains and riparian zones (including stream channels), and on trial rehabilitated areas on the Ranger mine waste rock stockpiles for comparison. The investigation incorporated remote sensing and on-ground, field measurement techniques. It is anticipated that this comparison may also provide an insight into the extent of damage that may occur on the future rehabilitated mine landform as a result of such cyclone events.

Remote sensing

Multispectral satellite data (Landsat TM5 path106/row69) were acquired that covered the Alligator Rivers Region (ARR) pre-and post-cyclone Monica. Dates captured were 15/04/05 (the only available cloud-free image pre- cyclone Monica for the month of April), 04/05/06 (first available image after the cyclone) and 21/04/07 (acquired to assess the recovery of vegetation one year after the cyclone). The acquired Landsat data were orthorectified and radiometric corrected. To enable accurate assessment of change between different image dates, it is important to ensure a common radiometric response between each image (Coppin et al. 2004). Radiometric calibration was performed on all digital counts for each Landsat multispectral band using a robust regression technique, as described in Furby & Cambell (2001).

Image transformations were performed to produce the broadband greenness index, SR (Simple Ratio Index), for all image dates. Broadband greenness indices (such as SR) measure the general quantity and vigour of green vegetation (Campbell 1996). A statistically significant relationship ($r^2 = 0.91$, $n = 123$) was found between SR and canopy cover extent measured from very high resolution QuickBird satellite data captured post cyclone Monica (23/06/06). This relationship was used to estimate the percentage canopy cover for individual pixels across each Landsat image.

Percentage canopy cover was estimated for images acquired both pre- and post-cyclone for the three focus areas in this study – Nabarlek mine and Ngarrradj and Gulungul catchments (see Figure 1). These data provided an estimate of loss of canopy cover for all three focus areas 10 days and 1 year after the cyclone. The estimated total loss of canopy cover for each of the study areas was 24% for Nabarlek, 24% for Ngarrradj and 19% for Gulungul (Table 1). The tabulated results also show that there was a marked recovery of about 17% canopy cover for both Ngarradj and Nabarlek with 6% recovery estimated for the Gulungul catchment. It appears that as the cyclone moved inland and wind velocity declined, loss of canopy was reduced.

Geographic Information Systems will be used to investigate relationships between the spatial extent of treethrow, and the physical characteristics of the region represented in, for example, corresponding digital elevation models (DEMs) and land-system information datasets. The integration of spatial (remote-sensing) and field-based information in a GIS can be used to improve understanding of the extent and degree of impact that an extreme event such as cyclone Monica may have on rehabilitated mine landforms and landscapes in the ARR.

Table 1 Estimated canopy cover (ha) pre- and post- cyclone Monica for Gulungul, Ngarradj and Nabarlek using the Vegetation Index measured from Landsat TM5 satellite data

Image date	Gulungul (6645 ha)	Ngarradj (5085 ha)	Nabarlek (1333 ha)
(pre) 15/04/05	1843 ha	1634 ha	320 ha
(post) 04/05/06	1391 ha	962 ha	187 ha
(post) 21/04/07	1496 ha	1243 ha	243 ha
Total Loss	**19%**	**24%**	**24%**
Total recovery	**6%**	**17%**	**17%**

Field site selection methods

Field data were collected from Gulungul Creek catchment, the Ranger mine site and the rehabilitated Nabarlek mine lease area. A total of fifty-five 30 x 30 m plots were sampled, 31 in Gulungul Creek Catchment, 15 at Nabarlek and nine on the Ranger mine site. 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 the 15/04/05) 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 field work due to Aboriginal 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. Plot sites on Ranger were located on experimental rehabilitation areas representing three different soil treatments.

Field methods

Within each plot at each of the 55 sites, a number of variables were measured for trees ≥ 2 m in height, including: identification to species level (fallen and standing); diameter at breast height (DBH); tree fall orientation; living or dead and the extent of damage. Also measured were the dimensions of the crater caused by treethrow and the volume of material uplifted (termed pit and mound respectively) that is potentially available for erosion. At each mound, an assessment was made as to the likelihood that the displaced soil would be washed back into the depression by rainfall or transported by overland flow to the surrounding surface and beyond. Slope and aspect of each of the plots were estimated and will be checked against the one-second DEM. Plot slope angle was usually less than 5%.

At each of the sites in the Gulungul Creek catchment and the Nabarlek area, soil samples were collected from near the middle of the plots. For this, an auger was used to excavate soil to a depth determined by either reaching bedrock or no recovery due to the water table. Samples were field-characterised by colour and texture before their return to the laboratory. Two soil samples were also collected from the rootball of one fallen tree in the plot, one from the uprooted surface material the other from the base of the pit. Surface samples only were taken from the nine sites on the Ranger mine site waste rock dump. The total number of soil samples collected was 393 (Table 2).

Table 2 Number of soil samples collected from each sample area

Site	Number
Gulungul catchment	264
Nabarlek	117
Ranger Mine site	12
Total	393

The soil samples are currently being described (colour, texture and gravel, sand and clay content) and will be assigned a Folk (1954, 1974) texture class. Detailed particle size analysis (PSA) of one soil sample collected from the top 10 cm of each plot will be undertaken to characterise the surface sediment type.

Progress

Field data have been entered into electronic format and checked to ensure accuracy. Approximately 90% of the soil samples have been described and PSA of 16 of the 31 surface samples from the Gulungul Creek catchment has been completed.

A quantitative assessment of the impact of Cyclone Monica on stream suspended sediment transport within Gulungul Creek for the 2006–07 wet season was undertaken using the continuous turbidity and flow recorded at the upstream and downstream gauging stations. To determine whether a change in sediment transport characteristics has occurred or not, fine suspended sediment (mud) load data collected during 2006-07 were compared to mud load data collected prior to the cyclone using: (1) Before-After-Control-Impact, Paired difference design (BACIP) analysis, and (2) a relationship derived between event suspended sediment load and corresponding event runoff characteristics (see Moliere et al, this volume for details). The analysis showed that there was no significant difference in event mud load data collected during 2006-07 compared to that observed prior to the cyclone. However, the event mud load measured *during* the cyclone was significantly elevated compared to the corresponding event discharge characteristics. In other words, while the event mud load which occurred during the cyclone was elevated along Gulungul Creek, Cyclone Monica did not have a long-term impact on sediment transport characteristics within Gulungul Creek catchment. This lack of enhanced longterm erosion is a significant finding given the amount of tree-throw (as described above) that occurred in the catchment, and particularly in the riparian zone where erosion effects would be most evident. This indicates that soil disturbed during the cyclone was flushed out during the cyclone discharge.

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Part 3: Jabiluka

Contents[1]

[1] List of papers grouped by Key Knowledge Need.

Monitoring sediment movement and hydrology in the Ngarradj catchment

D Moliere, M Saynor & KG Evans

Background

At the 14th ARRTC meeting (September 2004) the committee members expressed a need for continued hydrology monitoring at existing ***eriss*** gauging stations. These data would be very important should mining proceed at Jabiluka, particularly since many gauging stations in the region are no longer operational and there is thus a lack of contemporary regional streamflow information. Continuous rainfall, runoff and turbidity data from the three gauging stations within the Ngarradj catchment are collected. These data are being used to derive indicators for minesite impact in the event that the Jabiluka project proceeds.

Progress

Some water samples continue to be collected and analysed for turbidity and suspended sediment concentration during each wet season to ensure that the turbidity-mud concentration relationships developed during previous wet seasons (Moliere et al 2005, 2007a) are still valid.

The mud concentration data collected upstream and downstream of Jabiluka during 2006–07 were used to establish preliminary trigger values for (1) an event-based Before-After-Control-Impact, paired difference design (BACIP) (Stewart-Oaten et al 1986, 1992), and (2) relationships between event mud load and corresponding event runoff characteristics. Both approaches are described in detail Moliere et al (2007b).

During the 2006-07 wet season, 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, causing data loss for 6 weeks between 28 February and 11 April 2007 (the equipment at the two upstream stations was not damaged by the flood). Consequently, only five mud pulse events were recorded at the downstream station, all prior to the flood event. The observed mud loads for these five events were not elevated relative to (1) the mud loads measured at the upstream stations, and (2) the predicted loads using event discharge characteristics. In other words, treefall associated with Cyclone Monica (which occurred 25 April 2006) did not contribute to elevated mud loads within the Ngarradj catchment during the 2006–07 wet season. Given the extensive damage to vegetation – both riparian and general land surface – this was an unexpected result. A possible explanation may be that early wet season rainfall, which occurred prior to flow commencing, may have allowed sufficient regrowth of groundcover to protect the land surface from elevated sediment transport within the catchment.

The flood-damaged station downstream of Jabiluka will be elevated prior to the start of the 2007–08 wet season to avoid data loss during a similar magnitude flood in the future. Data collection will continue at this station to establish a comprehensive baseline dataset which can be used to assess impacts on stream suspended sediment loads should mining proceed at

Jabiluka. Monitoring at the two upstream stations will now cease given that a sufficient upstream record has been collected for validation purposes.

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Assessment of the significance of extreme events in the Alligator Rivers Region: Impact of Cyclone Monica on riparian vegetation (*Allosyncarpia ternata* rainforest), in-channel large wood loadings, channel erosion and treethrow in the Ngarradj Creek catchment

M Saynor & WD Erskine[1]

Background

The presence of large wood in channel is important in terms of habitat as well as helping to stabilise the channel. Many northern Australian rivers still contain essentially in tact riparian forests and natural loads of large wood, especially where they are located in various types of reserves, such as national parks. Two such study reaches are located in the Ngarradj catchment on East Tributary and upper Ngarradj Creek. The channel at both sites exhibited a 'laterally stable unconfined meandering river' which is characterised by a sand-bed, a sinuous pattern, a continuous but narrow floodplain, and a narrow but well vegetated riparian corridor (Erskine et al 2005). Field studies were completed during the dry season of 2002 to determine the large wood loads for these two reaches. The results from this work found that the large wood loadings varied between 184 and 302 m^3/ha of channel but varied by over one order of magnitude for each unit length of channel (Erskine et al 2007). Small diameters dominate in terms of large wood numbers but large diameters dominate the volume. Most large wood is relatively short in comparison to channel width and hence is readily moved. Blockage ratios of large wood are usually small (<0.06) and do not often impact on channel hydraulics.

During the 2005–2006 wet season cyclone Monica impacted greatly on the Ngarradj catchment, resulting in a large amount of tree fall through out the catchment and along the stream channel boundaries, resulting in some places in the complete removal the riparian monsoon rainforest. Figures 1 and 2 show the comparison before and after Cyclone Monica.

A KKN priority is to have an understanding of the possible impacts of extreme events within the ARR but usually in the context of extreme rainfall and runoff. Cyclone Monica, which occurred in April 2006, was an example of an extreme event. However, it did not have the accompanying rainfall and subsequent runoff that is usually associated with a cyclone and its rain depression. Most of the immediate damage was caused by high wind velocities in areas that had saturated soils towards the end of the wet season and resulted in significant treethrow across the lowlands, floodplains and channels. As the event occurred very late in the wet season and there was not the normally expected heavy rain the creek bed did not experience sufficient flow to cause the expected scour and sediment movement due to the increase in large wood loadings in the channel. It was anticipated that there would be considerable scour

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and movement of bedload during the following wet season 2006–2007 during which there was a very large rainfall event which caused a large flood event.

Figure 1 Upper Swift Creek gauging site 5 April 2006, prior to Cyclone Monica

Figure 2 Upper Swift Creek gauging site 25 October 2006, after Cyclone Monica

The existence of the large woody loadings within the creek along the two reaches prior to Cyclone Monica as well as an extensive pre Cyclone Monica data set on channel parameters such cross sections, particle size etc provided a unique opportunity to study the impacts before and after this extreme event.

A project was initiated following the cyclone to remeasure the large wood loadings, re survey channel cross sections, collect bed material samples so that before and after comparisons could be made. Erosion pins were also re-installed along each of the reaches. The project was run in collaboration with Newcastle University with Professor Wayne Erskine taking the lead role and ***eriss*** providing field and logistical support.

