Part 1: Ranger – current operations

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1 List of papers grouped by Key Knowledge Need

Development of a contaminant pathways conceptual model for Ranger uranium mine

R van Dam & P Bayliss

The ARRTC Key Knowledge Need (KKN) 1.2.1 states that:

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

This paper summarises the progress on the contaminant pathways conceptual model only. The assessment of the ecological risks via the surface water pathway was reported to ARRTC at the 16th Meeting in September 2005, and has only been briefly reported on again, in KKN 5.1 'Undertake an ecological risk assessment of Magela floodplain to differentiate mining and non-mining impacts'.

Background

A conceptual model of contaminant pathways from the operational phase of Ranger uranium mine is being developed. Progress has been documented by Finlayson and Bayliss (2003), van Dam et al (2004) and van Dam et al (2006). The primary purpose of the conceptual model is to place the off-site contaminant issues at Ranger in a risk management context. Moreover, the final product, as well as being used for formal risk assessment (see summary for ARRTC KKN 5.1.2), will serve as a communication tool for both scientists and traditional owners.

Progress

A meeting was held on 31 May 2006 between key ***eriss*** and EWL Sciences staff to discuss the technical aspects of the model. Good feedback was received and, on the whole, EWL Sciences affirmed the technical information contained within the model. Consultation with Traditional Owners to obtain their inputs on the content of the model has not progressed as far as originally planned through 2005–2006 owing to the higher priority that had to be given to initiating the consultation process for closure planning.

Given that the current model represents probably the last iteration of a conceptual model of contaminant pathways for the operational phase of the Ranger mine, and has already been verified as being technically appropriate, it could be considered more important to focus Traditional Owner consultation time on the subsequent phase of this activity. That is, developing a contaminant pathways conceptual model for the closure and rehabilitation phase of the mine.

Steps for completion

If possible and considered a priority, consultations will still be held to seek Traditional Owner views and opinions on the issues and the conceptual model itself. A number of minor revisions/amendments will be made to the conceptual model, and the surface water pathway sub-model will be refined and incorporated into a comprehensive journal publication on the Magela floodplain ecological risk assessment.

References

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Chronic toxicity of uranium to *Lemna aequinoctialis* and *Amerianna cumingi*

R van Dam, A Hogan, M Houston, S Nou[1] & N Lee

Background

Historically, uranium has been the primary toxicant of concern for the aquatic ecosystems downstream of Ranger uranium mine (van Dam et al 2002). Consequently, many ecotoxicological studies had been undertaken for almost 20 local aquatic species, to assess the toxicity of uranium and the influence of selected environmental variables (eg pH, alkalinity, water hardness, dissolved organic carbon) on the toxicity of uranium.

However, the majority of data for these species were derived from acute toxicity rather than chronic toxicity test endpoints. The latter are required for the derivation of *high reliability* water quality guidelines (ANZECC & ARMCANZ 2000). Chronic data exist for only five of those species tested: namely the green alga, *Chlorella* sp. (72-h growth inhibition); the cladoceran, *Moinodaphnia macleayi* (3-brood reproductive inhibition); the green hydra, *Hydra viridissima* (96-h population growth inhibition); the chequered rainbowfish, *Melanotaenia splendida inornata* (7-d survival and growth); and the purple-spotted gudgeon, *Mogurnda mogurnda* (7-d exposure/7-d post-exposure survival and growth).

Based on a cumulative probability (loglogistic) distribution of no-observed-effect-concentration (NOEC) data for these five species that range from 18 to 810 µg/L, a site-specific water quality trigger value (TV) for uranium of around 6 µg/L has previously been derived (Hogan et al 2005). This value represents the concentration that should be protective of 99% of species with 50% confidence. Notably, the TV has high uncertainty surrounding it, as demonstrated by the 95% confidence limits of 0.3–103 µg/L. Moreover, two of the five species represented are fish, which appear to be generally less sensitive to uranium than invertebrate and algal species. Thus, in order to increase confidence in the site-specific TV, chronic toxicity data for additional species, ideally, representing additional taxonomic groups and trophic levels, were required.

The tropical duckweed *Lemna aequinoctialis* is a small aquatic floating macrophyte that occurs in lentic and low-flow waterbodies throughout northern Australia (Cowie et al 2000), including the Alligator Rivers Region (ARR). The freshwater snail *Amerianna cumingi* is a hermaphroditic snail that occurs in lentic and lotic waterbodies within a restricted range that encompasses the ARR (Smith 1992). Both species are of high ecological importance as food sources for other organisms and in their respective roles as a primary producer and detritivore. Toxicant effects on these species are assessed in the laboratory by observing the growth of exposed *L. aequinoctialis* and changes in the egg production of *A. cumingi* over 96 h and comparing them with individuals of the same species maintained in clean water (Riethmuller et al 2003, Houston et al 2007).

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Fate of uranium in test system

In order to accurately calculate exposure concentrations throughout the tests, it was essential that the fate of uranium in the test system was understood. Adsorption of uranium to the test container can reduce the uranium dissolved in the test waters and result in the organisms being exposed to a lower than expected concentration. Without quantifying this loss, the toxicity of uranium could be significantly underestimated. It is important to note that, in addition to adsorption of uranium to the test containers/tubes, losses of uranium from the test waters can also be due to uptake and accumulation by the test organisms. As the uptake of uranium by the organisms represents the exposure to uranium, the relative proportions of uranium 'lost' from the test waters due to (i) adsorption to test containers/tubes and (ii) accumulation by the test organisms need to be determined before exposure concentrations can be appropriately adjusted. To address this, uranium concentrations were measured in test waters periodically throughout each test and in the duckweed/snail tissues at the conclusion of a test.

A small but significant loss of uranium (8–18%, $P < 0.05$) was detected in the *L aequinoctialis* test system over the four day duration. Samples taken at 48 h indicated that the majority of the uranium was being lost within the first half of the test, after which uranium concentrations remained relatively stable. When integrated over a four day period (by calculating the area under the curve), these losses ranged from 6–13%, with the proportion lost positively correlated with the initial uranium concentration in the water. Plant tissue measurements indicated that uranium uptake by the plants accounted for approximately 50% of the uranium 'lost' from the test waters. As the overall 'loss' was quite small, and a significant proportion of this was shown to be taken up by the plants, it was decided that no adjustment of the exposure concentrations would be required.

A more substantial loss of uranium was observed from the test waters in the *A. cumingi* tests, with samples taken 24 h after each water change containing 30–70% less uranium than at the start of the test. Figure 1 shows an example of uranium loss for one of the uranium treatments over the 96 h test duration. Because waters are changed daily during *A. cumingi* experiments, and the loss of uranium over each 24 h period was found to be gradual and to decrease in magnitude over the duration of the test, it was essential that the final losses were integrated over each 24 h period, and then over the entire 96 h before calculating the exposure concentrations. Using data from three different experiments, and regardless of whether total or dissolved uranium concentrations were used, it was found that uranium loss over the entire test duration was approximately 25% of the uranium concentration at the start of the test.

An experiment designed to address the uptake of uranium by the snails was being conducted at the time of preparation of this summary and hence could not be reported at this stage. Therefore, the toxicity results reported below are based on uranium concentrations adjusted according to the total losses measured (ie. corrected concentration is 75% of the initial uranium concentration). Should a significant proportion of the 'lost' uranium be found to have been taken up by the snails, then the exposure concentrations will be adjusted accordingly

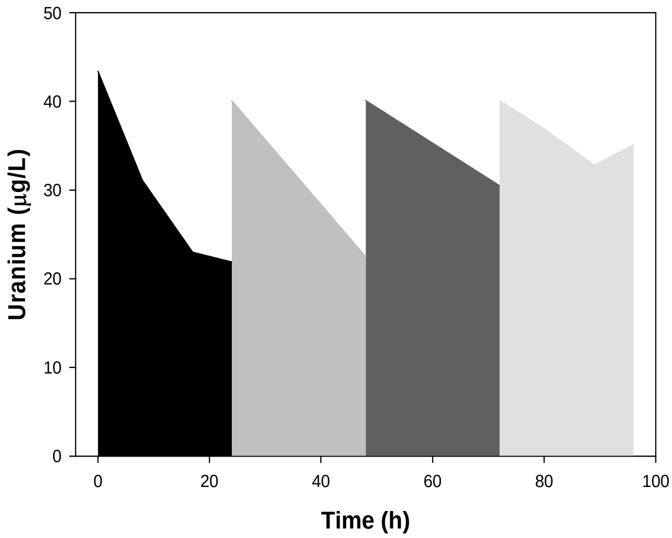

Figure 1 Loss of uranium in an *A. cumingi* test solution initially containing 43 µg/L uranium (measured). Each shaded bar represents a 24 h period between water renewals.

Toxicity of uranium to *L. aequinoctialis*

The toxicity of uranium to *L. aequinoctialis* was reported to ARRTC at the 16th Meeting in September 2005, and is summarised again here for reference. The effect of uranium exposure on the growth of *L. aequinoctialis* is shown in Figure 2. An IC_{10} (concentration resulting in a 10% inhibition of egg production) of 250 (lower/upper 95% confidence limits: 207/288) µg/L uranium was calculated from these data. The IC_{10} is generally considered to be a measure of an 'acceptable' concentration (ie. one that will not result in unacceptable ecological effects at the population level). The IC_{50} could not be calculated, but was >2850 µg/L uranium. When compared to the other local freshwater species that have been assessed for their sensitivity to uranium, *L. aequinoctialis* was found to be less sensitive than most. Only the two fish species, the northern trout gudgeon (*Mogurnda mogurnda*) and the chequered rainbowfish (*Melanotaenia splendida inornata*), have been reported to be less sensitive.

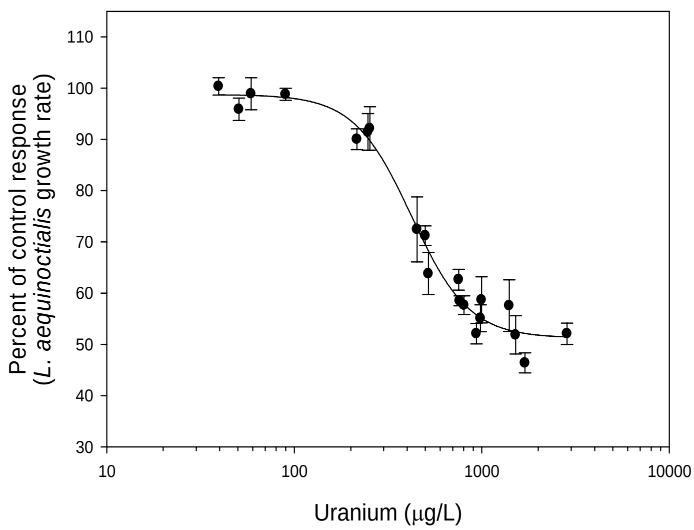

Figure 2 Effect of uranium on the growth rate of *L. aequinoctialis*, expressed as a percentage of the control response (Control growth rates for the three tests were 0.43, 0.42 and 0.45, respectively). The fitted curve represents a 4 parameter loglogistic model (r^2 = 0.98, n = 20, P<0.0001).

Toxicity of uranium to *A. cumingi*

The effect of uranium exposure to *A. cumingi* is shown in Figure 3. Based on the four definitive tests, *A. cumingi* was found to be highly sensitive to uranium, with an IC_{10} of 22 (lower/upper 95% confidence limits: 6/46) µg/L uranium and an IC_{50} of 250 µg/L uranium (an upper confidence limit could not be calculated). Based on these data, *A. cumingi* appears to be more sensitive to uranium than most other species that have been tested. Of the five species already used to derive the current uranium TV, only the water flea *Moinodaphnia macleayi* has been found to exhibit similarly high sensitivity to uranium. It is noteworthy that although the intra-treatment responses of *A. cumingi* tend to be inherently highly variable (as evidenced by the large error bars in Figure 3), the inter-test concentration-response relationships are quite consistent (and in fact were not significantly different from each other based on analysis of covariance; F = 0.563, df = 3, P = 0.647).

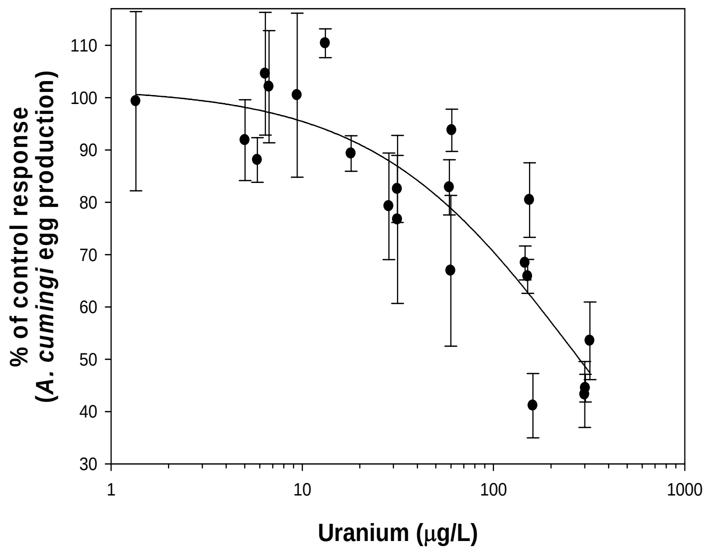

Figure 3 Effect of uranium on the egg production of *A. cumingi*, expressed as a percentage of the control response (control egg numbers for the three tests were 198, 133 and 241, respectively). The fitted curve represents a 3 parameter loglogistic model (r^2 = 0.75, n=21, P<0.0001).

Steps for completion

Analysis of the results from the snail uranium uptake experiment will be completed in 2006–2007. A further experiment to quantify the amount of uranium taken up by the lettuce fed to the snails throughout the tests will also be undertaken. The uranium exposure concentrations for the toxicity tests will be adjusted if necessary to account for losses of uranium from solution during the tests and the associated toxicity estimates recalculated. The results for both the *L. aequinoctialis* and *A. cumingi* experiments will be submitted for publication in an international journal.

A paper summarising the uranium toxicity results for *L. aequinoctialis* and *A. cumingi* has recently been presented by Alicia Hogan at the Interact 2006 conference in Perth.

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Chronic toxicity of uranium in Magela Creek water to a local freshwater fish

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

Background

Two of the five NOEC values currently used to derive the *high reliability* site-specific water quality limit for uranium in Magela Creek represent estimates for two fish species based on mortality after only relatively short exposure periods (Holdway 1992). The NOEC of 400 µg/L for the purple-spotted gudgeon (*Mogurnda mogurnda*) was based on a 7-d exposure/7-d post-exposure test design, while the NOEC of 810 µg/L for the chequered rainbowfish (*Melanotaenia splendida inornata*) was based on a 7-d exposure test design. Although such estimates satisfy the current ANZECC/ARMCANZ Water Quality Guidelines criterion for a 'chronic' endpoint, their appropriateness as indicators of longer-term, sub-lethal effects is highly questionable. Consequently, to increase our confidence in the uranium limit, an Honours project was initiated to investigate the sub-lethal effects of uranium on one or both of the above two fish species, over an exposure duration of 28 days.

Methods

The project was divided into several discrete parts: (i) development of an appropriate 28-d protocol (based around OECD (1998)); (ii) assessment of uranium toxicity over 28-d; and (iii) derivation of a revised uranium limit. Part 1 involved studying certain test conditions (the diet/feeding regime of the test animals and the stocking density of the test animals) in order to optimise larval growth and minimise adverse water quality impacts, and characterising the fate of uranium in the test system. Larval growth, measured as body length and wet and dry weight, was selected as the primary test endpoint.

Progress

Kim Cheng, from CDU, co-supervised by Professor David Parry, commenced the project in April 2006. Initial test development was focused on *M. mogurnda*, as this species was already established in the ***eriss*** ecotoxicology laboratory. The first experiment showed that larval *M. mogurnda* survived well on a diet of *Artemia* (brine shrimp) nauplii. A second experiment demonstrated that additions of *Artemia* over different time periods (2, 4 and 6 hours) did not significantly affect the concentration of dissolved uranium in the test waters. Unfortunately, the *M. mogurnda* broodstock ceased spawning after these initial two tests.

After several weeks with no further spawning, a decision was made to switch to chequered rainbowfish as the test species. A permit was obtained and approximately 250 adult and juvenile fish were collected from the Nourlangie Creek catchment and established as a broodstock. The broodstock spawned almost immediately, and test development re-commenced. A new set of experiments was developed, focusing on optimising the diet of

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[1] Charles Darwin University, Darwin NT.

larval rainbowfish (which typically require a more diverse diet consisting of smaller-sized food particles than larval *M. mogurnda*). The majority of work since then has focused on this aspect of the test method, with limited success. The problems resulted in substantial delays and, as a consequence, Kim switched to part-time studies.

Steps for completion

A further series of experiments was planned to assess the effect of larval stocking density on larval survival and growth, and to incorporate advice received from outside experts (Doug Holdway and Craig Humphrey) on how to improve survival and growth. If these experiments fail to improve larval survival and growth, the project will probably revert to using *M. mogurnda*, as the broodstock recommenced regular spawning in August 2006.

References

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

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

Background

Over the past three years, the ***eriss*** ecotoxicology laboratory has been developing and maintaining a regular reference toxicity testing program for the species comprising the routine toxicity testing suite. The reference toxicant being used is uranium. Reference toxicity testing during 2004–05 focused on the green alga, *Chlorella* sp., and the green hydra, *Hydra viridissima*. This summary outlines the progress made during 2005–06.

Progress

The results of the reference toxicity tests completed in 2005–06 are summarised in Table 1.

Table 1 Uranium reference toxicity test results during 2005–06 for *Chlorella* sp., *Hydra viridissima*, *Lemna aequinoctialis* and *Mogurnda mogurnda*.

Species	Endpoint	IC_{50}[1]/LC_{50}[2] (µg/L)	Test Code
Green alga (*Chlorella* sp.)	72-h cell division rate	Invalid test[3]	708G
		Invalid test[4]	712G
Green hydra (*Hydra viridissima*)	96-h population growth	75 (65–97)[5]	707B
		Invalid test[3]	717B
		79 (69–88)	719B
		83 (76–88)	725B
Duckweed *(Lemna aequinoctialis)*	96-h population growth	>854	716L
		Invalid test[3]	732L
		763 (156–1349)	739L
Northern trout gudgeon *(Mogurnda mogurnda)*	96-h sac fry survival	1785 (1604–1943)	724E

1 Concentration that causes a 50% inhibition of the test endpoint.
2 Concentration that is lethal to 50% of the test organisms (sac-fry survival test only).
3 Invalid due to poor control growth. See main text for discussion and steps to rectify this issue.
4 Invalid due to unacceptably high incubation temperature.
5 Values in parentheses represent 95% confidence intervals.

Chlorella sp.

Two reference toxicity tests were completed for *Chlorella* sp. However, both tests were invalid. At least one of these tests was affected by low control growth rates, which also compromised the tests run in 2004–05. The issue of low control growth is discussed in greater detail below.

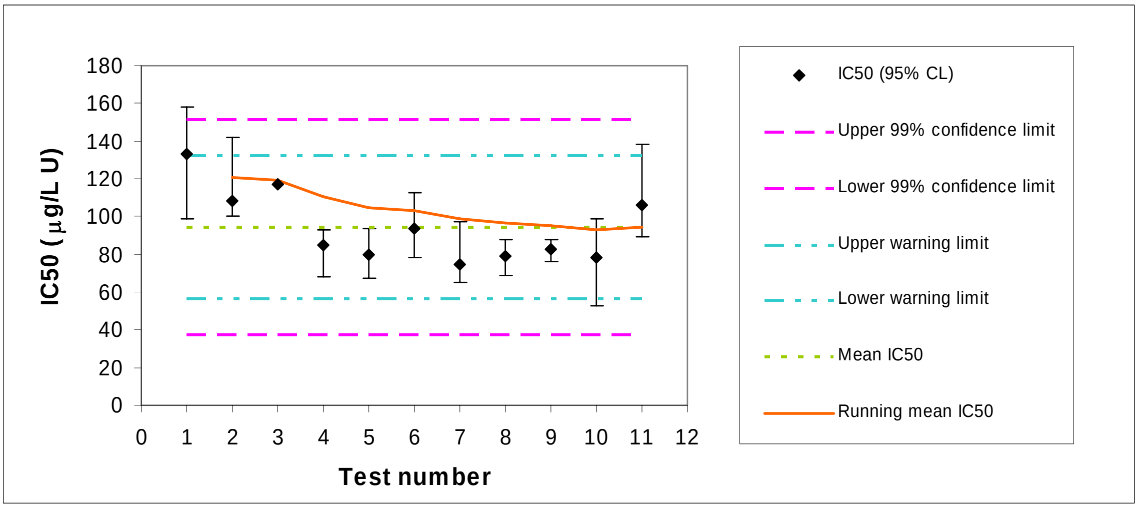
H. viridissima

Four reference toxicity tests were completed using *H. viridissima*. One of the tests was invalid due to low control growth (refer to discussion below on low control growth). The IC_{50} values of the remaining tests did not exceed existing warning limits (ie. two standard deviations above or below the running mean; see Figure 1A) indicating a consistent experimental technique and response of the test species to uranium over time. The current running mean IC_{50} is 93 µg/L (see Figure 1A).

L. aequinoctialis

Three reference tests were completed using *L. aequinoctialis*. In the first test, the concentration range was insufficient to capture a 50% response, and the result could not be used in a control chart. The second test was invalid due to low control growth, while the third provided a valid result. It is worth noting that the two tests that met the acceptability criterion for control growth were on, or only slightly above, the minimum acceptable control growth criterion (ie an increase of at least 48 fronds over 96 h). The issue of poor control growth is discussed in greater detail below. As tests using this species only commenced in 2005–2006, there are currently insufficient data to generate a control chart (a minimum of five valid test results are required construct a control chart; Environment Canada 1990).

A



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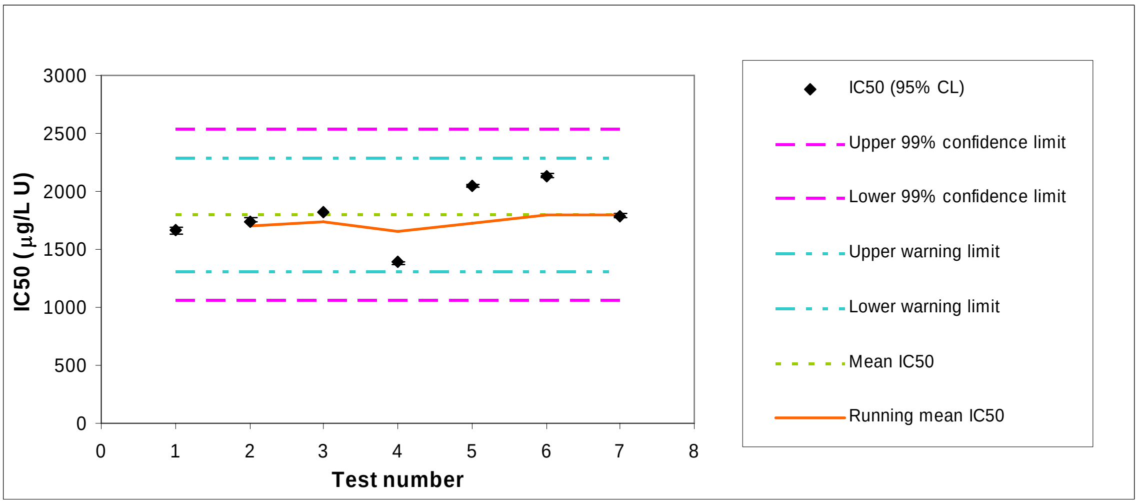

Figure 1 Reference toxicant control charts, based on uranium toxicity (IC/LC_{50}) data for (A) *Hydra viridissima* and (B) *Mogurnda mogurnda*.

M. mogurnda

One reference toxicity test was completed using *M. mogurnda*. The current control chart, based largely on historical toxicity data from Riethmuller et al (2000), is shown in Figure 1B. The current running mean IC_{50} is approximately 1800 µg/L (see Figure 1B).

Low control growth

Chlorella sp.

Low control growth was identified as a problem in algal reference toxicity tests in both 2004–05 and 2005–06. Notably, however, acceptable control growth has been consistently measured for toxicity tests using natural Magela Creek water. Analyses of the key components of the stock solutions used to prepare the synthetic softwater (SSW) testing medium used for reference toxicity tests demonstrated that all but the $NaHCO_3$ stock were within 10% of their nominal concentrations. Consequently, the $NaHCO_3$ stock was re-prepared in October 2005 to ascertain if this was the cause of the original problem. However, similarly low control growth was observed in tests undertaken after this stock was renewed. Measures taken to return the algal culture to an axenic condition in August 2005 also resulted in little or no improvement in algal growth in synthetic softwater.