The overall objectives of the project were:

1 To assess the impact of the in-channel woody debris on channel erosion and aquatic habitat.

2 To assess cyclone-induced changes in riparian vegetation structure.

3 To determine the significance of treefall on the lowlands in the Ngarradj catchment.

Methods

During the 2006 dry season a total inventory of the spatial distribution, orientation, composition, arrangement and blockage ratios of large wood in the channel and on the floodplain and lowlands of East Tributary gauging station and Upper Swift Creek gauging stations was undertaken. The permanently marked cross sections were resurveyed to enable comparison with cross sectional surveys between 1998 and 2003 (Saynor et al 2004). Bed material samples were also collected from each of the cross sections to allow comparisons with those previously collected (Saynor et al 2006). Long profiles along the centre line of the creek for each of the reaches were surveyed for comparison with similar surveys completed in 2002. Erosion pins were also installed at locations along the banks as well as in rootballs of fallen trees to investigate bank erosion processes.

Progress and results

Cross sections surveys (Figure 3) show that some scour did occur, during and/or following the event, in some parts of the channel as shown by a deeper bed level than previous surveys.

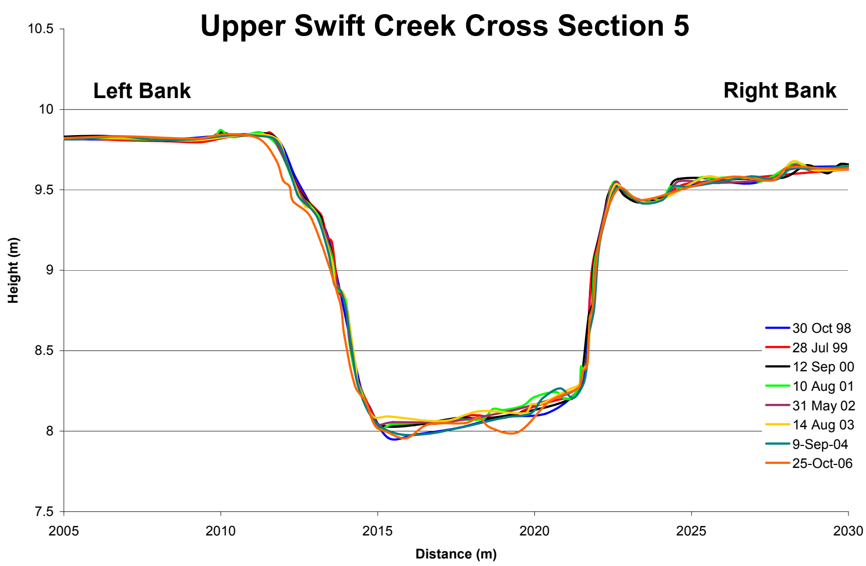

Figure 3 Cross section survey for Upper Swift Creek

Due to a serious illness, Professor Erskine, has not been able to complete analysis of the 2006 dry season. During the 2007 dry season the permanently marked cross sections will be resurveyed, erosion pins measured and bed material samples collected for future analysis. Analysis of the results will be undertaken if/when Professor Erskine recovers sufficiently or when staff resources become available.

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Part 4: Nabarlek

Contents[1]

[1] List of papers grouped by Key Knowledge Need.

Develop revegetation management options for Nabarlek encompassing new agreed closure criteria

P Bayliss, M Bush, S Bellairs,[1] G Fox & J Boyden

Introduction

The principal ongoing rehabilitation issue at Nabarlek is that it has not reached a status where the NT Government will agree to issue a Revegetation Certificate to the mine operator. In 2003 ARRTC recommended that a quantitative revegetation assessment be undertaken and that, following its completion, management options be proposed to the Nabarlek Minesite Technical Committee (MTC) for consideration. Additionally, ARRTC recommended that Nabarlek be used as an analogue for the rehabilitation of Ranger. The aim of this project is to assist the Nabarlek MTC stakeholders develop realistic and practical management options for revegetation that incorporate new agreed closure criteria. A necessary prerequisite is the definition of new agreed closure criteria in consultation with the Traditional Land Owners.

Progress to date

Assessment work undertaken by ***eriss*** in the late dry season of 2003 and the late wet season of 2004, in collaboration with Charles Darwin University, showed clearly that revegetation is largely unsuccessful in terms of the original goal of 'blending in with the surrounding savannah woodland'. Vegetation on the minesite is characterised by: extensive cover of grassy weeds and, hence, extreme fire risk; acacia shrubland nearing the end of its successional life; and a very low density and recruitment of trees and shrubs. During this period two postgraduate studies at Charles Darwin University, supervised by Dr Sean Bellairs and ERISS staff, were completed (Stefani Vink: The Revegetation of the Nabarlek Uranium Mine 2004; Judy Manning 2004: Sustainability of Revegetation at Nabarlek Minesite – An Assessment of Wet Season Nutrient Cycling and the Soil Seed Bank). Additional ground surveys were planned five years later (2009–2010) to assess new revegetation works by the mine operator. However, since 2004 the Nabarlek minesite and surrounds have suffered three severe fires (November 2004 & 2005, May 2006) and a direct hit by Cyclone Monica (April 2006). Hence, given the severity and frequency of concurrent ecological disturbances, a decision was made to re-sample ground transects in the late dry season of 2006 and the late wet season of 2007 to assess their combined impacts. Data are currently being analysed and will be documented as an Internal Report. Additionally, in collaboration with Dr Bellairs, a publication timetable for the Nabarlek vegetation assessment work has been drafted. During the late dry season survey in 2006 a separate and more intensive tree-fall study was undertaken to independently assess the impact of Cyclone Monica, and to establish a sound baseline to monitor post-cyclone recovery rates of vegetation on and off the minesite.

In 2005 ***eriss*** reviewed sections of the NLC's draft 'Secondary Standards for Nabarlek Minesite Close-out' pertaining to revegetation, weed and fire management. In November

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2006 OSS and ***eriss*** reviewed the mine operators draft Closure Plan for Nabarlek minesite, which advocated a major shift in the original closure goal of a successional woodland to a mixed landuse model comprising commercial grassland and sedge seed ‘orchards’. The SSD recommended that whilst seed orchards are options worth investigating, more supporting information and analysis is required, in particular a detailed business risk analysis, before implementation. The mine operator is currently re-considering their closure options. ***eriss*** will continue to assist the Nabarlek MTC in developing new closure criteria if requested, and will also continue to provide technical advice in the development of revegetation management options, particularly with respect to methods to monitor and assess revegetation success and risks from weeds and fire.

Radiological impact assessment of the rehabilitated Nabarlek site

A Bollhöfer & B Ryan

Introduction

The aims of this project are to complete the radiological assessment of the rehabilitated Nabarlek site, assess the success of rehabilitation at Nabarlek and apply the lessons learned to the rehabilitation of Ranger uranium mine. In order to determine radiation doses to people accessing the rehabilitated Nabarlek site, a radiological risk assessment, including the main exposure pathways, is needed. This risk assessment needs to consider:

1 terrestrial gamma doses on site;

2 inhalation doses from radon progeny and dust;

3 radionuclide concentration in surface soils and erosion/stability assessment; and

4 measurement of radionuclide uptake into edible plants growing on and off site.

Fieldwork, data analysis and interpretation on Tasks 1, 2 and 3 are complete. The collection of bush food samples is ongoing and results will be incorporated into a complete dose model for the site. However, data collation for task 4 has been held up due to cyclone Monica in 2006, the flood in 2007 and frequent fires during the last two years.

Results

Monthly doses were calculated for access of the site during the dry season. For the dose assessment it was assumed that areas on site were used for camping, although areas off-site along Cooper Creek are more suitable and likely due to their proximity to drinking water. The ingestion pathway was not considered in this assessment.

Local people accessing the site for hunting or maintenance of vegetation plots will access the site during daytime hours only, which will considerably lessen the doses received via the inhalation of radon progeny, as radon concentrations vary with the time of day at Nabarlek (Martin et al 2004). Consequently, the inhalation dose due to radon progeny has been calculated as doses received during the day time and doses received during the night, while camping on site.

Terrestrial gamma doses

An airborne gamma survey and subsequent groundtruthing of the airborne data has made it possible to estimate the area wide terrestrial gamma dose rates on the rehabilitated mine site, and at environmental background sites in the greater Nabarlek area (Martin et al 2006). The terrestrial gamma dose rates have been translated into effective doses received per month, using a conversion factor of 0.79 $Sv\cdot Gy^{-1}$ (UNSCEAR 2000) and are reported in Table 1.

Inhalation doses

The diurnal and annual variation of radon concentration at Nabarlek has been determined using permanent radon monitors deployed on site from 1997 to 1999. These variations have ben published in an earlier report and journal paper (Bollhöfer et al 2004, Martin et al 2004). In addition, 78 passive radon monitors (PRM) were distributed across the site in 2005, to determine the geographical variability of airborne radon concentration, and the results are published in Bollhöfer (2007).

Table 1 Monthly doses received during the dry season via terrestrial γ radiation and the inhalation of radon progeny and dust (data modified from Martin et al 2006, Hancock et al 2006, Bollhöfer 2007)

	γ dose rate [mSv per m]	RDP inhalation [mSv per m]			Dust inhalation [mSv per m]
		day time	night time	24hrs	
EP-2	0.21	0.5	1.1	1.6	0.02
Unit 7	0.57	0.4	0.9	1.3	0.42
PROP	0.20	0.1	0.2	0.3	0.02
SPROP	0.20	0.3	0.8	1.1	0.02
EP-1	0.27	0.2	0.5	0.7	0.03
PIT	0.29	0.2	0.4	0.6	0.04
WRD	0.26	0.1	0.4	0.5	0.03
WRDROP	0.27	0.2	0.4	0.6	0.03
OSP	0.14	N/A	N/A	N/A	0.01
Plant & offices	0.10	0.1	0.1	0.2	0.00
Aistrip LAA	0.05	0.1	0.1	0.2	0.00
Total area (weighted)	0.18	0.15	0.35	0.5	0.02
ENV	0.05	0.1	0.1	0.2	0.00

Radiation doses to people from the inhalation of radon progeny have been determined from the radon concentration in the air via the following equation:

$$E_{RDP} = h_{RDP} \cdot C_{RDP} \cdot t \qquad \text{Eqn 1}$$

With:

E_{RDP}: effective dose due to the inhalation of radon decay products [μSv]

h_{RDP}: dose conversion factor [$\mu Sv \cdot (\mu Jh \cdot m^{-3})^{-1}$]

C_{RDP}: radon progeny potential alpha energy concentration, PAEC [$\mu J \cdot m^{-3}$]

t: inhalation time.

The radon progeny PAEC has been determined from the radon concentration measured, using an appropriate equilibrium factor, F. Akber and Pfitzner (1994) have determined an equilibrium factor of 0.45–0.60 for the dry season in Jabiru. The daytime dose conversion factor was assumed to be 27 nSv $(Bq \cdot h \cdot m^{-3})^{-1}$ equilibrium equivalent radon concentration, assuming an unattached radon fraction f_p of 0.12 for Nabarlek and inhalation by a nose breather (1.2 $m^3 \cdot hr^{-1}$) undertaking light to moderate activities (such as hunting or planting) on site (for details see Bollhöfer 2007). For the night time, a dose conversion factor of 9 nSv

$(Bq \cdot h \cdot m^{-3})^{-1}$ equilibrium equivalent radon concentration was assumed as suggested by UNSCEAR (2000). Table 1 shows the monthly doses received via the inhalation of radon progeny for access of the site during the day and night (12 hrs) and access for 24 hrs, respectively.

Radiation dose received from the inhalation of radioactivity trapped in or on dust is conventionally determined via the collection of dust on high or low volume air filters, and subsequent measurement of the gross alpha activity collected on the filters. This has not been performed at the Nabarlek mine site. However, the airborne activity concentration can also be estimated using an equivalent soil concentration S_E, defined as the ratio of the concentration of a radionuclide in air C_a $(Bq \cdot m^{-3})$ to that in the soil C_s $(Bq \cdot kg^{-1})$ (Nicholson 1988).

Akber (1992) has determined S_E for the Alligator Rivers Region to ~$7.2 \cdot 10^{-7}$ $(Bq \cdot m^{-3}$ air)/$(Bq \cdot kg^{-1}$ soil), which has been used in this assessment (this value can vary by orders of magnitude however shouldthere be disturbance of the soil by activities such as sweeping or traffic by humans or vehicles that stirs up dust and soil particles).