Laboratory staff had noticed, in recent tests, that after adjustment to the prescribed pH of 6.00 ± 0.15 units the pH of the SSW continued to drop whilst in storage. This is thought to be a combined result of not allowing the pH to equilibrate fully during pH adjustment and the lack of buffering capacity in such a low ionic strength medium. Interestingly, in the three experiments where the control growth was lowest, the pH at the start of the test was at or below pH 5.85, even in the presence of a biological buffer (1.3 g/L (1 mM) N-2-Hydroxyethylpiperazine-N'-2-ethanesulphonic acid (HEPES)). Franklin et al (1998) demonstrated that *Chlorella* sp is highly sensitive to changes in pH, and that the optimal pH for *Chlorella* sp. growth is 6.5. This is supported by the valid algal reference tests run in 2004–05, which had a starting pH of 5.95 or above. The fact that this optimal pH had not been previously noted in the protocol for testing with *Chlorella* sp. will be rectified.

Future reference toxicity tests using *Chlorella* sp. will, therefore, be conducted at a starting pH of 6.5 ± 0.15. In addition, the SSW pH adjustment procedure will be amended to allow greater equilibration time, while an investigation into the use of the HEPES buffer to maintain pH at 6.5 will also be undertaken (ie. although satisfactory in Magela Creek water, 1 mM HEPES may be insufficient in SSW).

H. viridissima

With only one out of ten *H. viridissima* reference toxicity tests undertaken in this program being invalid due to low control growth, and the most recent test demonstrating excellent growth, it is unlikely that there is a problem with the experimental techniques used or the health of the laboratory culture. Without a long-term data set for comparison, it is difficult to say whether low control growth in SSW may occur periodically for this species. Consequently, the results of future *H. viridissima* reference toxicity tests will be closely monitored to gauge whether low control growth occurs with unacceptable frequency.

L. aequinoctialis

While sufficient control growth was observed for two of the three tests undertaken using *L. aequinoctialis,* the growth rate of one of these only just met the criterion, and the other was only marginally higher. The nutrient concentrations used in toxicity testing with this species were optimised in experiments using natural Magela Creek water to identify the minimum additions of NO_3 and PO_4 that would sustain good plant growth while minimising the likelihood of unwanted interactions with test toxicants. Therefore, a nutrient trial similar to that undertaken in Magela Creek water will be undertaken in SSW to determine the most appropriate regime for this artificial medium. In addition, although the SSW is modelled on the inorganic constituents of Magela Creek water and is therefore the most relevant standard medium available, it is likely to be lacking organic compounds such as tannins and humic acids that are present in natural waters and which may be required for optimal plant growth. This may also explain why good plant growth is obtained with this nutrient regime in natural Magela Creek water, but is substantially lower in SSW.

Steps for completion

The reference toxicity testing program at ***eriss*** is ongoing and further experiments using the four species already tested are scheduled for 2006–07. In addition, reference toxicity testing using the water flea *Moinodaphnia macleayi* will be initiated using the 48-h immobilisation test protocol. It is expected that approximately four reference toxicity tests for each species will be completed in 2006–07.

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Toxicity of magnesium in Magela Creek water to local freshwater species

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

Background

Magnesium sulfate ($MgSO_4$) is the dominant surface water contaminant associated with Ranger uranium mine (Ranger). Although this salt is generally considered to be of low toxicity, aquatic surveys around Ranger in the mid-1990s showed correlations between changes in macroinvertebrate community structure and increasing $MgSO_4$. This finding prompted a full ecotoxicological investigation, including: identification of the dominant toxic ion; assessment of Mg toxicity in extremely soft local creek water, in both the laboratory and in the field; and the influence of calcium (Ca) on Mg toxicity in the laboratory. Much of this research has been described in previous Supervising Scientist Annual Reports (see Supervising Scientist 2002; 2003) and more recently, van Dam et al (2006).

During 2005–06, the aim was to complete the remaining experiments to quantify the effects of amelioration by Ca and to analyse all data to derive a site-specific trigger value for Mg.

Progress

Ten experiments were done in 2005–06. Of these, two experiments (both for *Mogurnda mogurnda*) were invalid (the first due to additions of incorrect volumes of $CaSO_4$ stock solutions, and the second due to excessive control mortality). The remaining tests are briefly summarised below.

(i) Toxicity of $MgCl_2$

As part of the component to determine the relative toxicity of the cation Mg^{2+} and the anion SO_4^{2-}, the toxicity of $MgCl_2$ to the snail *Amerianna cumingi* was assessed (Figure 1). The IC_{50}[1] of Mg when added as $MgCl_2$ (based on linear interpolation) was 20 mg/L (lower/upper 95% CLs: 17/22 mg/L). This was very comparable to the average IC_{50} for Mg of 18 mg/L when added as $MgSO_4$. Considering that sulfate, when added as $NaSO_4$, had no effect on *A. cumingi* up to a concentration of 340 mg/L, these results confirmed that it is the cation Mg^{2+} that is toxic to the snail species. This same conclusion had been reached previously for *Hydra viridissima* and *Lemna aequinoctialis*.

[1] Concentration that results in a 50% inhibition of the test endpoint (eg *Amerianna cumingi* egg production)

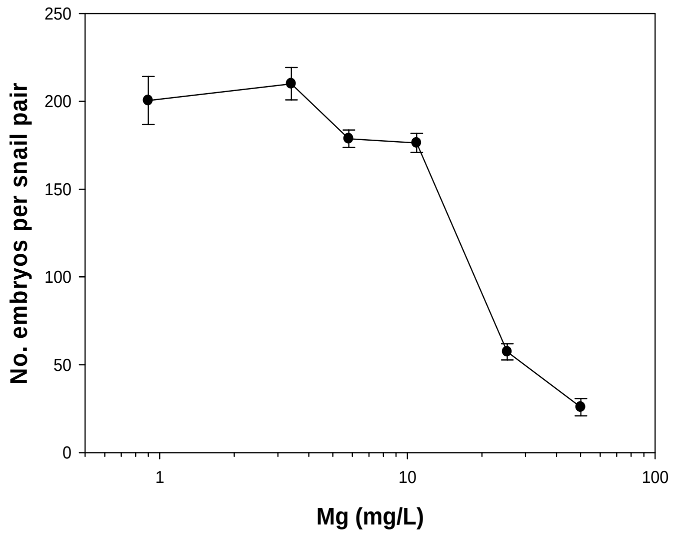

Figure 1 Effect of Mg, when added as $MgCl_2$, on embryo production of the aquatic snail, *Amerianna cumingi*. Data points represent the mean (± SEM) of three replicates.

(ii) Toxicity of Mg at the 'safe' Mg:Ca ratio of 9:1

The remaining tests were focused on assessing the toxicity of Mg at the 'safe' Mg:Ca ratio of 9:1, which had been derived previously based on experiments using *H. viridissima*, *L. aequinoctialis* and *A. cumingi* (as described in van Dam et al 2006), to *Moinodaphnia macleayi* (3 tests), *Chlorella* sp. (3 tests) and *L. aequinoctialis* (1 test). Only the data for *M. macleayi* are described here. Figure 2 displays the toxicity of Mg to *M. macleayi*, with data from four separate experiments (three from 2005–06 and one from a previous experiment) being pooled (analysis of covariance indicated that the concentration–response relationships for each of the experiments were not significantly different; F = 0.064, df = 3, P = 0.978). Based on the fitted regression model (3 Parameter Sigmoid, r^2 = 0.92, n=29, P<0.0001), the IC_{15} and IC_{50} (upper-lower 95% CLs) were 45 (20–65) and 138 (115–160) mg/L, respectively. Interestingly, the IC_{50} value is about two-fold higher than the IC_{50} of 64 mg/L for Mg at background Ca concentrations, indicating that Ca has a relatively small ameliorative effect on the toxicity of Mg to *M. macleayi*. In contrast, results from experiments using *L. aequinoctialis* and *M. mogurnda*, indicated there was a 100-fold reduction in the toxicity of Mg (based on IC/LC_{50}s) when the Mg:Ca ratio was maintained at 9:1. The results for the other species were being analysed, compared and interpreted at the time of preparation of this summary.

Steps for completion

Two fish (*Mogurnda mogurnda*) toxicity tests were required to complete the testwork for this project. These tests were completed in July and August 2006.

All toxicity data associated with the Mg toxicity project, which comprised well over 50 experiments, have been quality checked and are being re-analysed. Once this analysis has been completed, revised trigger values for Mg at Mg:Ca ratios >9:1 and <9:1 will be calculated and communicated to stakeholders via the Ranger Minesite Technical Committee. The work is currently in the process of being written up for publication.

A poster summarising the final Mg toxicity project results has recently been presented by Mel Houston at the Interact 2006 conference in Perth.

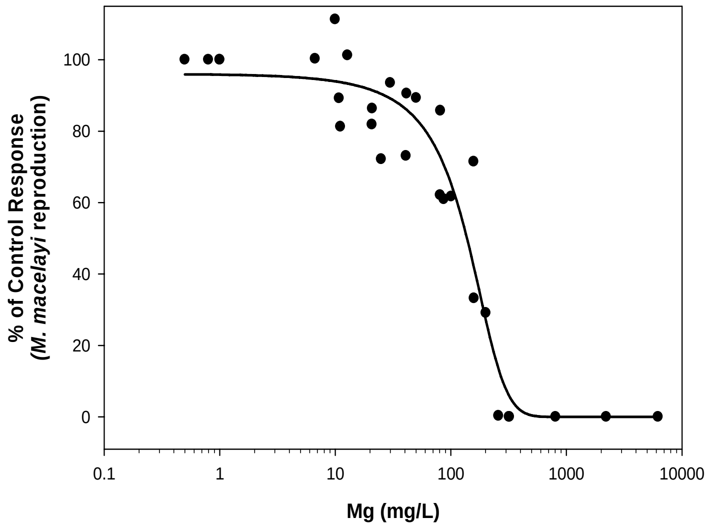

Figure 2 Effect of Mg on reproduction of the cladoceran *M. macleayi*, when the Mg:Ca (mass) ratio is kept constant at 9:1. Plot based on pooled data from 4 separate experiments, with each data point representing the mean of three replicates (individual error bars not shown). The fitted curve represents a 3 parameter sigmoid model ($r^2 = 0.92$, $n = 29$, $P < 0.0001$).

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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

Background

Several factors, including a number of above average wet seasons, the need to keep the base of Pit#3 dry for mining and the removal of Djalkmara Billabong (a wetland used to hold and polish run-off from the minesite prior to controlled release into Magela Creek), prompted Energy Resources of Australia (ERA) to consider alternative methods for the reduction of onsite waters at Ranger Mine. Proposals for the treatment and discharge of two grades of water, namely 'pond' and the more highly contaminated 'process' water, were discussed with stakeholders throughout 2004–2005. Whilst a pilot plant demonstrated (through water chemistry analysis) that the quality of the treated pond water would likely be suitable for direct release, it was agreed that for additional assurance, SSD would undertake ecotoxicological testing on the reverse osmosis (RO) permeate from the newly commissioned plant prior to release into the Corridor Creek wetlands.

ERA completed commissioning the water treatment plant for pond water in December 2005 and provided SSD with assurance that the permeate being produced at the time was representative of future outputs, and thus, ready for ecotoxicological testing. SSD staff from the Jabiru Field Station sampled the permeate on 12 December 2005 and ERA staff delivered the sealed sample bottles to the ***eriss*** Darwin laboratories for testing the same day.

Methods

Five local organisms, a unicellular alga (*Chlorella* sp.), macrophyte (duckweed; *Lemna aequinoctialis*), cnidarian (*Hydra viridissima*), crustacean (water flea; *Moinodaphnia macleayi*) and a fish species (*Mogurnda mogurnda*), were exposed to concentrations of 30, 44, 67 and 100% treated pond water permeate and a Magela Creek water control. All dilutions of the permeate were undertaken using freshly collected Magela Creek water.

Results and discussion

Exposure to the treated pond water permeate had no effect on the growth of the two plant species and the hydra, nor on the survival of the fish. However, the crustacean (*M. macleayi*) was shown to produce significantly less offspring when exposed to the two highest concentrations of permeate (67 and 100%) (Figure 1). Thus, the Lowest-Observed-Effect-Concentration (LOEC) and No-Observed-Effect-Concentration (NOEC) were 67% and 44% permeate, respectively.

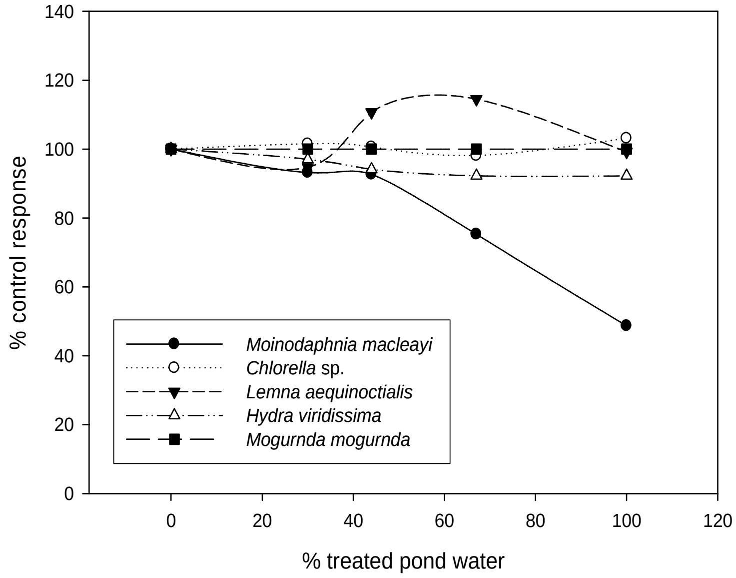

Figure 1 Response of five local species to treated pond water permeate. Each data point represents the mean of three (10 in the case of *M. macleayi*) replicates (individual error bars not shown). Note that the elevated response observed in the *L. aequinoctialis* test was found to be not significantly different from the control ($P>0.05$).