The activity concentration of a radionuclide in air, C_a, has then be calculated using:

$$C_a\ [Bq \cdot m^{-3}] = C_s \times S_E$$

With:

C_s: average soil activity concentration

S_E: equivalent soil concentration, $7.2 \cdot 10^{-7}$ $(Bq \cdot m^{-3})/(Bq \cdot kg^{-1})$.

Using this equation, dose conversion factors from ICRP72 (1996) and assuming equilibrium in the uranium and actinium series and a natural activity ratio of $^{235}U/^{238}U$, and a breathing rate of 22.2 $m^3 \cdot d^{-1}$ [UNSCEAR 2000] doses from the inhalation of dust at Nabarlek have been calculated for various areas on site and the site as a whole. In general (apart from unit-7) these doses are low compared to the radon and external gamma pathways (see Table 1).

Radionuclide erosion off site

Two separate assessments, using an erosion modeling approach (Hancock et al 2006) and stable lead isotopes (Frostick et al 2007), respectively have shown that erosion and deposition of sediments containing elevated levels of trace meals and radionuclides from the rehabilitated site occurs. However, the impact on downstream sediments is minor as contaminant loadings are relatively low and are masked by the metal and radionuclide concentrations found naturally.

Using the stable $^{206}Pb/^{207}Pb$ ratio as a source tracer in the sediments it was found that approximately six per cent of the lead deposited in a sediment core from Cooper Creek, immediately downstream of Nabarlek, originates from a uranium rich on-site source with an endmember $^{206}Pb/^{207}Pb$ ratio of 3.13. This contribution however is likely to have existed pre mining, due to erosion of the outcropping orebody, as indicated by the presence of radiogenic material throughout deeper, pre-mining sections of the sediment cores downstream of Nabarlek.

Groundwater

Radionuclide activity and dissolved metal concentrations in borewaters from the decommissioned and rehabilitated Nabarlek uranium mine have been published (Ryan & Bollhöfer 2007) and have been reported at the last ARRTC meeting. ^{226}Ra is the most important radionuclide for the assessment of ingestion doses due to its high dose conversion factor and its relative mobility in groundwater. Although the remnants of spray irrigation can still be seen in the bores situated at the northern end of the mine and groundwater flow directions are generally eastwards towards Cooper Creek, in agreeance with the regional groundwater flow patterns calculated by Salama (1986), ^{226}Ra concentrations measured in bores in the vicinity of Cooper Creek and Cooper Creek West are generally low (< 6 mBq·l) and unlikely to significantly affect water quality of Cooper Creek.

Summary

Table 2 summarises the doses received for 1 month and 3 month access of the site, respectively, excluding the ingestion pathway. It illustrates that for full time occupancy of the fenced site of 1 month, a total effective dose of approximately 0.5 mSv above typical environmental background doses would be received. Pre-mining doses on-site, specifically from the inhalation of radon, would have been much greater but are unknown and thus a comparison with post mining conditions cannot be made.

Table 2 Total doses received on-site and at environmental background areas

Site	Monthly dose [mSv/m]	Assumed occupancy	Dose at mine area	Background dose	Total annual dose
Camping on site	0.70	1 month	0.70	2.75	3.45
		3 months	*2.10*	*2.25*	*4.35*
Activities during the day (12 hrs)	0.24	1 month	0.24	2.88	3.12
		3 months	*0.72*	*2.63*	*3.35*
Background	0.25	12 months		3.0	3.0

Background doses are calculated by multiplying time spent off mine site by the background dose rates e.g. camping for 3 months on minesite results in background dose = 0.25 mSv/m x 9 months offsite = 2.25 mSv

A much more realistic scenario of access to the site during daytime hours only, for one month during the dry season, will result in an effective dose of 0.12 mSv above background. An access period of 3 months would result in 0.35 mSv above background doses.

Steps for completion

Nabarlek groundwater collection is ongoing, including metal and trace metal concentrations, and aliquots from this year's collection have been received and are being analysed. Future work at Nabarlek will focus on bushtucker collection and analysis, and an estimate of plant radionuclide uptake factors for the site. Furthermore, an estimate of Cooper Creek upstream and downstream radionuclide activity concentrations is needed, in particular ^{226}Ra, to determine the mine related contribution, if any, of radionuclides in the water column downstream of Nabarlek. This will then complement the site dose model to be developed for Nabarlek.

Acknowledgments

All bore samples collected between 1996 and 2005 and associated field data were supplied by the Minerals and Energy Group of the Department of Primary Industry and Fishery and Mines.

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Part 5: General Alligator Rivers Region

Contents[1]

[1] List of papers grouped by Key Knowledge Need.

Significant habitats and species in the Alligator Rivers Region

C Humphrey

Background

This project is being conducted in response to a specific recommendation of the IUCN and the World Heritage Committee. Their concern was that there might be endemic species of animals that, while not at risk from direct contaminant pathways associated with any development at Jabiluka, could be affected by indirect secondary pathways.

Surveys have been conducted on the aquatic fauna of seeps and springs in the stone country of KNP including in the vicinity of Jabiluka. Particular attention is being paid to the crustacean groups, the isopods (family Amphisopodidae), freshwater crabs (family Parathelphusidae) and prawns and shrimps (families Atyidae and Kakaducarididae) that occur in these habitats. The material collected in these surveys is being provided to taxonomists and molecular biologists to distinguish and describe new species collected from the sites. If species are identified that are only found in the vicinity of Jabiluka, consideration will be given to pathways for potential impact and possible monitoring programs.

This work could also be of direct relevance to any possible mining developments in Western Arnhem Land. In addition, these endemic species may be vulnerable to the invasion of cane toads and their life stages, through either toxicity (direct contact or from consumption) or competition with tadpoles for food and space.

Progress to date

Current work is taxonomic, with southern collaborators from Griffith University, BioAccess Australia and The Australian Museum conducting morphological and molecular genetic studies on the samples. Work to date is confirming that the macro-crustacean groups from seeps and springs are very diverse, with very localised endemism.

One of the crustacean groups is an endemic genus of phreatoicidean isopod (*Eophreatoicus*) that has exceptional species-level diversity. Dr George Wilson from the Australian Museum and ***eriss*** received a research grant from the Australian Biological Resources Study (ABRS) in 2004 to describe isopods of KNP and western Arnhem Land. Dr Wilson has made particular note about the results acquired from the Jabiluka region – collected from Ngarradj, Catfish, North Magela and Wirnmuyurr Creeks, and three small tributaries draining the Jabiluka outlier itself. It appears that there is a ‘species flock’ around the Jabiluka region with a number of these small creeks returning 2 or 3 co-occurring species, restricted just to these streams.

It has been postulated that this diversity and endemism is a consequence of the extreme age and persistence of the stone country and its associated fresh waters, and isolating mechanisms including fragmentation of habitat (long-term climate changes, erosion) and the generally poor dispersal characteristics of the crustacean groups. One model being proposed is that climatic variation over the Neogene (broad period of geologic time starting 23 million years ago to the present) has promoted speciation during both periods of aridity (when populations become isolated) and wetter periods (when animals are able to disperse). This ‘species pump’ model has also been used to explain diversity of vertebrates in the wet tropics of Australia.

The isopod studies are nearing completion after some additional samples were collected in 2006–07 from Jabiluka sites from which there was insufficient material for analysis. The results of the studies are currently being prepared for publication. The results have profound implications for the conservation values of the sites and resident faunas. For any future work associated with mining activity at Jabiluka, groundwaters associated with surficial and deep aquifers, and connectivity (if any), would need to be well understood in order to properly assess potential impacts upon these (dry season) groundwater-dwelling animals.

In relation to freshwater shrimps, the taxonomy and systematic relationships of the endemic family, Kakaducarididae, have been examined at multiple levels using morphology and mitochondrial and nuclear DNA sequences. The studies are nearing completion and the results are currently being prepared for publication (eg Page et al in press). Amongst the two kakaducaridid genera, *Leptopalaemon* and *Kakaducaris*, the analyses have confirmed the presence of one species of *Kakaducaris* and four species of *Leptopalaemon*, three of the latter species being new and currently undescribed. The molecular genetics datasets place the Kakaducarididae within the Palaemonidae, where its shows a close relationship to the diverse genus Macrobrachium. The two kakaducaridid genera, *Leptopalaemon* and *Kakaducaris*, form a strong clade but are well differentiated from each other. Within the Leptopalaemon, each sample site is distinct, suggesting isolation and a lack of contemporary gene flow.

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Landscape-scale ecological risk assessment of Magela floodplain to differentiate the actual impacts of mining from broader non-mining stressors

P Bayliss, J Boyden, D Walden, R Bartolo, R van Dam & C Humphrey

Introduction

The aims of the Landscape Program were to help differentiate mining and non-mining impacts, and to contribute to the broader assessment of the World Heritage (WH) values of Kakadu National Park (KNP). The Ecological Risk Assessment project is the final project of the 'Landscape-scale analysis of impacts' Program established in 2002 on the recommendations of the Independent Science Panel (ISP) 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. The analysis framework, underlying conceptual models, quantitative approaches and preliminary results have been outlined in detail in previous ARRTC reports and Research Summaries (see Bayliss et 2006, Bayliss et al 2007). The approach basically combines an Ecological Risk Assessment of key diffuse non-mining landscape-wide threats to susceptible World Heritage values on Magela floodplain with a quantitative Ecological Risk Assessment of point source threats from four key chemical contaminants (uranium, manganese, magnesium & sulfate) released from Ranger into the surface water pathway (KKN 1.2.1).

The original ISP Landscape Program has basically finished, apart from a series of publications and final reports. A journal manuscript is currently being drafted on the combined Ecological Risk Assessment of Magela floodplain that will include the conceptual contaminants pathway model developed in KKN 1.2.1 (due Dec 2007). Two SSRs have been completed and due for publication in 2008 (SSR192: A GIS compendium for landscape-scale ecological risk assessment on the Magela floodplain and the broader ARR; SSR194: An ecological risk assessment of major wetland weeds on the Magela Creek floodplain, Kakadu National Park). Four other journal manuscripts are currently being drafted (see below). Other outputs include five conference presentations (International Environmetrics conference, China 2006; Society Risk Assessors conference, Melbourne 2006; NT Water Summit, Darwin 2006; Kakadu Landscape Change Symposia, Kakadu National Park 2007; AusIMM conference, Darwin 2007), with two in preparation (Asian Remote Sensing conference, November 2007, Malaysia; Greenhouse2007, October 2007).

A new direction for landscape-scale ecological risk assessment research is currently being charted within ***eriss*** and with stakeholders (including ARRTC members), and will build on past efforts and experiences on the Magela floodplain and recent work in the Daly River catchment undertaken for the external Tropical Rivers Inventory and Assessment Program (TRIAP). Additionally, the new risk assessment work at the landscape-scale will incorporate outputs from a site-specific Ecological Risk Assessment of the transition and rehabilitation phases of Ranger (see below).

Progress to date and future directions

Magela

Two key results from the initial ecological risk assessment of the Magela floodplain reported at ARRTC17 and documented in Bayliss et al (2006, 2007) were: (i) diffuse landscape-scale risks were several orders of magnitude greater than point source risks to Magela surface waters from Ranger uranium mine; and that (ii) of the landscape-scale risks, damage from para grass > feral pigs > unmanaged fire. Hence, para grass weed is currently the greatest threat to Magela floodplain natural and cultural values. At this time 35% of the area of the floodplain is affected by para grass and of this impacted footprint area approximately one-third is effectively a para grass monoculture, with the native floodplain vegetation having been extinguished. The area of para grass coverage is doubling in extent on average every 5 years. The spatially-explicit quantitative risk assessment of para grass on Magela is underpinned by extensive modelling work to predict spread rates into native wetland habitats, and very-high resolution (VHR) remote sensing data captured by the Quickbird™ satellite used to map the distribution of wetland vegetation. In collaboration with PAN-Kakadu and Charles Darwin University, ***eriss*** currently supports an extension to its para grass risk assessment work through a postgraduate study, outlined below.