A full ICP-MS scan of metals in the treated pond water found that all metals, other than uranium, were below or around the limits of detection and hence, unlikely to be contributing to the observed toxicity. The 4 µg/L of uranium in the treated pond water was less than what has been shown to cause toxicity to *M. macleayi* in Magela Creek water. However, natural Magela Creek water typically contains 2–8 mg/L dissolved organic carbon (DOC), which has been shown to ameliorate uranium toxicity. A notable characteristic of the permeate was that DOC was below detection limit (<1 mg/L). The absence of DOC in the permeate may, therefore, result in greater uranium toxicity to *M. macleayi*.

An alternative hypothesis is that the absence of DOC and possibly other essential ions from the permeate may result in a reduction in reproduction and survival of *M. macleayi* as a result of nutrient/ion deficiencies. To test these hypotheses, we are planning to test the toxicity of uranium in a 'synthetic' Magela Creek water that simulates the inorganic composition of the water but contains no DOC, and to compare the results to experiments conducted in natural Magela Creek water.

Regardless of the reason for the effect on *M. macleayi*, the treated pond water could be considered only moderately 'toxic', and any potential for an effect on downstream biota could be avoided by diluting the permeate as it is released off-site. It should be noted that the preferred distribution-fitting method for deriving acceptable concentrations/dilutions could not be used in this case because only one of the five species tested responded to the permeate.

An acceptable dilution factor of 1 part permeate to 23 parts Magela Creek water (ie. 4.4% permeate) was calculated using the 'safety factor' approach outlined in the ANZECC/ARMCANZ Water Quality Guidelines (ie. the NOEC for *M. macleayi*, of 44% permeate, was divided by a default safety factor of 10). This dilution factor was adopted as one of the primary criteria for the release of treated pond water to Magela Creek.

Steps for completion

The toxicity of uranium to *M. macleayi* in the absence of DOC, will be assessed. Toxicity testing of treated process water is expected to take place once the plant has been commissioned for this water grade in 2007. The results from the testing of the treated process water may indicate the need to focus more closely on ammonia toxicity. Ammonia was inferred to be the primary toxicant in RO permeate water produced from the process water treatment pilot plant in 2001 (Camilleri et al 2002).

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Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer

Introduction

Radon is a radioactive noble gas that exhales from the earth's surface. It is an element in the uranium decay chain, and thus the radon exhalation flux depends, amongst many other factors, on the concentrations of uranium and radium in the soil. Radon decays with a half life of 3.8 days and the radon decay products (RDP) – ^{218}Po, ^{214}Pb and ^{214}Bi – can be retained in the lungs after inhalation. Their subsequent decay can deliver a significant dose to the soft tissue of the respiratory system. The greatest fraction of natural exposure of humans to radiation originates from the inhalation of radon decay products (Porstendörfer 1994).

Similarly, radioactive elements trapped in or on dust (long lived alpha activity or LLAA) can deliver a radiation dose to the respiratory system once inhaled and trapped in the lungs. The Supervising Scientist Division conducts an atmospheric monitoring program at Jabiru, Jabiru East and close to Mudginberri Billabong (Map 1), measuring RDP and LLAA. Research and development projects are conducted into the sources, behaviour and atmospheric pathways of radon and dust originating from Ranger mine.

The results of the atmospheric monitoring program are periodically compared with results from ERA's atmospheric radiological monitoring program. In addition radon gas is measured continuously on a half hourly basis at the Mudginberri Four Gates Rd Radon Station 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 provides a statistical summary of RDP data from Mudginberri, Jabiru and Jabiru East measured by ***eriss*** from early 2002 to December 2005. Median RDP concentrations at Jabiru and Mudginberri are 0.037 and 0.040 µJ/m^3, respectively. Concentrations at Jabiru East are generally higher (median of 0.064 µJ/m^3) and show more variation. The 75th percentile concentration at Jabiru East is twice as high as at Jabiru and Mudginberri.

Energy Resources of Australia estimates the mine-derived RDP using a wind correlation model developed by ***eriss***, and subsequently calculates exposure via the radon pathway. Table 1 shows the average RDP concentration and annual total doses received from the inhalation of RDP at Jabiru provided by ERA (ERA 2006) and calculated from ***eriss*** data (in brackets). This dose assumes an occupancy of 8760 hrs (1 year) and a dose conversion factor for the public of 0.0011 milli Sievert (mSv) per µJ/hr/m^3. ERA also reports an average mine-derived RDP concentration of 0.03 µJ/m^3 for the 1125 hours that the wind was blowing from the mine in 2005, which results in a mine-related dose calculated for 2005 of 0.037 mSv in Jabiru, less than 4% of the public dose limit of 1 mSv/y.

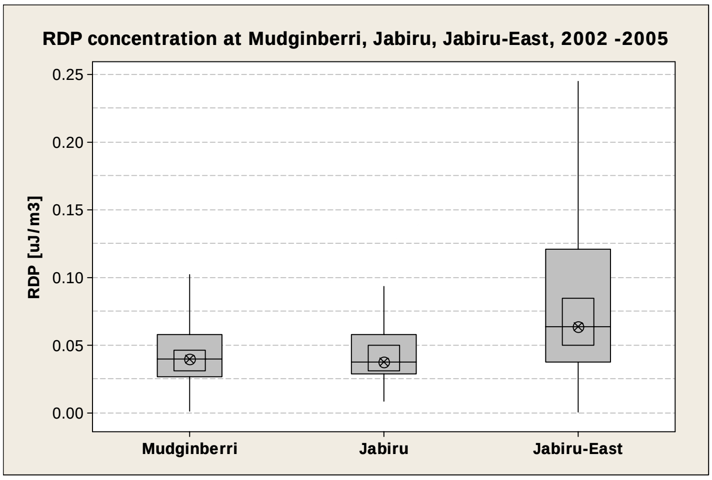

Figure 1 Statistical analysis of 4 years of radon decay product data at Mudginberri Four Gates Road radon station, Jabiru and Jabiru East

Table 1 Radon decay product mean concentrations at Jabiru and Jabiru East, and total and mine derived annual doses received at Jabiru in 2003–2005

		2003	2004	2005
RDP mean concentration [μJ/m³]	Jabiru East	0.075 (0.101)	0.103 (0.095)	0.097 (0.097)
	Jabiru	0.065 (0.043)	0.079 (0.063)	0.088 (0.052)
Total annual dose [mSv] Jabiru		0.63 (0.41)	0.76 (0.61)	0.85 (0.50)
Mine derived dose [mSv] at Jabiru		0.011	0.014	0.037

Dust pathway

The dust inhalation pathway has been quantified using an innovative approach (Bollhöfer et al 2006). Stable lead isotope ratios have been used to discriminate between natural dust and dust originating from the mining and milling of uranium at Ranger. To illustrate the differences in isotopic ratios, Table 2 shows the lead isotope ratios measured in various primary sources of lead.

Table 2 Typical lead isotope ratios in various primary lead sources

	Primordial	**Lead ores** (*Doe 1970*)		**Australian uranium ores**	
Ratio	(*Tatsumoto et al 1973*)	Broken Hill	Mississippi valley	Koongarra orebody 2 (*Dickson et al 1985*)	Ranger (*Gulson et al 1992*)
$^{206}Pb/^{207}Pb$	0.90	1.04	1.33–1.39	7.36	9.69
$^{208}Pb/^{207}Pb$	2.87	2.32		0.724	0.0049
$^{206}Pb/^{204}Pb$	9.31	16.01–16.12		394.1	10602.5

Using the temporal and spatial variability (Figure 2) of lead isotope ratios measured in the vicinity of Ranger, the average annual contribution from Ranger uranium mine to the long-lived alpha activity concentrations at Jabiru East has been estimated to be almost 40%. The contribution at Jabiru is much lower. Assuming that a person lives at Jabiru and works 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.

Although proven to be trivial for the public in the Alligator Rivers Region, dispersion of radionuclides in dust may be more significant in drier, less regulated environments elsewhere in Australia. The large differences in lead isotope ratios of natural soils and uranium mineralised material and the use of ICPMS techniques for isotope analyses, make this a quick and reliable method for apportioning doses between sources, which may be attractive for mining industry managers and regulators.

Summary

Monitoring of the radon and dust exposure pathways has shown that at present the only significant contribution to radiological exposure of the public via inhalation, is the inhalation of mine-derived radon decay products. The current contribution is much less than the public dose limit of 1 mSv per year and is of no concern. Ongoing atmospheric monitoring will continue to provide re-assurance to the public that inhalation of mine derived radionuclides remains low.

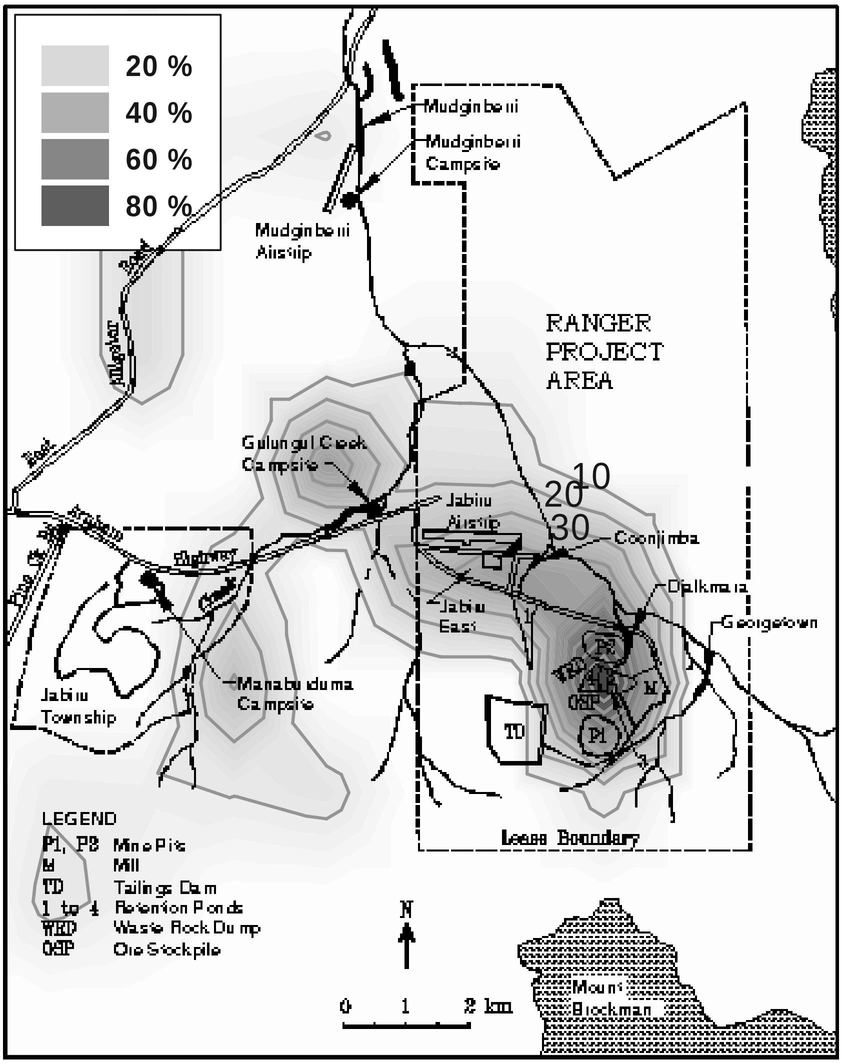

Figure 2 Percentage contribution of mine origin airborne dust in the vicinity of Ranger (from Bollhöfer et al 2006)

Steps for completion

This project is an ongoing project. The routine monitoring of dust and radon progeny will continue at the three monitoring sites of Jabiru, Mudginberri Four Gates Road Radon Station and Jabiru East. A simultaneous sample collection regime, ideally via high volume air samplers at the source and receptor locations, with measurements of stable lead isotope ratios, uranium, thorium and long-lived alpha activity concentrations, provides an ideal tool for dose estimation and unambiguous source apportionment for the dust inhalation pathway. A set of on-site and environmental monitoring dust filters covering a complete annual cycle should be collected to test the method covering a complete annual cycle. Samples will be analysed by ICPMS to calculate the variability of the radiation dose from inhalation of dust originating from Ranger throughout the year. This validated method may be ideal for other sites, where the dust inhalation pathway is more significant.