Extension to para grass risk assessment work via a postgraduate study

A key result of previous ***eriss*** risk assessments of para grass on Magela floodplain has been the identification of critical knowledge gaps for effective management. There is incomplete information on the distribution and abundance of para grass across the entire floodplain, and the different rates of invasion and, hence, impacts in specific habitats. By addressing these information gaps and providing the capability to detect small satellite outbreaks, para grass weed may be more strategically and cost-effectively managed in extensive and remote wetlands. Very high spatial resolution remote sensing is providing encouraging results in this regard. The first trial QuickBird™ capture of the central portion of the Magela floodplain in the late wet season of 2004 delivered an overall map classification accuracy of 86%, when measured against training field data. For para grass, accuracy ranged from 90 to 97% across its three visibly distinct forms (Table 1).

Table 1 Percentage accuracy produced for each map class derived from the 2004 QuickBird™ capture measured from ground-validation data

Map class	[1]Accuracy
Para grass 1 (high greenness form)	90
Para grass 2 (moderate greenness form)	91
Para grass 3 (low greenness form)	97
Hymenachne acutigluma	38
Eleocharis spp	99
Mixed lilies & open water	72
Open water	70

1 The percentage of correct class predictions based on assessment against a subset of training site ground data

Results indicate that VHR remote sensing can therefore provide more comprehensive and cost-effective information on the distribution and abundance of para grass and native vegetation, and the rate of para grass colonisation into key habitats, over the entire floodplain.

In the late wet season of 2006 the entire Magela floodplain was captured by Quickbird^{TM} (Figure 1) and, at the same time, an intensive field validation campaign was undertaken to systematically sample para grass and associated native floodplain vegetation types, and to measure the range of spectral classes of para grass.

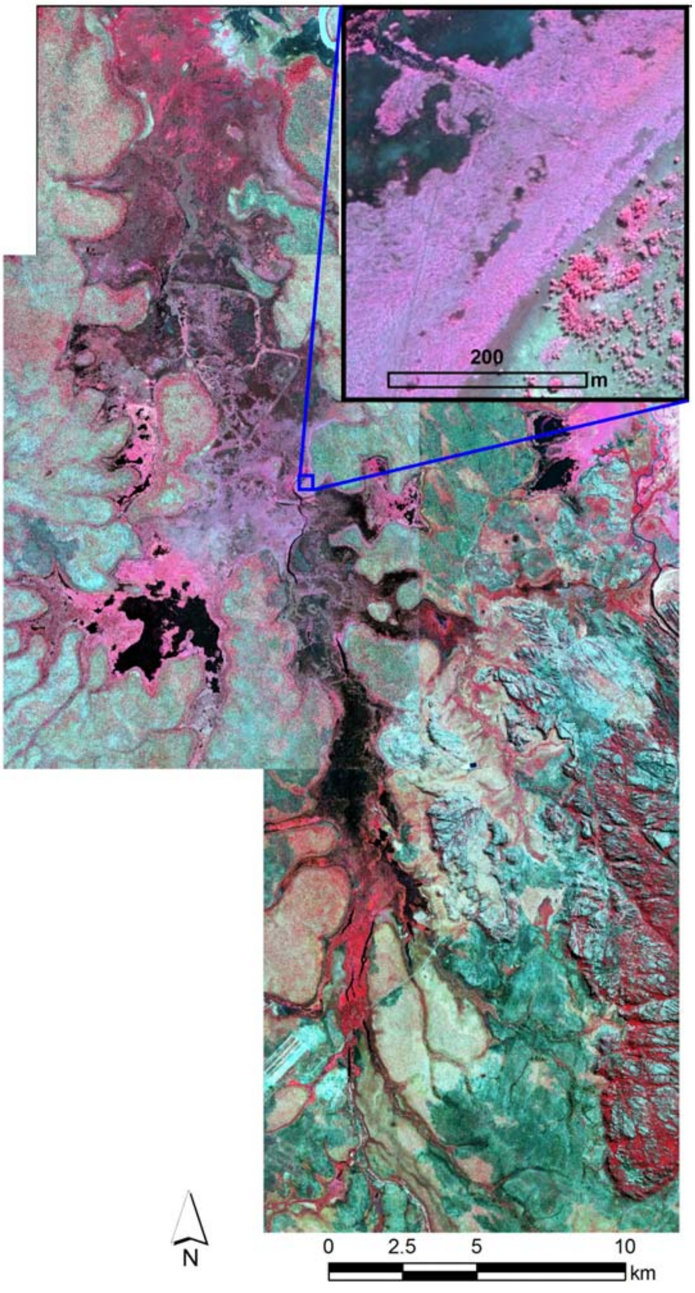

Figure 1 False colour composite VHR QuickBird™ image of the Magela Creek floodplain (April/May 2006). The high-resolution box shows a lighter band of para grass along the margins of the floodplain (appearing purple in colour reproduction).

The postgraduate study will combine all existing temporal and spatial data on para grass across the Magela floodplain to undertake a more comprehensive spatially-explicit risk analysis, modelling and assessment. For example, native wetland habitats at greatest risk to para grass invasion will be determined using more advanced habitat suitability models developed in earlier ***eriss*** studies. Two journal manuscripts are currently being drafted: 'Evaluation of VHR remote sensing for monitoring the environmental weed, para grass, on tropical floodplains of Kakadu National Park'; and 'A spatially-explicit ecological risk assessment of para grass weed on Magela floodplain, Kakadu National Park: invasion rates and implications for cost of control.'

Boggy Plain

Five years of vegetation surveys have been undertaken at Boggy Plain (South Alligator River catchment) in the wet and dry seasons since the Traditional wetlands burning project commenced in 2002. Analysis of data examining the influence of floodplain burning and hydrology on wetland vegetation dynamics is nearly complete, and a manuscript is currently being drafted for publication in a book on Fire Ecology and Management in northern Australia (Chp 6. Burning wetlands for resource protection), to be published by the Tropical Savannas Management Cooperative Research Centre.

Communicating results from the Landscape Impacts Program

Results of the Landscape Program, in particular the Ecological Risk Assessment of Magela floodplain, will be communicated to all ARR stakeholders and Traditional Owners through a series of landscape symposia organised and facilitated jointly with PAN-Kakadu. The symposium 'Landscape Change' is the first in the series and was held in March 2007, providing an overview of agents of landscape change and setting the scene for all other symposia and workshops listed below.

- Landscape Change Symposium March 6 & 7 2007 – completed (see summary below)
- Weeds Workshop 27 & 28 November 2007
- Feral Animal Control Workshop February 2008
- Fire Management Workshop April 2008
- Climate Change Workshop July 2008
- Ecological Risk Assessment Symposium November 2008 – risk assessment and adaptive management. This will essentially be the summary and wind up forum for all preceding symposiums and workshops.

Kakadu National Park Landscape Change Symposium – March 2007

Parks Australia North hosted the Kakadu National Park (KNP) Landscape Change Symposium at the South Alligator River Inn in March this year. This is the first in a series of symposia and workshops focused on agents of change in Kakadu National Park (KNP) that will have relevance to similar environments in the ARR and elsewhere. The three main aims of the initial symposium were to: (i) provide an overview of the different agents of change, setting the scene for specific fora to follow later; (ii) enable the effective two-way transfer of knowledge between KNP staff, researchers, the Kakadu Research Advisory Committee (KRAC) members, stakeholders, and Traditional Owners; and (iii) place research and scientific knowledge into a management context so that questions to Park Managers and Traditional Owners can be posed regarding future management frameworks and research directions. ***eriss*** staff were on the Landscape Symposia Planning Committee, delivered presentations at the overview symposium and help facilitate workshop sessions on a range of topics including risk assessment and management (P Bayliss – 'Using a risk assessment approach to manage landscape change'), climate change (R Bartolo – 'The status of climate change research in the Kakadu landscape context'), invasive species and the rehabilitation of Ranger (ie incorporating the RPA back into the Kakadu landscape, D Jones – 'Uranium mining in the Kakadu-landscape: issues for operations and closure'). Climate change issues have not been explicitly addressed in previous ARRTC progress reports although it is listed as potentially a key regional threat. Hence, the workshop discussion on the status of current climate change knowledge in relation to KNP's management plan objectives is summarised below.

- KNP will be subject to the impacts of climate change including sea level rise, increase in temperature, increases in extreme events such as hot spells and storm surges, possible increase in tropical cyclone intensity and changes to localised rainfall patterns;
- The coastal environment of KNP is dynamic and habitat change has occurred in the past due to fluctuations in sea level. The freshwater floodplains are young in age; and
- Impacts of climate change will affect: Bininj use of natural and cultural resources; fire regimes; flood inundation patterns in freshwater systems; location of biodiversity; and availability of freshwater to both the natural environment and people.

The main threats to landscape health in KNP will include: saltwater inundation of freshwater coastal environments due to sea level rise and storm surge events (Fig 2); the response of mangrove communities to rising sea level; and more intensive fire regimes may eventuate due to hotter dry seasons (extreme event hot spells) and these hot fires may result in a decline of monsoon forest.

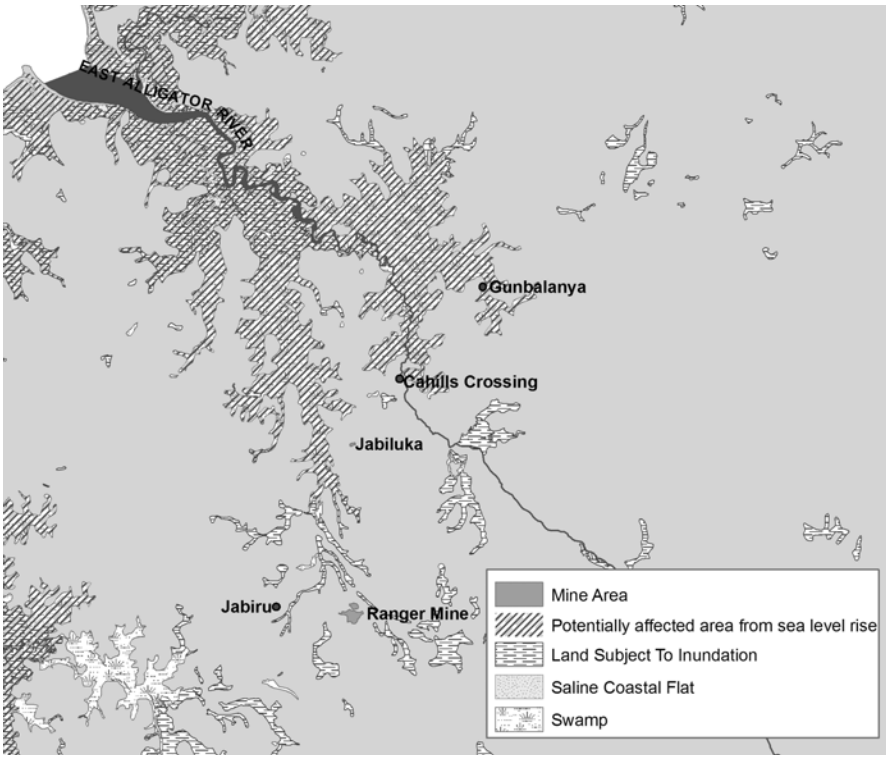

Figure 2 Wetland areas of the East Alligator River, including the RPA, which may be potentially affected by sea level rise

Future directions for ecological risk assessment

The current huge difference between non-mining and mining-related risks highlighted by the Magela ecological risk assessment may reduce when on-site water management systems at Ranger change in the transition between mine production and mine closure, and during rehabilitation. The transition between operations and closure, and the rehabilitation phase, requires a detailed and explicit risk assessment in itself that encompasses key stressors, pathways and agreed assessment endpoints. Hence, the following two new KKNs have been proposed to ARRTC in the current round of revisions in order to capture future risk assessment needs in a strategic, transparent, coordinated and efficient manner:

1 KKN 2.6.1 Site-specific Ecological Risk Assessment of the rehabilitation phase, especially in the transition period between cessation of operations and the landform construction of the landform, and the stabilisation phase of the landform following initial rehabilitation.

2 KKN 5.1.1 Develop a landscape-scale Ecological Risk Assessment framework for the ARR (specifically the catchment of Magela Creek) that incorporates, and places into context, current uranium mining activities and the rehabilitation of Ranger into the broader landscapes of Kakadu National Park.

The Magela catchment is one of the most studied tropical catchments in Australia and, as a consequence, much biophysical knowledge and good quality time series data are available for more advanced ecosystem and risk modelling. However, whilst much biophysical knowledge has accumulated over 30 years it still lacks overall coherency and, hence, power to generalise across a range of management contexts. This is because much of the knowledge base is still site specific and discipline based, and has yet to be drawn out and integrated into coherent and testable ecosystem modelling frameworks. This in turn constrains development of knowledge-based Ecological Risk Assessment and adaptive management frameworks.

Spatially-explicit ecosystem process models are currently being developed to facilitate Ecological Risk Assessments over different time frames and spatial scales (see ARRTC18). Such models will be driven by rainfall-discharge submodels of the Magela catchment (see KKN 2.1.5 'Assessment of the significance of extreme events in the ARR'), with direct links to the population dynamics of indicator plant and animal species. Several plant-animal population models have already been developed using historical data. Synergies between the developing TRaCK (Tropical Rivers & Coastal Knowledge) program and ***eriss*** landscape studies in the ARR would be mutually beneficial. Hence, ***eriss*** plans also to collaborate with TRaCK consortium partners to value-add to current ecosystem and risk modelling work through studies on material flows and budgets, and to develop remote sensing methodologies to map flood-inundation using a combination of radar and optical platforms.

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Assessment of the radiological status of the Sleisbeck mine and the Rockhole residue site

A Bollhöfer, K Pfitzner, B Ryan, P Martin[1], M Fawcett[2], L Dunn[3] & DR Jones

Introduction

The Commonwealth Government announced provision of funds in its 2006 budget for the rehabilitation of the South Alligator River Valley (SARV) mine and mill areas over a four-year period. The planned first phase will involve rehabilitating the Gimbat area and the old Sleisbeck mine. In subsequent years other the remaining sites will be rehabilitated. Probably the most extensive works will involve rehabilitating the Rockhole mine residues and earlier containment sites. The material currently located at these sites will be recovered and transferred to a purpose built containment. A decision is yet to be made on any required rehabilitation works at Rockhole mine creek, pending the outcomes from review of ongoing annual investigations of contaminant loads in the creek.

During the first stage of this project historical data were assessed and relevant literature reviewed. Airborne gamma surveys have been flown in the area (Pfitzner & Martin 2000; Pfitzner et al 2001) and whereas there is a good knowledge of the status of the Rockhole residues (Tims et al 2000, Bollhöfer et al 2002) a tight line-spaced AGS from 2002 at Sleisbeck (Pfitzner et al 2003) was needed to be further groundtruthed to provide better information about the external gamma exposure of the public that may access the site.

It is possible that radionuclides leach from radioactive material dumped at Sleisbeck and the Rockhole tailings area into the underlying soil profile. Consequently, to achieve the required overall reduction of external gamma radiation, the originally deposited mine waste as well as any significantly containminated horizon of the underlying soil profile needs to be removed. Soil profiles have been sampled to determine the vertical distribution of radionuclide activity concentrations to provide a more rigorous estimate of the volumes of material to be excavated at Sleisbeck and the Rockhole tailings site.

Results

Sleisbeck mine

The high resolution airborne gamma survey (AGS) of the Sleisbeck site provided information about the extent of residual radiological contamination. The survey resulted in the collection of airborne radiometric data (eU, eTh, K and total count rates), magnetic, and digital elevation data. A flight line spacing of 25 m, with 250 m tie line spacing, and an aircraft height of 40 m were chosen for the survey, providing high spatial resolution of the images. A total of 600 km line data were obtained. A detailed ground based gamma radiation survey was conducted in 2006, to correlate count rates measured from the airborne platform with soil activity

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concentrations and gamma dose rates on ground, respectively. The ground-based gamma measurements were performed in the mid dry season when surface soil moisture content is very low. A portable 3”×3” NaI(Tl) spectrometer collected 512 channel gamma data over the energy range 0–3 MeV and external gamma dose rates were measured using environmental dose rate meters. Emphasis was given on the evaluation of the equivalent uranium (eU) data from the AGS. This channel represents gamma rays detected from the 1.73–1.76 MeV decay of ^{214}Bi. As ^{214}Bi is a radioactive progeny of ^{226}Ra, elevated count rates indicate the presence of ^{226}Ra in the soil, rather than uranium.

Count rates at Sleisbeck in the eU channel ranged from 17-2182 counts per second. High resolution Quickbird satellite data was acquired to relate the airborne count rates to the land cover features (Figure 1). Areas of highest count rates are largely confined to the mine and truck dump areas. Two surface ground transects covering the area of the three truck dumps were chosen in 2006 to provide ground level radiometric data to validate and calibrate the airborne gamma data acquired for the area.

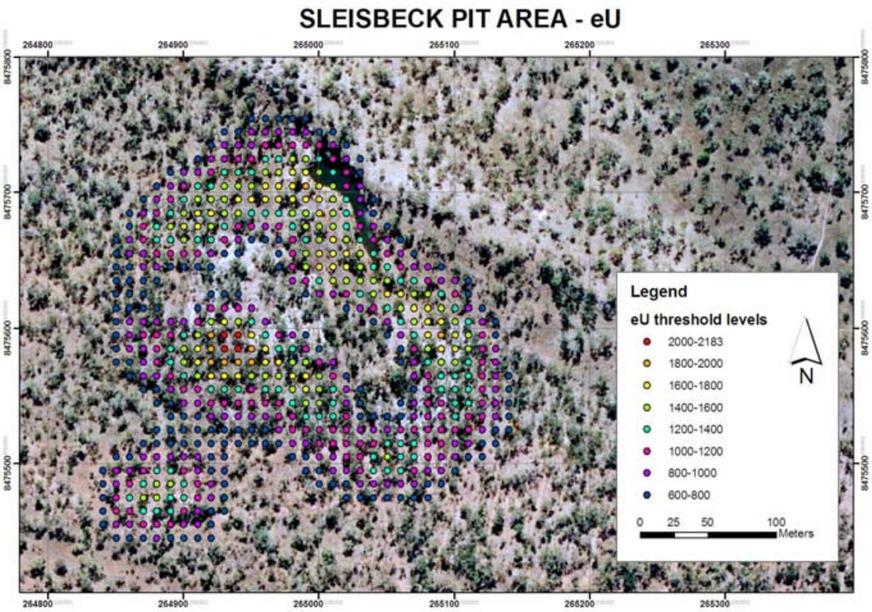

Figure 1 Quickbird data of the Sleisbeck pit area with the airborne countrates [s^{-1}] overlaid

Conversion factors to estimate terrestrial gamma dose rates from the airborne counts were derived using the ground data and used to determine area wide terrestrial gamma dose rates. These are shown in Table 1. Typical environmental background terrestrial gamma dose rates range from 0.10–0.14 µGy·hr^{-1}, whereas the area around Sleisbeck is characterized by values between 0.6–0.8 µGy·hr^{-1}. Maximum values from the AGS amounted to 2.3 µGy·hr^{-1}, but measurements on the ground indicate that this value is exceeded at localized areas within the old truck dumps.

Table 1 Average terrestrial gamma dose rates (total and contribution from eU, eTh and K-40) in Sleisbeck area

Site	γ dose rate [µGy/hr]	eU [µGy/hr]	eTh [µGy/hr]	K-40 [µGy/hr]	ΣeU,Th,K [µGy/hr]
Total area surveyed	0.12–0.17	0.10±0.02	0.07±0.02	0.011±0.002	0.18±0.03
Sleisbeck dump area	0.57–0.76	0.82±0.15	0.05±0.02	0.033±0.007	0.91±0.15
Environmental areas	0.10–0.14	0.08±0.01	0.06±0.02	0.007±0.002	0.15±0.02
Maximum	1.7–2.3	2.6±0.5	0.19±0.06	0.087±0.019	2.9±0.5

Trenches were dug at areas of highest measured surface activity concentrations, and profile samples collected down the wall of the trenches to determine the extent of leaching of radionuclides into the soil profile (Figure 2). This strategy was important from a remediation point of view to ensure that all of the contaminated material is removed and that any significant residual contamination of the underlying soil profile is also taken out.

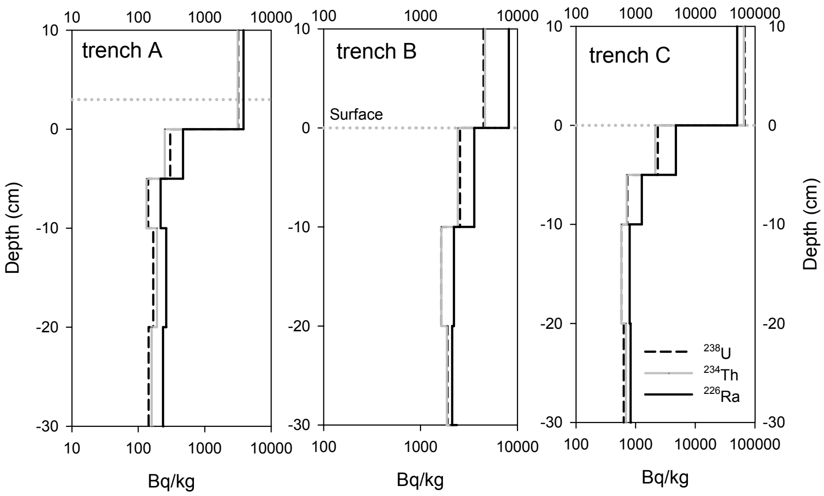

Figure 2 Radionuclide activity concentration profiles in trenches A, B and C

Based on visual inspection and dose rate measurements, a total truck dump area of approximately 1.2 ha needs to be rehabilitated. Taking into account the average pile heights at the dumps (~0.6–0.7 m) this will result in a total volume of material to be removed of approximately 8000 m^3. The investigation of the soil profiles taken along the trench walls showed that the maximum penetration depth of radionuclides is 5-10 cm below the waste rock – top soil interface (Figure 2). Scraping off 5 cm of topsoil underlying the dumps, will result in an additional 600 m^3 of material to be removed. Hence, the anticipated total volume of material to be relocated is likely to be less than 10 000 m^3.

The removal of mineralized material will lead to a reduction of terrestrial gamma dose rates. Assuming that ^{238}U, ^{232}Th and ^{40}K activity concentrations of the residual soil will be <600 Bq·kg^{-1}, 30 Bq·kg^{-1} and 130 Bq·kg^{-1}, respectively, an average terrestrial gamma dose rate of less than 0.3 µGy·hr^{-1} is expected. This is approximately 3 times lower than the average in the vicinity of the pit and truck dumps at present, and approximately 2–3 times above typical environmental background dose rates in the region. The results of this study have been accepted for publication in the Journal of Applied Radiation and Isotopes (Bollhöfer et al 2007).

Rockhole residues

A tightly spaced groundbased gamma survey was conducted at the Rockhole residues area in the South Alligator River valley at the end of June 2007, to determine the extent of radiological contamination at the former tailings dam footprint, south of Gunlom Road. Environmental dose rate meters were used at a resolution of approximately 10–15 metres. A total area of approximately 5 ha was investigated and a total of 262 readings were taken.

Four trenches were dug at, and in the vicinity of, identified hot spots in the area to determine the extent of penetration of radionuclides into the soil. These trenches were investigated in the field using a surface contamination probe, and are currently being analysed at the ***eriss*** laboratories in Darwin.

The aerial extent of radiological contamination has been mapped in a GIS environment, and. Figure 3 outlines the approximate areas that need to be remediated for various maximum dose rate guideline values. These guideline values represent approximately 10 times, 7.5 times and 5 times, respectively, of the natural background dose rates. The area increases drastically with decreasing guideline values. This result is displayed graphically in Figure 4 where the area to be rehabilitated is plotted versus various dose rate guideline values.