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Monitoring of groundwater at Ranger

B Ryan & A Bollhöfer

Introduction

The aim of the Ranger groundwater program is to investigate the dispersion of contaminants through the groundwater pathway, from both radiological monitoring and hydrogeological modelling perspectives. Shallow and deep aquifers, and sources of contaminants need to be characterised, both geochemically and hydrogeologically. Once the aquifers have been characterised, in terms of distribution and transport of contaminants and aquifer interactions, models can be developed that predict future groundwater quality, solute transport and its ultimate impact on surface water. The monitoring program will need to be continued for an extended period following the rehabilitation of the site to assess the success of the rehabilitation and the integrity of the pits as tailings repositories.

Radionuclide activity and metal concentrations are measured in groundwater samples collected annually at the end of the dry season from bores around the site. Heavy metal concentrations are determined via ICPMS-OES, and radionuclides of interest are radiochemically separated from the sample, and measured via alpha spectrometry.

As thorium and lead are particle reactive and readily adsorbed and removed from solution, it is not expected that either of these elements will migrate significant distances through the groundwater aquifer unless the water is acidic. This is also assumed to be the case for actinium. Consequently, the reduced list for U-series radionuclides potentially contaminating the groundwater is: ^{238}U, ^{234}U and ^{226}Ra. These radioisotopes are the focus of the groundwater monitoring program.

The ^{234}U/^{238}U activity ratio can potentially be used to identify mine-related sources, as they 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).

Progress to date

The focus for 2005–06 was a compilation and evaluation of groundwater data for Nabarlek (Supervising Scientist Internal Report, Ryan & Bollhöfer, 2006), summarised under KKN 4.2.1 The lessons learned from this work will aid in analysing, interpreting and contribute to modelling the Ranger groundwater data.

Ranger bore water samples were collected by the Northern Territory Department of Primary Industry, Fishery and Mining for ***eriss*** in 2004 and 2005, with aliquots being prepared in the ***eriss*** laboratories for radioisotope analysis. ICPMS-OES analysis of all archived groundwater samples was completed by Charles Darwin University in 2005 for barium, calcium, iron, sulphur, sodium, potassium, magnesium, manganese, vanadium, uranium and strontium. Uranium isotope and radium analyses via alpha spectrometry of archived samples have been completed.

Steps for completion

The metal and radionuclide activity concentration results are now being collated and analysed, and interpretation started of the long-term groundwater data for uranium isotopes and radium. Particular attention is being placed on assessing groundwater movement and identifying any sources of contamination.

ERA has significantly reduced its groundwater monitoring program in recent years and now only has four statutory groundwater monitoring sites. A further 22 bores are sampled in the ERA operational monitoring program at frequencies of quarterly and monthly. ***eriss*** has now acquired sampling equipment to target specific bores. With tailings being stored in Pit #1 and eventually in Pit #3, there will need to be an increase in monitoring bores in the vicinity of the two pits without neglecting important bores in the vicinity of the tailings dam.

It is planned to undertake initial investigations in late October 2006 to profile alluvial water in the Magela Creek channel and near surface groundwater in the Magela Land Application Area, in collaboration with staff from the Northern Territory Department of Primary Industry, Fisheries and Mines. An EM31 geophysical instrument will be used for this purpose. Electrical conductivity measurements will be made over the area and, if feasible, a map produced showing the conductivity down to a particular depth. The data may be used to locate anomalous conductivity zones and investigate potential source inputs of solutes from the Magela and Djalkmara land application areas that may be entering the alluvial flow system as a density plume.

Acknowledgments

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

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Introduction to SSD's stream monitoring program for Ranger, 2005–06

C Humphrey & D Jones

The SSD operates an integrated chemical, 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, this is the first 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 in indicators

- *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);
  - Continuous monitoring: use of multi-probe loggers for continuous measurement of pH, EC, turbidity, temperature and dissolved oxygen in Magela Creek, and EC and turbidity in Gulungul Creek;
- *Toxicity (including creekside) monitoring* of reproduction in freshwater snails and survival of fish fry (four-day tests conducted at fortnightly intervals);
- *Bioaccumulation* – concentrations of chemicals (including radionuclides) in the tissues of freshwater mussels and fish in Mudginberri Billabong to detect far-field effects including those arising from any potential accumulation of mine-derived contaminants in sediments (mussels sampled every late-dry season, fish sampled biannually in the late dry season).

(ii) Assessment of changes in biodiversity

- *Benthic macroinvertebrate communities* at stream sites (sampled at the end of each wet season); and
- *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.deh.gov.au/ssd/monitoring/magela-bio.html).

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

M Iles

Routine weekly sampling program in Magela Creek

The first water chemistry samples for the SSD surface water monitoring program for the 2005–06 wet season were collected from Magela Creek on 6 December 2005, one day after flow was first observed at the downstream statutory compliance point (‘Magela d/s’, Map 2). Weekly sampling was conducted throughout the wet season, and continued until the creek ceased to flow on 28 August 2006, with the following exceptions: (i) following an accidental irrigation of the Magela Land Application Area with pond water on 21–22 January 2006, additional sampling of Magela Creek was undertaken on 23 January 2006; and (ii) in the last week of April 2006, sampling did not occur after Tropical Cyclone Monica passed over Jabiru on 25 April 2006 because sites were inaccessible. SSD collected its last sample on 24 August 2006 shortly before Magela Creek ceased flowing.

The values of water quality indicators for the wet season, including the period immediately following the irrigation incident, have been within limits/guidelines (Iles 2004) set by the Supervising Scientist for the protection of the aquatic environment and are within the range seen in previous years.

The summary statistics for the upstream and downstream key water quality data from both the SSD and ERA water quality monitoring programs are shown in Table 1. The time series data for uranium concentrations from both the SSD and ERA routine and investigative monitoring (following the irrigation incident) are shown in Figure 1. There is good agreement between the datasets of both organisations.

Table 1 Summary of Magela Creek 2005–06 wet season# water quality up and downstream of Ranger

Parameter	Guideline or Limit*	Organisation	Median		Range	
			Upstream	Downstream	Upstream	Downstream
pH	5.0 – 6.9	SSD	6.4	6.4	5.6 – 6.8	5.9 – 6.8
		ERA	6.3	6.4	5.5 – 6.7	5.8 – 6.7
EC (µS/cm)	43	SSD	14	17	7.9 – 20	8.5 – 23
		ERA	12	15	4.8 – 20	6.9 – 23
Turbidity (NTU)	26	SSD	2.0	2.2	0.9 – 14	0.8 – 18
		ERA	2.	2.	1 – 11	1 – 14
Sulfate‡ (mg/L)	Limited by EC	SSD	0.2	0.7	0.1 – 0.4	0.3 – 3.4
		ERA	0.2	0.8	0.1 – 0.6	0.3 – 3.8
Magnesium‡ (mg/L)	Limited by EC	SSD	0.6	0.9	0.2 – 1.1	0.3 – 1.4
		ERA	0.5	0.8	0.1 – 0.9	0.2 – 1.2
Manganese‡ (µg/L)	26	SSD	4.4	4.9	2.2 – 13	2.1 – 16
		ERA	3.9	4.1	1.9 – 10	3.2 – 16
Uranium‡ (µg/L)	6	SSD	0.014	0.048	0.003 – 0.044	0.014 – 0.153
		ERA	0.018	0.064	0.006 – 0.060	0.014 – 0.145

ERA data taken from the ERA Weekly Water Quality Report 18 August 2006; ‡ dissolved (<0.45 µm); # SSD results from the last sampling event, 24 August, outstanding at time of report writing; * A compliance limit applies to uranium, management guidelines apply to all other parameters shown.

Uranium, manganese, magnesium and sulfate median values from both datasets were higher downstream of the mine but the concentrations were very low and not of environmental concern. Uranium concentrations remained well below (<3% of) the limit (Figure 1). The low values are indicative of the pattern of improved water quality seen in the past four wet seasons and demonstrated by the uranium results for the last five years (Figure 2).

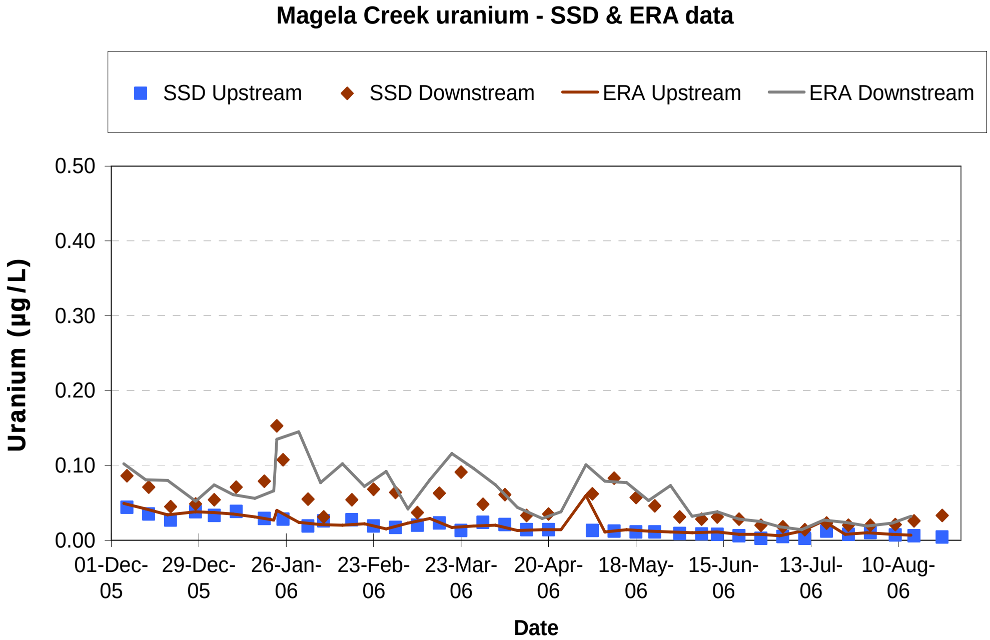

Figure 1 Uranium concentrations measured in Magela Creek by SSD and ERA during the 2005–06 wet season

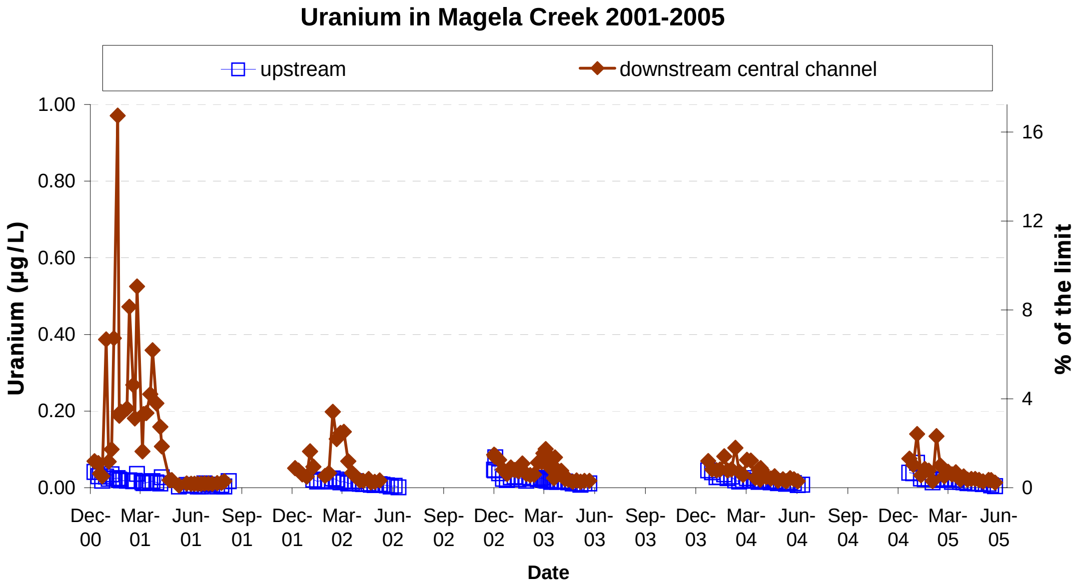

Figure 2 Uranium concentrations in Magela Creek since the 2000–01 wet season (SSD data)

Electrical conductivity (EC), whose guideline value provides a management tool for the control of magnesium and sulfate concentrations, was also slightly higher downstream but compared to the guideline value the difference was small. The manganese, pH, and turbidity medians are similar at both sites for the SSD and ERA datasets.