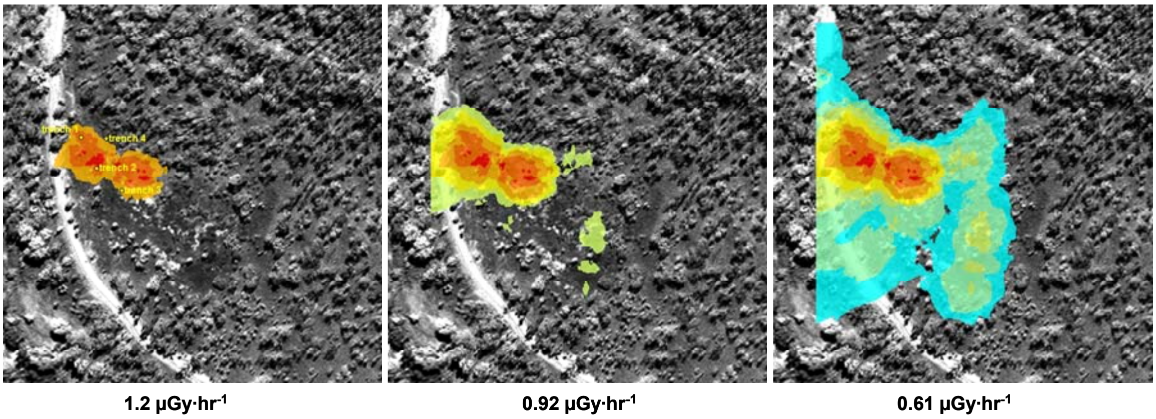

Figure 3 Approximate areas to be remediated at the Rockhole Residue site for various dose rate guideline values

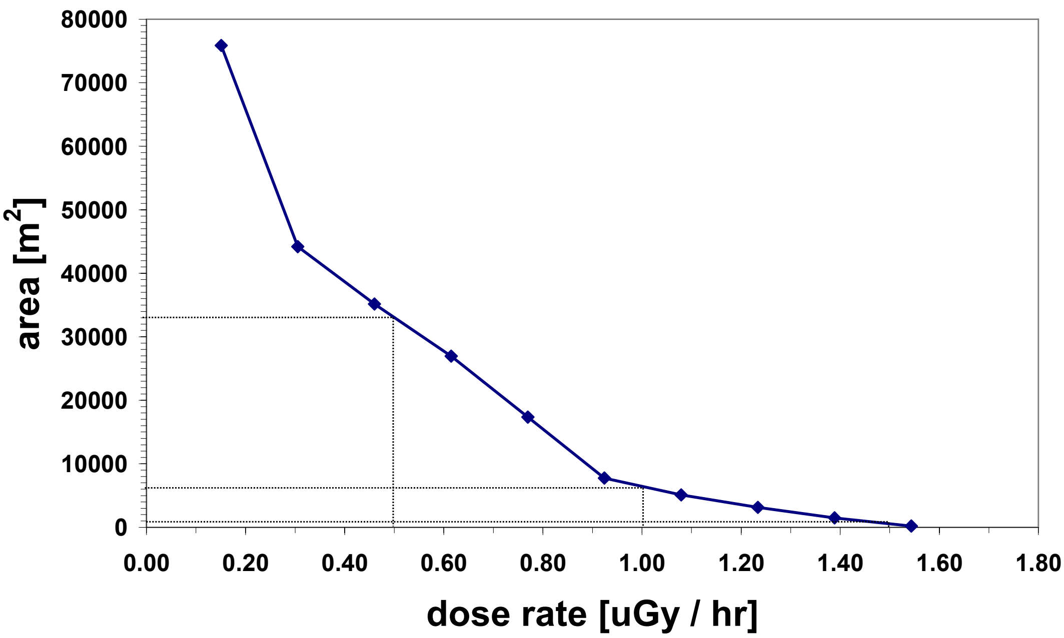

Figure 4 Area to be rehabilitated [m^2] plotted versus various dose rate guideline values

The total volume of material that needs to be removed will be calculated for varying terrestrial gamma dose rate guideline values, once analyses of samples from the trenches have been finalised.

Acknowledgments

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R van Dam, A Hogan & A Harford

Assessment of the effects of $MgSO_4$-rich wastewater from Argyle Diamond Mine to downstream aquatic ecosystems

C Humphrey & R van Dam

The Tropical Rivers inventory and assessment project (TRIAP)

R van Dam, R Bartolo & P Bayliss

AusAID Public Sector Linkages Program (PSLP) Project for Indonesian mine environment regulators

R van Dam & D Jones

The use of toxicity testing protocols to assess the effects of waste water discharges from Rio Tinto Alcan – Gove Operations

R van Dam, A Hogan & A Harford

Since 2005, SSD has been assisting Rio Tinto Alcan – Gove Operations (RTA–Gove; formerly Alcan Gove Pty Ltd) with the evaluation, development and application of appropriate tropical marine toxicity testing protocols for assessing the effects of alumina refinery discharges to the marine environment of Melville Bay, north-east Arnhem Land, Northern Territory. In 2006–07, two activities were undertaken for RTA-Gove:

1 Completion of a review of the status of tropical marine ecotoxicology in Australia (Marine Health Monitoring Program [MHMP] Project No 27); and

2 Commencement of a study to modify two tropical marine toxicity testing protocols and apply them to specific issues associated with RTA-Gove's main refinery discharge (MHMP Project No 34).

The tropical marine ecotoxicology review was a collaboration between SSD, Australian Institute of Marine Science (AIMS), Charles Darwin University (CDU), CSIRO Land & Water Centre for Environmental Contaminants Research (CECR) and Deakin University. The current state of the science was reviewed with regards to single species toxicity testing and biomarker assessment/monitoring using, or that could be adapted to, tropical marine species. The information was used to make recommendations for short- and long-term studies necessary to develop appropriate single species toxicity testing and/or biomarker assessment/monitoring protocols that could be applied to predict and measure ecological exposure and effects of RTA-Gove discharges in Melville Bay.

The second project, which commenced in March 2007 and is due for completion in March 2008, is looking at modifying existing sub-lethal toxicity test protocols for the tropical marine diatom, *Nitzschia closterium* (72-h cell division rate), and the coral, *Acropora tenuis* (3-h fertilisation & 18-h metamorphosis), to make them more relevant to environmental conditions at RTA-Gove (particularly temperature), and then applying the protocols to various issues associated with the main discharge. The project involves four short studies, assessing:

1 the species' upper thermal tolerance;

and, subsequently, at environmentally relevant temperatures, the effects on the species of exposure to:

2 three key metals, aluminium, vanadium and gallium;

3 the refinery discharge; and

4 an elutriate of sediments from within the discharge zone.

Information from these studies will help RTA-Gove build and enhance the industry's corporate knowledge base regarding the potential impacts of marine discharges. In addition, the project has the added value of contributing to an overall, longer-term aim of developing a larger suite of regionally-relevant toxicity test protocols for Australian tropical marine species.

Assessment of the effects of $MgSO_4$-rich wastewater from Argyle Diamond Mine to downstream aquatic ecosystems

C Humphrey & R van Dam

Since 2005, the SSD has undertaken a number of studies for Argyle Diamonds Pty Ltd (Argyle) in response to concerns that seepage waters arising from waste rock dumps could be impacting on downstream aquatic ecosystems and that this matter might need to be specifically addressed as part of the closure planning process. Magnesium sulfate ($MgSO_4$) is the primary component of the seepage. The desk-top, laboratory- and field-based studies have been designed to assess the ecological effects of the $MgSO_4$-rich seepage waters upon the receiving waters.

The collective results from the laboratory ecotoxicological and field ecological studies conducted on waters collected from the various water bodies, or in mine-exposed and reference streams adjacent to the Argyle mine, indicated a variety of responses from organisms to the altered conditions in the receiving waters. Responses measured in phytoplankton, zooplankton, macroinvertebrate and fish communities included enhancement, alteration and impairment, where the last-mentioned condition reflected loss of some taxa from exposed sites. Laboratory ecotoxicological results from exposures to Argyle discharge waters were broadly consistent with the field results. In particular the relative order of sensitivity of taxonomic groupings was consistent between field and laboratory results. There was also general consistency between field observations and laboratory predictions in terms of the water quality that should be sufficiently protective of aquatic biota.

It has been recommended that further work is needed to be able to (i) identify the major causes for observed effects of mine seepage waters under environmentally relevant conditions, and (ii) clearly distinguish mining (eg water quality, flow regime) from non-mining effects (eg loss of a potential downstream source of recruitment of stream-dwelling macroinvertebrate species as a consequence of impoundment of the Ord River). To address these issues the company has commissioned ***eriss*** to undertake further monitoring and field experiments designed to isolate and apportion the relative contributions that these factors make towards observed changes in receiving waters so that any required remediation efforts are appropriately focused.

The tropical rivers inventory and assessment project (TRIAP)

R van Dam, R Bartolo & P Bayliss

Background

'Australia's tropical rivers – an integrated data assessment and analysis' is more commonly known as the 'Tropical rivers inventory and assessment project' (TRIAP). TRIAP, funded by Land & Water Australia and the Natural Heritage Trust 2, is a collaborative effort between ***eriss***, James Cook University and the University of Western Australia, with additional involvement of the University of Wageningen in the Netherlands. The project aims to provide an information base to support the management of Australia's tropical rivers and consists 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 focus during 2006–07 was on sub-project 2, and a summary of this is provided below. Write-up of the final report for sub-project 1 is also currently underway by project collaborators at James Cook University. The final report for sub-project 3 was completed during 2006–07 and is being published in 2008 as a Supervising Scientist Report.

Study area

The TRIAP has operated primarily at two spatial scales: (i) the tropical rivers study area, comprising the 51 river/drainage basins for the Gulf of Carpentaria, Arafura Sea and Timor Sea; and (ii) focus catchments, namely the Fitzroy River (WA), Daly River (NT) and Flinders River (Qld) (see Figure 1).

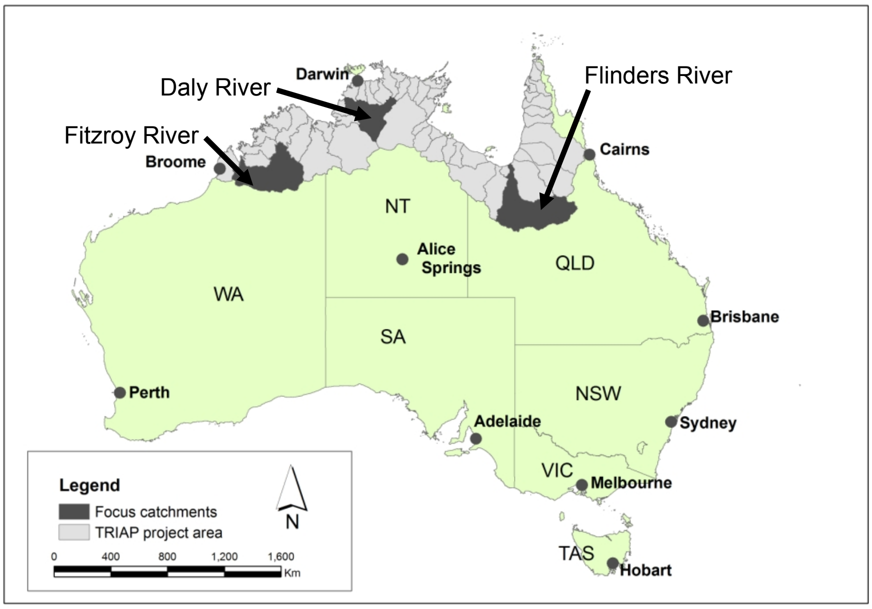

Figure 1 Delineation of the two scales at which the Tropical Rivers Inventory and Assessment Project is focusing: (i) the tropical rivers study area (catchments shaded light and dark grey); and (ii) focus catchments (shaded dark grey)

Sub-project 2 – Assessment of the major pressures on aquatic ecosystems

Objectives and scope

The objective of sub-project 2 is to develop a risk assessment framework applicable across multiple scales, which meets stakeholder needs, within the tropical rivers study area. In developing the risk assessment framework, semi-quantitative and quantitative risk analyses have been undertaken for selected ecological assets and threats. To support these analyses, the ecological assets of, pressures on, and threats to, the region's aquatic ecosystems have been described. Semi-quantitative risk analyses have been done for catchments across the whole of tropical rivers study area using data gathered during this sub-project and sub-project 1. The scope and timeframe of TRIAP allowed a detailed quantitative risk analysis methology to be developed and trialled only for the Daly River catchment.

Description of the region's ecological assets, pressures and threats

The major ecological assets, pressures and threats were identified and described for the northern tropical rivers region and each of the three focus catchments as an essential precursor to the risk assessments. As an example, a summary of the aquatic ecological assets (and associated key values) of the Daly River is shown in Table 1. The following pressures, and potential threats arising from them, were considered:

Pressures: horticulture, crop production, pastoralism, urban development, tourism and recreational and customary harvest, mining, climatre change.

Threats: groundwater extraction, surface water extraction, water impoundment, altered fire regime, land clearance/loss of native vegetation cover, introduced invasive flora, introduced invasive fauna, contamination.

Potential interactions between the ecological assets and the various threats arising from the pressures were depicted in an asset-threat matrix (not shown here, but will be presented in the final report).

Semi-quantitative risk assessments

Regional semi-quantitative ecological risk analyses for TRIAP were spatially explicit (ie. GIS-based), and hence employed a tool known as the Relative Risk Model (RRM). The RRM is a robust methodology that utilises spatial data to examine the interaction of multiple pressures/threats to key aquatic habitats, and their subsequent effects on ecological assessment endpoints (Landis & Wiegers 1997). The method has been shown to better focus investigative studies, data collection and the decision making process. The pressures/threats and key aquatic habitats are ranked in order of importance based primarily on their aereal extents. Relative risk estimates are determined by propogating pressure/threat and habitat ranks through the model for each ecological assessment endpoint, for each risk region being assessed. This process is based on conceptual models that link the three elements (ie pressures/threats, key aquatic habitats, ecological assessment endpoints), as depicted in Figure 2 for the Daly River.

Table 1 Summary of ecological assets and associated key values for the Daly River, as compiled from key literature sources

Asset	Details
Waterways	4860 km of riverine length, representing 9 geomorphic types, dominated by confined and constrained reaches and anabranching reaches. ***Additional values:*** *Freshwater discharge* – dependence of estuarine/marine fisheries on river discharge *Perennial flow* – discharge from underground aquifers and springs supports strong Dry season base flows in at least seven major rivers/creeks *Habitat for key species* – important species such as pig-nosed turtle, *Vallisneria nana* and various fish (eg. barramundi, freshwater sawfish) are dependent on riverine ecosystems *Water quality* – high, seasonally variable water quality; low ionic strength/alkalinity upstream of Daly basin (granite/sandstone aquifers), high ionic strength/alkalinity within Daly basin (limestone/dolostone aquifers)
Wetlands	Extensive, diverse and largely intact wetland complexes that are important in maintaining biodiversity, and that include 1 wetland complex of national importance ***Additional values:*** *Wildlife nurseries & habitat* – diversity of different wetland types provide a range of habitats and resources for different species under different conditions and at different times. *Erosion control / sediment retention* – floodplains and swamps reduce erosive power of surface runoff and trap sediments before reaching river channels *Water regulation* – the absorbent, dispersive and flow reduction characteristics of wetlands help retain water in the system and attenuate floods
Riparian vegetation	Diverse, largely intact vegetation communities that comprise 289 000 ha within the catchment; they support very high biodiversity and endemism relative to their exent, and have many important ecological and hydrological functions; vine thickets represent a particularly important riparian community ***Additional values:*** *Erosion control* – riparian vegetation increases bank stability and reduces flow velocity, minimising downstream sedimentation *Habitat for wildlife* – provide shade, nutrients and submerged habitat for aquatic species, and act as corridors and refuges for terrestrial an semi-aquatic species
Biodiversity	Waterbird status of the lower Daly River floodplain satisfies requirements for Ramsar listing; the Daly River supports the largest pig-nosed turtle population in Australia, and contains more species of turtle (8) than any other Australian river; 48 species of estuarine/freshwater fish, including the rare strawman
Threatened species & conservation reserves	Numerous EPBC-listed aquatic/semi-aquatic species, including freshwater sawfish, speartooth shark, northern river shark, freshwater whipray, false water rat; other species of significance include pig-nosed turtle, 'blackmast' strawman, exquisite rainbowfish, and two plant species – *Vallisneria* and *Spirogyra*. At least 10 conservation parks and reserves are located in the catchment.
Limestone and karst habitat	Extensive groundwater aquifers characterised by surface and subterranean karstic features such as vertical shafts, losing streams, springs, dolines, caves and solution sculptured limestone rock; the karst geology is of great importance to the catchment's hydrological regime; stygofauna are present but not well characterised

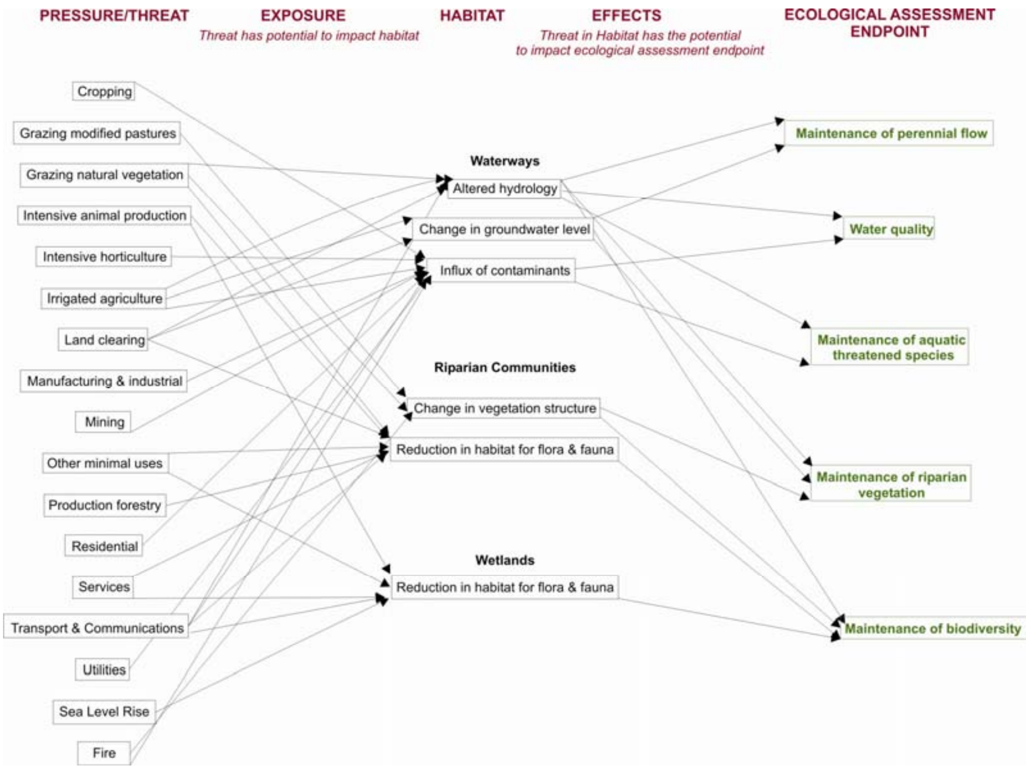

Figure 2 Conceptual model describing linkages between pressures/threats, key aquatic habitats and ecological assessment endpoints for the Daly River catchment

Risk characterisation enables calculation and subsequent comparison of risk estimates among risk regions, pressures/threats, habitats and endpoints to identify:

1 the risk regions where most risk occurs;

2 the pressures/threats contributing the most risk;

3 the habitats where most risk occurs; and

4 the ecological assets most at risk in the study area (Hayes & Landis 2004).

The RRM has been applied at both the tropical rivers study area scale, comparing 51 risk regions (drainage basins) and at the Daly River catchment scale, comparing 18 risk regions (primarily sub-catchments). In addition, various sensitivity and uncertainty analyses have been undertaken to test the rigour and robustness of the approach. As an example of the RRM outputs, Figure 3 shows the relative risk for the sub-catchments of the Daly River based on the various pressures/threats assessed. The three current key pressures/threats to the catchment are: grazing of native vegetation; transport and communications (eg. roads, railways, airports/aerodromes); and land clearing.

Quantitative ecological risk assessments

At the sub-regional scale, and/or when dealing with only one or two threats to a similar number of ecological assets/assessment endpoints, it should be feasible and appropriate to undertake more detailed quantitative ecological risk analyses. Three such analyses were undertaken for selected threats and assets for sub-regions of the Daly River catchment, as follows:

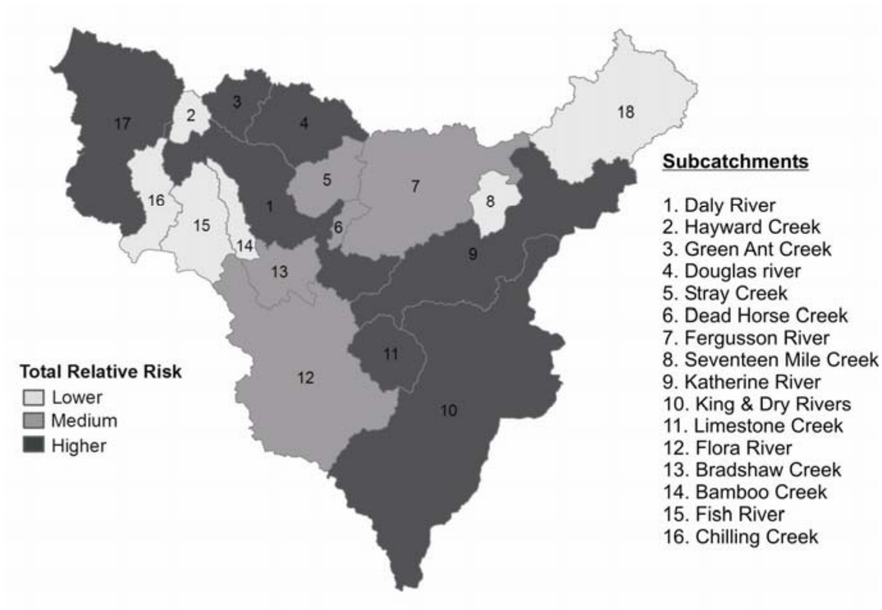

Figure 3 Total relative risk shown as higher, medium or lower for the 18 risk regions within the Daly River catchment. Refer to Figure 2 for the conceptual model that guided this assessment of relative risk.

i Risks of surface water extraction and weeds on floodplain health;

ii Risks of surface water extraction on in-stream health; and

iii Risks of land clearing on surface water quality.

Further details of the analyses are provided in Table 2. All the analyses focused on the use of empirical (data driven) ecological process models where data allowed (eg. relationships between wet season river flow and magpie goose nest density) and Bayesian Networks (BNs) to integrate the quantitative information with qualitative expert knowledge. BNs have proved broadly applicable in almost every ecological field where a decision problem involves taking risks in the face of uncertainty, variability and complexity (Burgman 2005).

Table 2 Summary of key ecosystem assets and threats, and ecological and measurement endpoints, used for quantitative ecological risk assessment of aquatic ecosystems in the Day River catchment

Ecosystem asset	Key threat	Ecological assessment endpoint	Ecological measurement endpoint
FLOODPLAIN	Water extraction	Magpie goose nesting success	Magpie geese nest density (numbers/km^2)
	Wetland weeds	Condition of magpie goose nesting habitat	% or area (ha) of nesting habitat displaced by wetland weeds
		Condition of magpie goose dry season habitat	% or area (ha) of refuge habitat displaced by wetland weeds
		Plant biodiversity	Risk probability (=exposure x effect) or % of plant species lost in infected area
INSTREAM	Water extraction	Barramundi catch success	Recreational & commercial barramundi catch
	Water extraction	Sustainable barramundi stock	Barramundi CPUE[1] population index
	Land clearing for land use	Surface water quality	Modelled total sediment & phosphorus exports (t/y) & modelled DIN[1] concentration (μM)

1 CPUE: Catch Per Unit Effort; DIN: dissolved inorganic nitrogen

Some results from the risk assessment of the effects of surface water extraction and wetland weeds on floodplain health are provided below. The risk assessment focused on three ecological assessment endpoints: (i) the health of magpie goose nesting success in the wet season (in relation to potential flow extraction and extent of weeds); (ii) the health of magpie goose dry season refuge habitat and (iii) plant biodiversity (in relation to weeds only). Based on existing data, predictive ecological process models were developed for the following processes:

- Effect of natural flow regimes on magpie goose nest density and the availability of geese to breed;
- Effect of the weeds, mimosa (*Mimosa pigra*) and paragrass (*Brachiaria mutica*) on magpie goose nesting and dry season refuge habitats; and
- Effect of weeds on floodplain plant biodiversity.