The water quality objectives set to protect the aquatic ecosystems downstream of the mine were achieved during the 2005–06 wet season. Available biological monitoring data (described later in this report) also indicate that the environment remained protected throughout the season.

Routine weekly sampling program in Gulungul Creek

The first water chemistry samples were collected from upstream and downstream sites in Gulungul Creek (Map 2) on 29 November 2005, during the first week after flow commenced in the creek. Weekly sampling was conducted throughout the wet season, and continued while the creek was flowing, except for the last week of April 2006 when sites became inaccessible after Tropical Cyclone Monica passed over Jabiru (on 25 April 2006). SSD collected its last sample on 15 August 2006 shortly before Gulungul Creek ceased to flow.

The upstream and downstream water quality data from both the SSD and ERA programs are summarised in Table 2, with uranium time series concentration data shown in Figure 3. There is good agreement between the datasets of both organisations and the overall water quality and seasonal trends for the 2005–06 wet season are comparable to those seen in previous years (Figure 4).

Although median values for most of the key variables were slightly higher downstream of the mine (Table 2), the concentrations were very low and not of environmental concern.

Table 2 Summary of Gulungul Creek 2005–06 wet season water quality upstream and downstream of Ranger

Parameter	Company	Median		Range	
		Upstream	Downstream	Upstream	Downstream
pH	SSD	6.3	6.5	5.4 – 6.7	5.7 – 6.7
	ERA	6.3	6.4	5.1 – 6.7	5.4 – 6.6
EC (µS/cm)	SSD	16	19	10 – 21	11 – 29
	ERA	13	15	8.7 – 24	8.4 – 26
Turbidity (NTU)	SSD	1.0	1.4	0.4 – 5.4	0.7 – 7.7
	ERA	1.	1.	<1 – 8.	<1 – 5.
Sulfate‡ (mg/L)	SSD	0.2	0.4	0.1 – 0.7	0.1 – 2.3
	ERA	0.2	0.5	0.1 – 1.2	0.1 – 1.8
Magnesium‡ (mg/L)	SSD	0.9	0.9	0.5 – 1.8	0.5 – 1.8
	ERA	0.8	0.8	0.3 – 1.6	0.4 – 1.3
Manganese‡ (µg/L)	SSD	2.1	3.6	1.2 – 8.5	2.0 – 18
	ERA	2.0	3.2	1.2 – 11	1.8 – 18
Uranium‡ * (µg/L)	SSD	0.054	0.095	0.030 – 0.169	0.058 – 0.393
	ERA	0.060	0.102	0.032 – 1.64	0.053 – 1.05

‡ dissolved (<0.45 µm), * limit = 6 µg/L

ERA measured elevated uranium on the first day of flow (Figure 3) when it sampled within hours of flow first occurring, resulting in considerably higher range maximum than SSD (Table 2). Uranium concentrations were below the limit and the concentration at the upstream site was higher than that at the downstream site. In mid-January 2006, SSD measured a higher than usual uranium concentration of 0.393 µg/L (less than 7% of the 6 µg/L limit determined for Magela Creek). None of these excursions is considered to be environmentally significant: Values this high experienced previously and for longer periods did not impact on the biodiversity (as assessed by the biological monitoring data at that time) of Gulungul Creek.

Available biological monitoring data (described later in this section) confirm that the environment remained protected throughout the 2005–06 season.

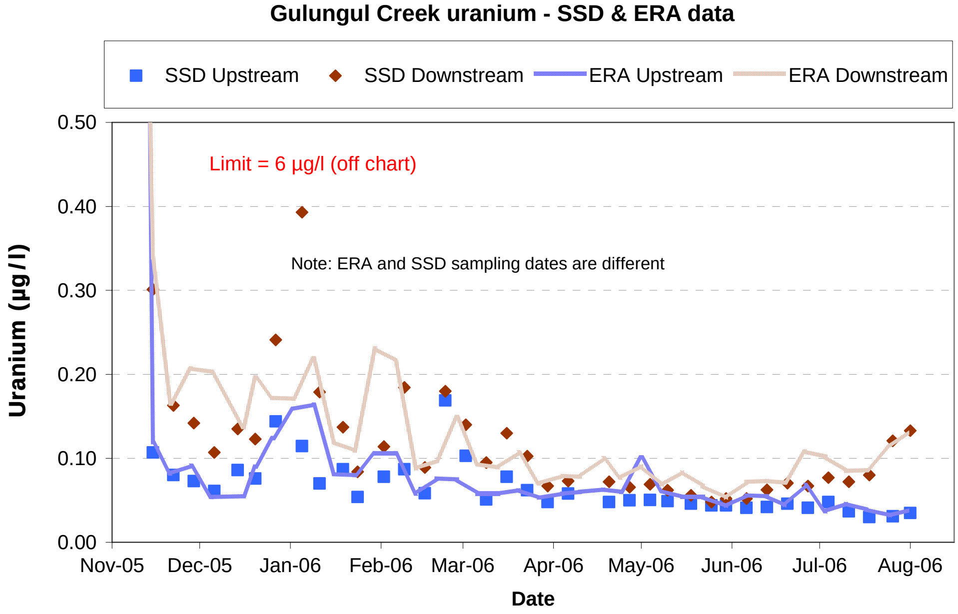

Figure 3 Uranium concentrations measured in Gulungul Creek by SSD and ERA during the 2005–06 wet season

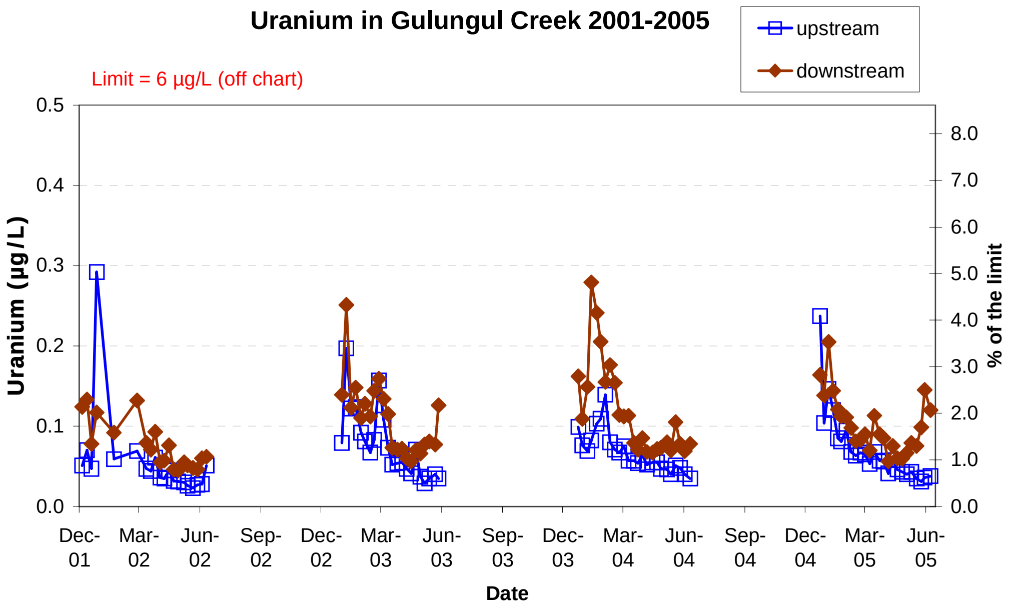

Figure 4 Uranium concentrations in Gulungul Creek between 2000 and 2005 (SSD data)

Reference

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Continuous monitoring of water quality

K Turner

For environmental protection and improved wastewater management associated with the Ranger mine site, there is a recognised requirement to track and quantify the movement of solutes originating from point and diffuse sources through the receiving Magela and Gulungul Creek systems. Continuous in situ measurement of key water quality variables using dataloggers placed at strategic locations on and off-site can meet these needs, particularly when linked to localised and catchment-wide rainfall and stream discharge data. Continuous monitoring will complement SSD's routine water quality monitoring program and enable detection of 'events' and exceedances that may be undetected by the weekly grab sampling regime of the routine program.

In addition to contributing to the current operational surveillance role, the implementation of continuous monitoring will enable the technology and data interpretation methods to be thoroughly tested for deployment during and beyond rehabilitation of the Ranger mine site.

Three loggers were deployed in Magela Creek at the start of the 2005–06 wet season – one located approximately 0.5 km downstream of the Magela Creek (upstream) control site (but still upstream of the mine surface-water influence) and another two located approximately 0.5 km downstream of the Magela Creek downstream compliance point (G8210009), on either side of the western-most channel (Map 2). The loggers were mounted on the pontoons housing the water intake lines for the creekside monitoring program. In situ water quality data (including electrical conductivity [EC], pH and turbidity) were collected at 15–20 minute intervals. Corresponding streamflow data were collected from upstream and downstream gauging stations on Magela Creek (by ERA and NRETA respectively).

Continuous EC, pH and flow data are measured by ERA at the RP1 discharge weir and in Corridor Creek at monitoring location GC2. Integration of these data with the ***eriss*** data from Magela Creek will enable a comprehensive real-time understanding of the link between site runoff and downstream water quality in Magela Creek.

Quality-control, spot check measurements were made using a calibrated portable field meter at the upstream and downstream Magela Creek sites and were very similar to the continuously measured values on both sampling occasions, indicating good calibration and performance of the in situ systems (Figure 1). The continuous data traces show that the difference between upstream and downstream EC values can be larger than indicated by grab sample data, such as the spot checks. This demonstrates how lower-frequency grab sampling methods, as used for the current routine water quality monitoring program, do not capture the full dynamic range of water quality behaviour.

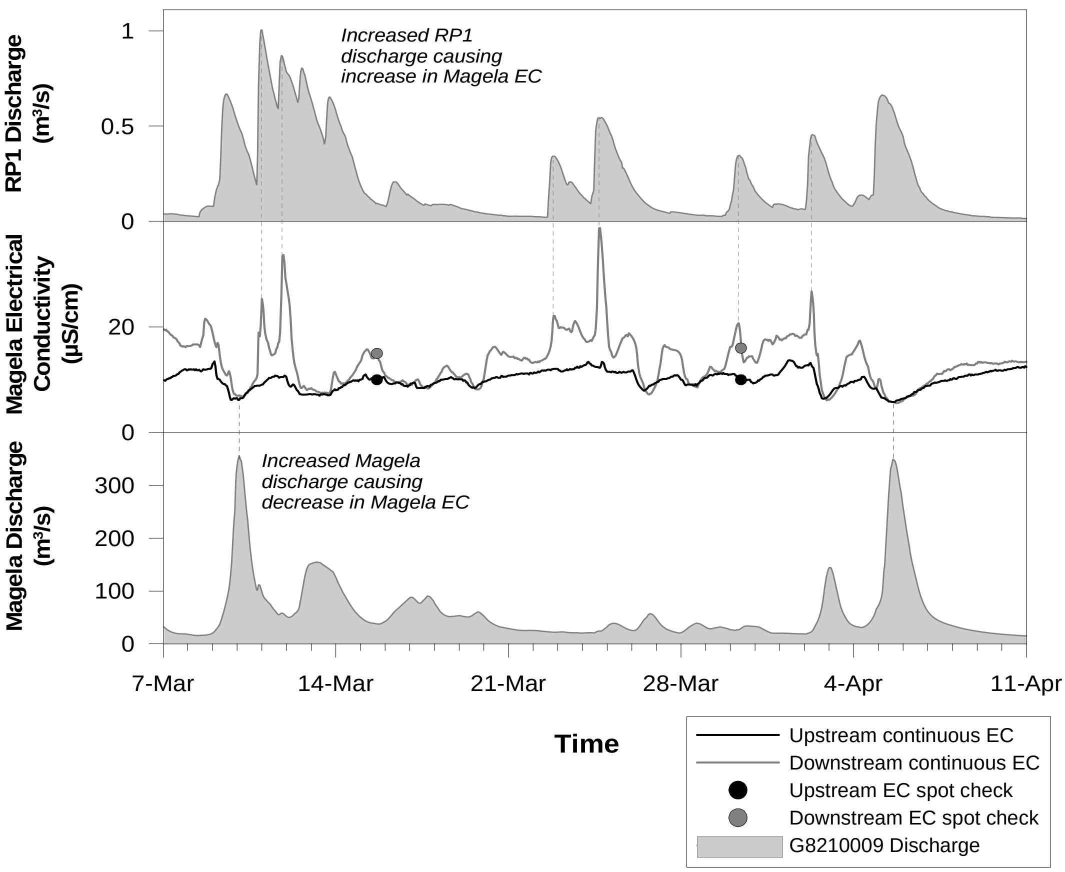

Figure 1 Continuous electrical conductivity (EC) at both the upstream and downstream sites on Magela Creek and quality-control spot checks measured using a calibrated portable field meter over mid-wet season months in 2006. The stream discharge measured at G8210009 and RP1 spillway is also shown.

Although the continuous monitoring data collected during 2005–06 have so far only undergone preliminary analysis, the interaction between inputs of RP1 water and variable dilution by flow in Magela Creek can be clearly seen (Figure 1). The EC downstream of the mine is generally higher and much more variable compared with upstream values. A number of peaks in downstream EC correspond to increased RP1 influence, caused by either an increase in RP1 discharge (due to localised rainfall over the mine site) or a decrease in Magela Creek discharge. Water samples analysed during some of these events provide evidence that elevated EC observed at the downstream site is attributed to elevated magnesium and sulfate concentrations (and to a lesser extent calcium concentrations), typically present in discharged mine wastewaters, particularly from RP1.

However, it should be noted that whilst short duration elevations in solute concentrations can now be detected by the continuous monitoring system, this does not imply that there are adverse effects upon the aquatic environment. The results from the creekside monitoring program, which integrates exposure of test organisms over a time period of one week, provide the assessment tool required to address this issue. The conclusion from the 2005–06 creekside monitoring program, the results from which are presented in detail below, showed no difference in response between the upstream (control) and downstream (impact) locations, indicating no adverse ecological effects arising from these transiently elevated solute concentrations.

The continuous monitoring data collected during the first year of deployment (2005–06) will be rigorously evaluated during 2006–07 in a whole-of-mine catchment context. Data analysis will include:

- accurate calculation of solute loads (previous estimates have required interpolation of the weekly grab sample results) to quantify and compare differences upstream and downstream of the mine and to investigate relative contributions from point (RP1 and GC2) and diffuse (Magela Land Application Area) sources;
- interpretation of observed spatial and temporal variation;
- identification of short-term trends; and
- derivation of more appropriate turbidity trigger values.

The data collected by the continuous loggers will aid interpretation of results from SSD's creekside and water quality monitoring programs. Following analysis and interpretation, the scope of the continuous monitoring program will be reviewed and refined, as required, for the next year of deployment.

Toxicity monitoring in Magela Creek

C Humphrey, D Buckle & R Luxon

Creekside monitoring

In this form of monitoring, effects of water released from the Ranger mine site are evaluated using responses of aquatic animals held in tanks on the creek side and exposed to creek waters. The responses of two test species are measured over a four-day period:

- reproduction (egg production) in the freshwater snail, *Amerianna cumingi* and
- survival of black-banded rainbowfish, *Melanotaenia nigrans* larvae.

Animals are exposed to a continuous flow of water pumped from upstream of the minesite (control site) and from the creek just below gauging station GS8210009 (Map 2), some 5 km downstream of the mine. At each of the two sites, duplicate pumps in the creek each feed water separately to: (i) in the case of snails, a container holding replicate (8) snail pairs (thus 16 pairs of snails exposed per site); and (ii) in the case of fish three containers, each container holding ten larval fish (thus 60 fish larvae exposed per site).

At the end of each four-day test, the mean number of eggs per snail pair and mean number of fish surviving per replicate are noted and compared for each of the upstream and downstream sites. Specifically, when data from the downstream site are subtracted from those at the upstream site, a set of 'difference' values can be derived. These 'difference' values may be compared statistically for different parts of the time-series. For example, 'difference' data for the wet season of interest may be compared with those from previous years. If they differ significantly, using a Student's *t* test, it may indicate a mine-related change. Since about 1996, creekside tests have been performed approximately every second week during the wet season. Tests usually commence in December and cease in early April, covering the period of significant creek flow in Magela Creek.

The results of the creekside trials are plotted as part of a continuous time series of actual and 'difference' data in Figure 1 for snail egg production, and in Figure 2 for larval fish survival. Descriptions of the sources of creekside data and data quality issues are provided in the Supervising Scientist's Annual Report for 2001–02 and web site (http://www.deh.gov.au/ssd/monitoring/magela-bio.html).

Seven creekside tests were conducted in the 2005–06 wet season. Significant pump failure occurred during the fourth test at the upstream site, to the extent that the test did not meet acceptance and validity criteria. The data for this test are displayed in the accompanying figures, however, they are not used in formal statistical analysis to detect and assess potential mining impact. (By convention, the upstream-downstream 'difference' value is omitted from the graphs of test organism responses to signify an invalid test.)

Amongst the snail tests, egg production at upstream and downstream sites was similar across all tests conducted for the wet season (Figure 1). The results also resemble the pattern of egg production observed in previous wet seasons with the possible exception of the relatively low egg production observed at the downstream site in the fifth test. This value was a consequence of significantly lower ($P<0.05$) egg production observed in the duplicate water drawn from the west bank of the creek at the downstream site (mean of 54 eggs per snail vial), relative to the corresponding duplicate water drawn from the east bank at this site (107 eggs per snail vial) and from the two duplicate waters drawn from the upstream site (117 and 123 eggs per snail vial). Corresponding spot water chemistry data collected during this test as part of the SSD's routine monitoring program did not indicate any significant elevation of analytes at this site. Additional water chemistry data, together with continuous datasonde records for key parameters including conductivity and pH, were also collected during this creekside test and the results did not show any significant issues with water quality. Thus the reduced snail egg production observed at the downstream west bank site during the fifth test does not appear to be mine-related.

Using the snail egg production data shown in Figure 1, 'difference' values for 2005–06 were compared with those from previous years. No significant difference was found (P>0.05).

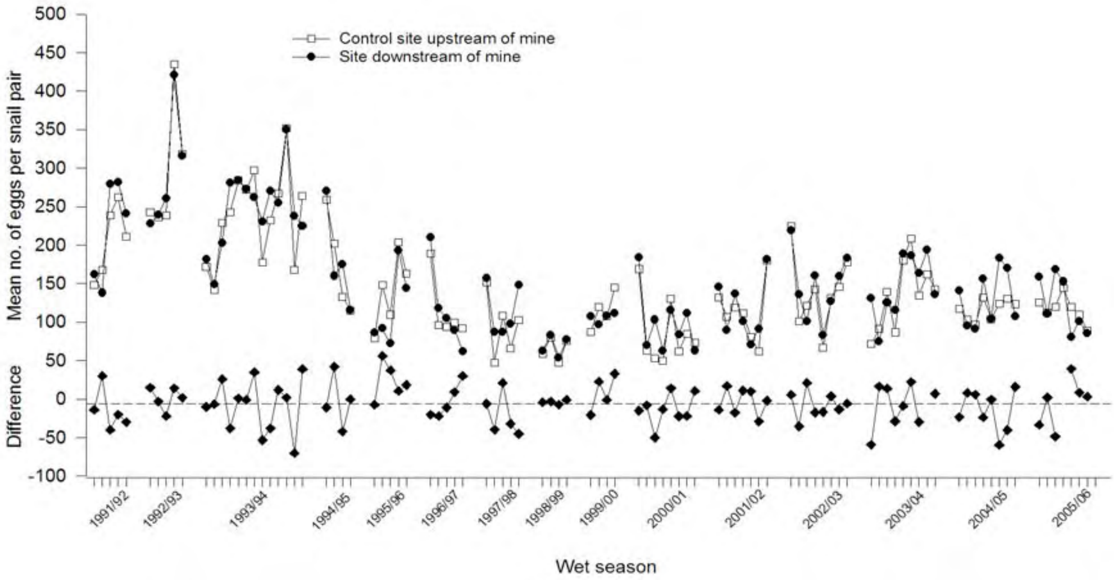

Figure 1 Creekside monitoring results for freshwater snail egg production for wet seasons between 1992 and 2006. (Snail egg production data for the first three tests of 1995/96, all tests for 1997/98, 1998/99 and 1999/00, and the last four tests in 2000/01, were provided by ERA.)

Across all fish tests, larval fish survival at upstream and downstream sites was consistent with the same relative survival rates observed in previous wet seasons with, typically, reduced survival at the upstream site relative to the downstream site (Figure 2). Possible causes for the lower survival at the upstream control site were discussed in the 2002–03 Supervising Scientist Annual Report.

From the collective creekside results, it was concluded that there were no adverse effects of discharged Ranger mine water on Magela Creek over the 2005–06 wet season.

In situ toxicity monitoring

While in situ testing has previously been investigated as a technique for biological monitoring in Magela Creek (Annual Research Summary 1987–88, 1988–89, 1989–90 and 1990–91), the method has remained undeveloped until now because of perceived occupational health and safety advantages of the creekside monitoring procedure (in particular, ready accessibility and safety of staff). However, the high resourcing demands of the existing creekside monitoring program coupled with refinement over the years of the protocols for the snail and fish tests, have led to a re-evaluation of the viability of in situ testing.

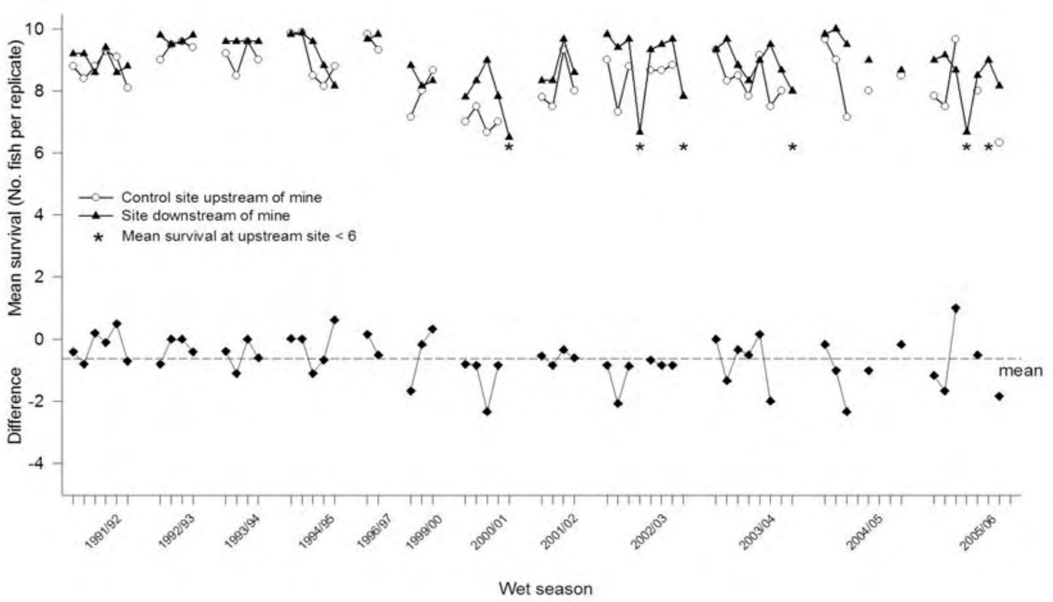

Figure 2 Creekside monitoring results for larval black-banded rainbowfish survival, for wet seasons between 1992 and 2006. (Larval fish survival data for the second test in 1999/00 were provided by ERA.)

Apart from substantially lower resource requirements, in situ testing has inherent technical advantages over the established creekside monitoring approach. These include removal of reliance on powered pumping systems in an area of high electrical storm activity, improved water flow-through and contact conditions for the test organisms, and portability. These advantages make the method appealing for future monitoring at Ranger and, potentially, also for use at other mine sites in the Northern Territory and elsewhere.

Accordingly, work commenced in the 2005–06 wet season to evaluate the potential for in situ deployment, inside floating cages in Magela Creek, of the same snail and fish tests currently used for the creekside monitoring program. This technique would provide a much more cost effective way of providing almost continuous biological monitoring of water quality in Magela Creek.

Preliminary studies involved developing a suitable design of holding vessels for test organisms, and assessing the reproduction response of freshwater snails to a number of holding conditions and feeding regimes. The 2005–06 wet season testwork also included a comparison between the in situ deployment and standard creekside tests of egg production by snails. This initial development work was done at the upstream creekside pump site (near the Magela upstream water quality monitoring site, Map 2).

Preliminary in situ tests were run in parallel with the creekside monitoring tests starting on the 17/02/06, 03/03/06 and 07/04/06 near the upstream creekside pump site. These trials investigated one of two possible feeding regimes: (i) daily feeding per current creekside monitoring protocol, and (ii) feeding only once, at the start of each four-day test. Daily feeding enabled direct comparison of results with those from the existing creekside monitoring program – an essential comparison for the initial stages of the test program. The inclusion of regime (ii) enabled parallel evaluation of a more streamlined protocol.

The results from the daily feeding in situ test are similar to those from the creekside monitoring control site and are almost exactly the same as those from the downstream creekside monitoring site for all three trials (Figure 3). The results obtained from the in situ tests in which food was provided only at the start of the test were encouraging, with close resemblance in egg production to that found for the daily feeding in situ test in the first two trials. If start-only feeding can be used for the in situ method, this will have substantial benefits for staff resourcing. It will also mean that this monitoring technique will be much more viable for extended deployment at less accessible (for example, Gulungul Creek) or more remote locations. Accordingly, both feeding regimes will be further evaluated during the 2006–07 wet season.