A BN for floodplain health was constructed that incorporated the three ecological assessment endpoints. Four scenarios were simulated to assess the independent and combined risks to floodplain health from flow extraction and wetland weeds (ie 0% and 20% flow extraction in the absence and presence of weeds). The BNs for the scenarios ‘0% water extraction, weeds absent’ and ‘20% water extraction, weeds present’ are shown in Figure 4. A simulated 20% wet season flow extraction had little overall influence on floodplain health either in the presence or absence of weeds. The major influence on floodplain health was the extent of weeds. The BN was extended to include nodes that allowed examination of the costs and benefits of different weed control scenarios. A control strategy that aimed for a 10% residual cover of mimosa significantly increased from 4% to 72% the probability of the Daly River floodplain being in ‘Good’ condition, at an initial cost of $0.75 million.

The approach adopted throughout the TRIAP was compatible with the project aim of developing appropriate analytical tools and establishing a framework that can be modified to accommodate different issues across different regions and scales, and different stakeholder perspectives.

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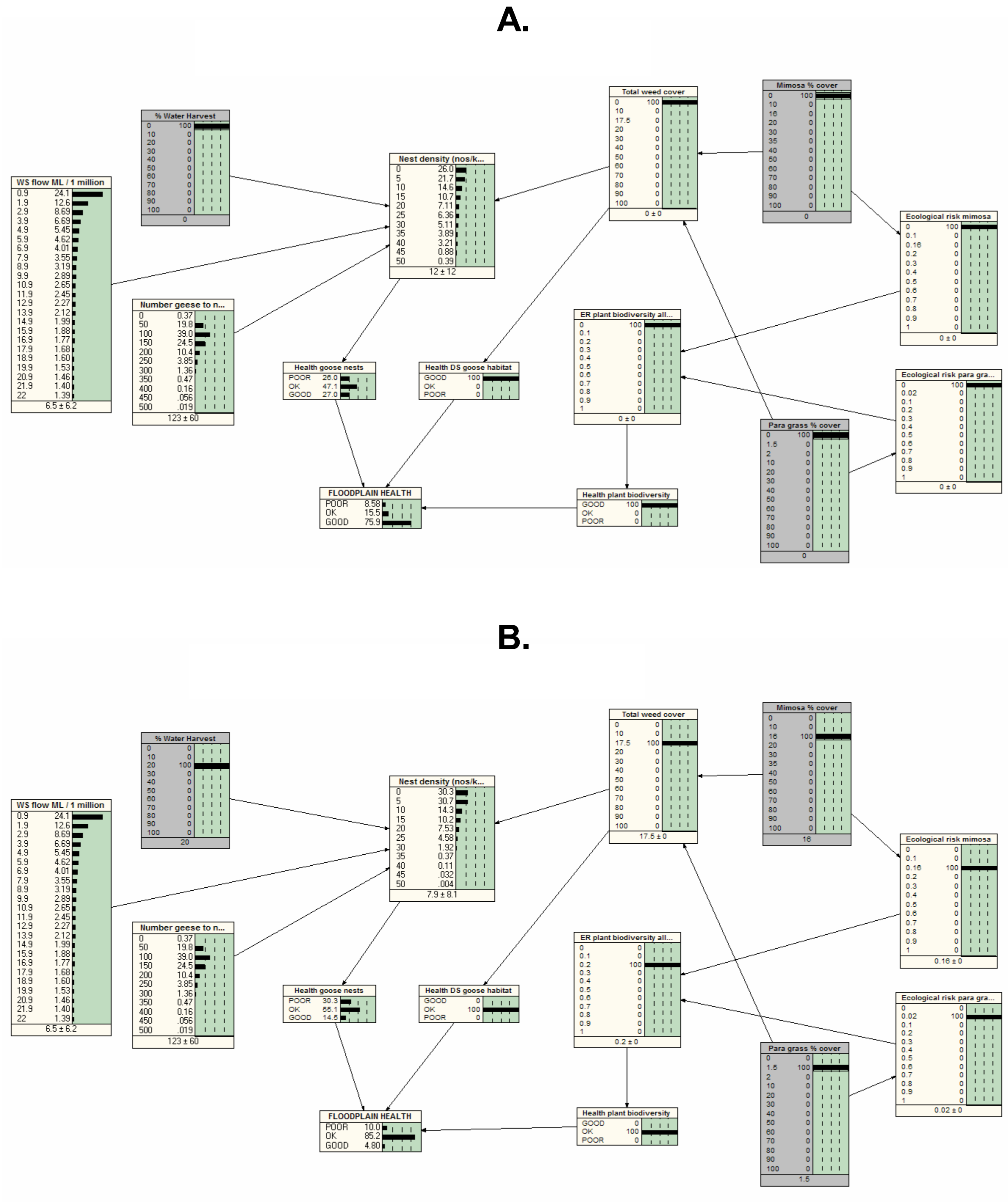

Figure 4 Bayesian Network for Daly River Floodplain Health based on two scenarios: (A) no weed effect and no surface water extraction; and (B) weed effects and 20% surface water extraction

AusAID Public Sector Linkages Program (PSLP) Project for Indonesian mine environment regulators

R van Dam & D Jones

Background

In 2006, SSD received funding from the AusAID Public Sector Linkages Program (PSLP) to undertake a training program for mining environmental officers from the Indonesian Ministry of Environment (MoE) and provincial environmental protection agencies. The training program aims to equip course participants with the knowledge and skills necessary to:

- appropriately assess, monitor and regulate environmental aspects of mining activities in Indonesia;
- train others to undertake such assessment and regulation.

The Activity comprised a four week training course in Darwin during November-December 2006, with a 10-day follow-up visit by two Australian officials to Indonesia, to provide subsequent assistance, advice and feedback, in August 2007. It was a collaboration between SSD, Charles Darwin University (CDU) and the Northern Territory Government (Department of Primary Industry, Fisheries & Mines, Department of Natural Resources, Environment & the Arts – EPA Program).

The official opening of the Indonesian Mining Environmental Training Course took place at CDU on Monday 20 November 2006. The training program was officially launched by the Hon David Tollner MP, Federal Member for Solomon.

Training course

Fifteen MoE/provincial mining environmental officials undertook four weeks (20 November – 15 December) of training in and around Darwin. The training course consisted of two parts: an 11 day technical training program in 'Mining environmental regulation and monitoring', followed by a 9 day training program in 'Workplace assessment and training' (ie train-the-trainer). Three key themes were covered during the technical course: *Minesite environmental regulation*, *Water quality impacts, monitoring & assessment* and *Minesite closure and rehabilitation*. These themes were selected following consultation with MoE. In addition to lectures and workshops, the technical training program included field visits to four mine sites in the Top End of the Northern Territory: the operational Ranger Uranium Mine; the abandoned Mt Todd Gold Mine; and the rehabilitated Rum Jungle and Woodcutters Mines. Figure 1 shows the course participants and SSD presenters at the abandoned Mt Todd Gold Mine, approximately 250 km south-east of Darwin.

The train-the-trainer course comprised four Units of Competence from the BSZ Training Package, Certificate IV in *Workplace Training and Assessment*, and was delivered by staff from CDU. The Units represented the minimum training requirement for a person to be able to (i) deliver training and (ii) assess the persons having received the training to determine their level of competence or understanding of the subject matter. All course participants easily met the attainment requirements for the Units.

Figure 1 PSLP mining environmental monitoring and regulation course participants and SSD presenters at Mt Todd Gold Mine, NT, 29 November 2006

Follow-up visit to Indonesia

From 19-31 August 2007, two Australian project team members, Dr David Jones (Supervising Scientist Division, SSD) and Mr David Dettrick (NRETA – EPA Program) travelled to Indonesia for the follow-up visit to the training course. Various meetings, workshops and visits were held during this period, including:

- discussions on the progress of MoE and the individual course participants in developing a training curriculum for a training course to be run locally and rolled out to other Indonesian MoE and provincial environmental regulators;
- workshops on various topics (with both large and smaller groups of staff from mining-related departments in MoE) related to minesite condition and rehabilitation, with a focus on site contamination criteria, rehabilitation criteria and water quality, with several presentations on these issues;
- a visit to MoE's training centre and analytical reference and services laboratories, with further discussions on the development of the MoE's mining environmental training curriculum; and
- an inspection visit to a medium sized coal mine, PT Indominco Mandiri, in Bodang in East Kalimantan to compare and discuss Australian perceptions of the company's environmental performance with that of the Indonesian regulators.

A draft curriculum for a five day technical training course has been developed by MoE and Provincial agency staff, with the first course expected to be run in the second half of 2008, with 30 participants. The ultimate objective of MoE is to train up to 500 staff within the Jakarta Office and in the Provinces.

During the latter stages of the visit, discussions centred around follow-on activities and collaborations arising from this Activity, including more specialist training courses (eg. in environmental licencing or biological assessment), the development of a 'sister' EPA program between the NRETA EPA Program and MoE, and exchange programs for MoE and NRETA and SSD staff to develop capacity in environmental approvals and assessment in different types of governance regimes.

Outcomes

From a survey conducted following the training course, and feedback provided to, and observations of, the Australian Officials during the follow-up visit to Indonesia, it was clear that the course participants gained valuable relevant knowledge from the Activity, and that the Activity has represented a small but significant step towards MoE improving its capacity in mining environmental regulation and monitoring. Future opportunities for follow-on activities will be considered in 2008 in the context of Departmental strategic priorities.

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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 membership 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 Environment and Water Resources] 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 2004–2006

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 and final rehabilitation expected to be completed by about 2016.
- 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:

1 maintain the attributes for which Kakadu National Park was inscribed on the World Heritage list;

2 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);

3 protect the health of Aboriginals and other members of the regional community; and

4 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:

5 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;

6 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;

7 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 program. 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

KKN 1.1.1 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 Owners derive a significant proportion of their food from bush tucker.

KKN 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

KKN 1.2.1 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.

KKN 1.2.2 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.

KKN 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 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.

KKN 1.2.4 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.

KKN 1.2.5 Assurance program 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 program 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.

KKN 1.2.6 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

KKN 1.3.1 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 programs although there is scope for some specific research and analysis in relation to the review of the occupational radiological monitoring program. More specifically, ARRTC supports the design and implementation of a new risk-based radiological monitoring program 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. 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

KKN 2.1.1 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.

KKN 2.1.2 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 programs listed below.

KKN 2.1.3 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.

KKN 2.1.4 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.

KKN 2.1.5 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.

KKN 2.1.6 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.

KKN 2.1.7 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.

KKN 2.1.8 Final landform design

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 Ecosystem establishment

KKN 2.2.1 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.

KKN 2.2.2 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.

KKN 2.2.3 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.

KKN 2.2.4 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 program. 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

KKN 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 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.

KKN 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. 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.

KKN 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.

KKN 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

KKN 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.

KKN 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. 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 program 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 program.

KKN 2.5.1 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.

KKN 2.5.2 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 program 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

KKN 3.1.1 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 program (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 program to collect baseline data required before development such as meteorological and sedimentary data.

3.2 Research

KKN 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 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

KKN 4.1.1 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.

KKN 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 should continue. The outcomes of this research will be very relevant to Ranger.

4.2 Assessment of radiological, chemical and geomorphic success of rehabilitation

KKN 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 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 programs, a potential major future research area is the possible development of broader, landscape scale programs that would enable possible effects of mining to be distinguished from those arising from other causes. Such a program 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 program, a review of the program to assist in determining future priorities needs to be undertaken.

KKN 5.1.1 Re-assess and prioritise the landscape program

A review is required, within a modelling conceptual and risk assessment framework, of the landscape wide program to determine options and priorities for the future development of this program.

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 program. 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.

KKN 5.2.1 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 program and occasionally conducts a low level radiological monitoring program, primarily for assurance purposes. ARRTC believes these should continue.

5.3 Develop monitoring program 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.

KKN 5.3.1 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 program could be established and the value of Koongarra as an analogue for pre-mining radiological conditions at Ranger could be investigated.

KKN 5.4.1 Baseline monitoring program for Koongarra

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 may also have some relevance as a control system for comparison to Ranger, Jabiluka and Nabarlek.

KKN 5.4.2 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

KKN 6.1.1 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

KKN 6.2.1 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

KKN 6.3.1 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

KKN 6.4.1 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.