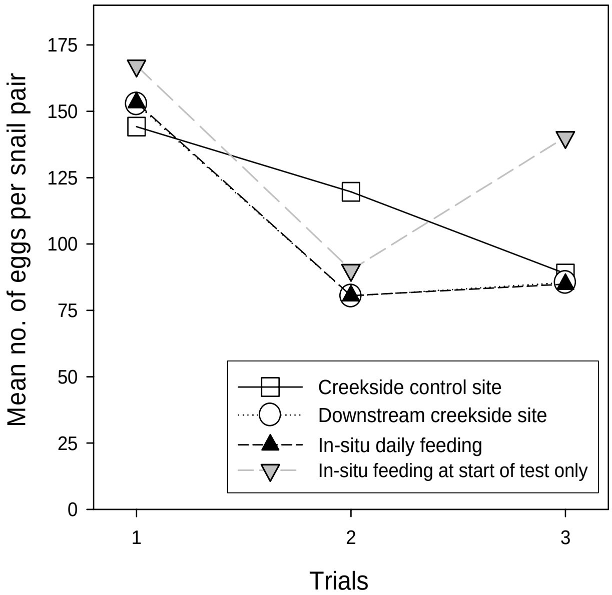

Figure 3 Comparison of freshwater snail egg production for routine creekside monitoring and two feeding regimes of in situ toxicity monitoring

Over a decade of baseline creekside monitoring test data has been obtained since 1991–92 (Figure 1) using the established creekside protocols and infrastructure. It is thus critical to ensure that the proposed in situ method yields comparable results before it can be phased in as the sole procedure in the future. To this end, further testing of the in situ deployment vessels and feeding regimes will be conducted in parallel with creekside monitoring over at least two wet seasons.

The testwork will be extended to the downstream site in the 2006–07 wet season. Future comparative tests will focus on the paired-site monitoring design employed for creekside monitoring (described above) and compare the ‘differences’ in responses between upstream and downstream sites for both test conditions and feeding regimes.

Bioaccumulation in fish and freshwater mussels from Mudginberri Billabong

K Turner, B Ryan, C Humphrey & A Bollhöfer

Mudginberri Billabong is the first major, permanent waterbody downstream (12 km) of the Ranger mine (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. In this context, fitness for consumption refers to levels of metals and/or radionuclides, as related to potential impacts of solutes from the minesite. Microbiological indicators are not measured as part of the SSD's monitoring program.

Any significant increases in metal and radionuclide concentrations in aquatic biota measured through time (or compared to an appropriate reference site) would also provide the potential for early warning of a developing issue with bioavailability of mine-derived solutes. This provides an ecosystem protection role for the bioaccumulation monitoring program, in addition to the human health aspect.

Bioaccumulation data have been obtained from Mudginberri Billabong since 1980 and from a control site (Sandy Billabong, channel, Map 3) since 2002. The concentrations of radionuclides and metals in freshwater mussels from Mudginberri and/or Sandy Billabongs between 1983–2003 were reported and discussed by Ryan et al (2005). Uranium and radium data from this report have been included in the time series of data discussed below.

The metal and radionuclide data from mussels in Mudginberri span a long period, but are intermittent from the early period. In addition, metals 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.

Uranium concentrations in freshwater mussels from Mudginberri and Sandy Billabongs are shown in Figure 1. Uranium in mussels has been reported to have a short biological half-life (Allison & Simpson 1989). This published conclusion is supported by the data in Figure 1, with the uranium concentrations in mussel flesh being low and no evidence of an increasing trend in concentrations with mussel age (Ryan et al 2005). In particular, the concentrations of uranium in mussels from both the 'exposed' and control sites are very similar.

Also of note in the top panel of Figure 1 are the time series data for acid leachable uranium in sediment from Mudginerri Billabong. There is certainly no evidence of an increase through time, with essentially constant levels between 1989 and 2001, and lower levels from 2001 onwards. The decrease after 2001 may be an artefact of changes in the sampling regime or analysis method. A more detailed investigation of the concentrations of metals and radionuclides as a function of particle size class in control and exposed billabongs will be carried out in the second half of 2007.

Apart from uranium and radionuclides, metals data in fish from Mudginberri Billabong are sparse prior to 2000. A focused regular (two year frequency) sampling program was initiated by ***eriss*** in 2000 to measure metal levels. A summary of the work carried out by ***eriss*** since 1995 is provided in Table 1. The analytes chosen include key analytes that are likely to pose the highest risk to the environment and for human consumption in aquatic species that are common food items for traditional owners.

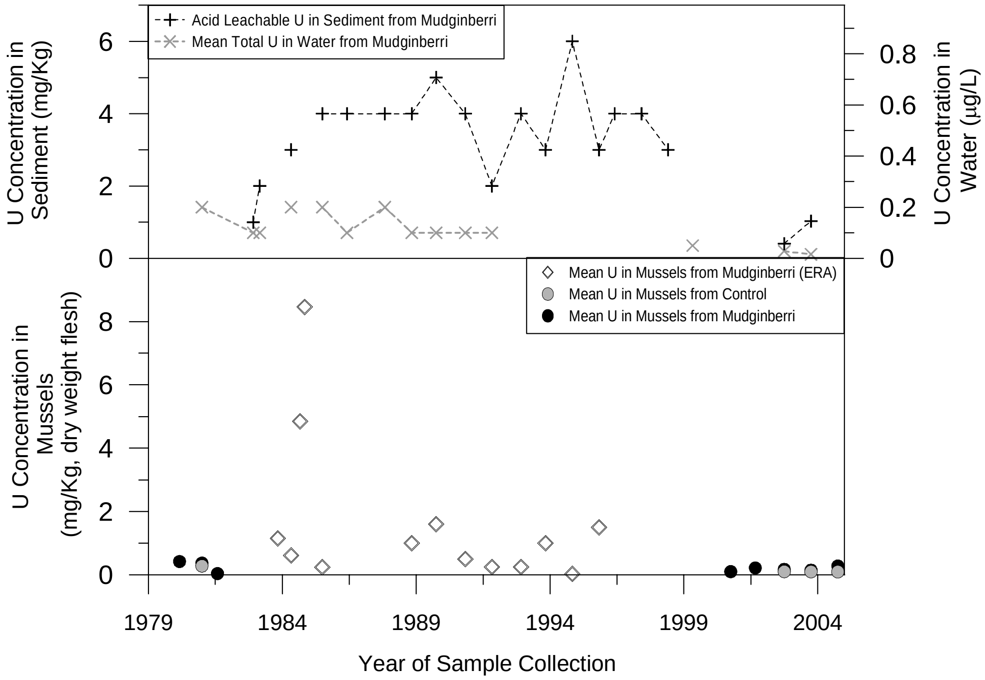

Figure 1 Mean concentrations of U measured in the mussel, sediment and water samples collected from Mudginberri billabong and control billabongs since 1979

Table 1 Summary of bioaccumulation sampling carried out by ***eriss*** in Mudginberri and Sandy Billabongs

Year	Species	Analytes	Tissues analysed
1984	AL	Cu, Zn	Flesh
1985	AL	Cd, Cu, Pb, Mn, Zn, U	Flesh
1988	AL	Cd, Cu, Pb, Mn, Zn, U	Flesh
1995	NE, AL, SJ, LC	Cd, Cu, Pb, Mn, Zn, U	Flesh, liver, kidney
1996	NE, AL, SJ, LC	Samples archived for future analysis	Flesh, liver, kidney
1997	NE, AL, SJ, LC	Samples archived for future analysis	Flesh, liver, kidney
1998	Various Catfish	Samples archived for future analysis	Bone, skin, gills, liver and kidney
2000	TA, AL	Ag, Al, As, Au, Ba, Bi, Ca, Cd, Co, Cr, Cu, Fe, Mg, Mn, Ni, Pb, Sr, U, Zn	Flesh, liver, kidney
2002	AL, TA, SJ, SB, LC	Al, Ba, Cd, Ca, Cr, Co, Cu, Fe, Pb, K, Mg, Mn, Na, Ni, Pb, Sr, U, Zn	Flesh, liver, bone, gill
2003	AD, AG, AL, NE, TA	Al, Sb, Ag, As, Ba, Ca, Cd, Co, Cr, Cu, Eu, Fe, Hg, K, Pb, Mg, Mn, Mo, Ni, Na, Re, Sb, Se, Sn, Sr, Zn, U, SO_4	Flesh, liver, bone, gill
2005	AL, TA, NH, NE, AG, SJ, AD, LC,	Cu, Pb, Mn, U, Zn	Flesh, liver, bone

AL – *Arius leptaspis* (forktail catfish); NE – *Nematalosa erebi* (boney bream); LC – *Lates calcarifer* (barramundi); SJ – *Scleropages jardini* (saratoga); TA – *Neosiluris ater* (eeltail catfish); AG – *Arius graeffei* (blue catfish); AD – *Anodontiglanis dahlia* (toothless catfish); SB – *Syncomistes butleri* (sharp-nosed grunter); NH – *Neosiluris hyrtlii* (Hyrtil's catfish).

After analysis and review of the available data, forktail catfish were identified as the most prospective species to monitor for uranium uptake, accumulating higher levels of metals than other common species inhabiting Mudginberri Billabong (Sauerland 2005). Thus, forktail catfish will continue to be monitored to provide early detection of mining impacts. Time series concentrations of uranium in the flesh of forktail catfish collected from Mudginberri and Sandy Billabongs are summarised in Figure 2, together with U concentrations measured in water and sediment.

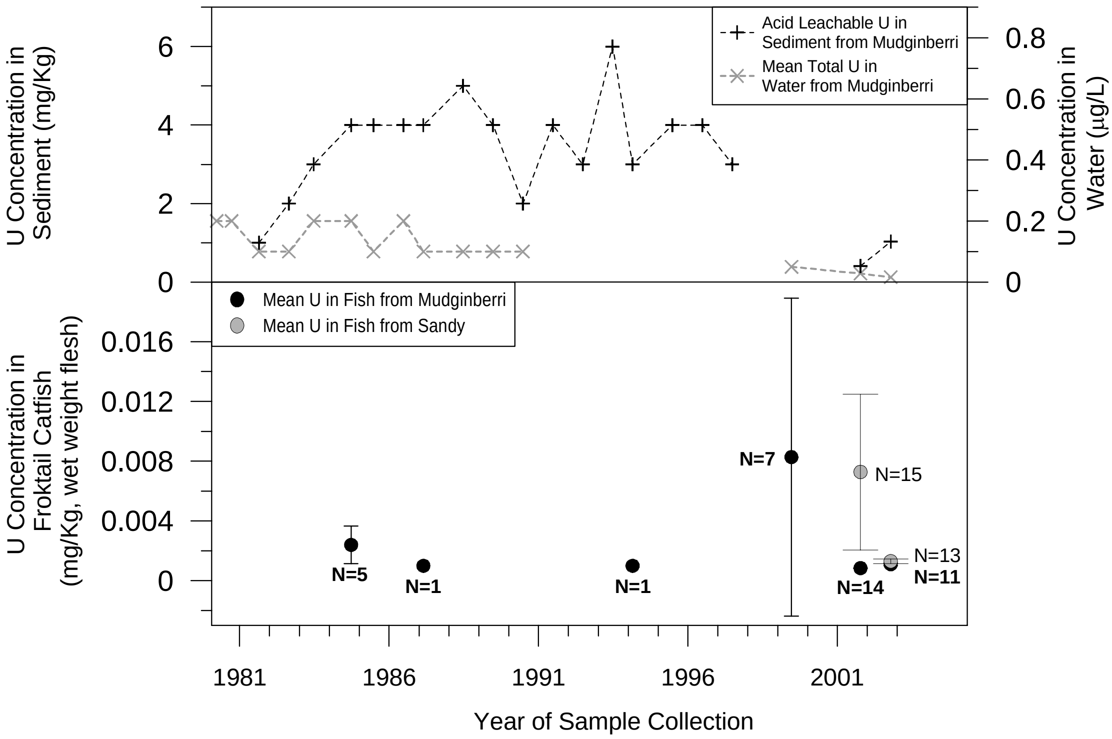

Figure 2 Mean concentrations of uranium measured in the flesh of forktail catfish, sediment and water samples collected from Mudginberri and Sandy Billabongs, since 1981. Error bars represent standard error.

Compared to uranium concentrations in the sediment, the concentrations in the flesh of forktail catfish are low (<0.02 mg/kg) with no significant variation over time (Figure 2). Error bars indicate that sample contamination was a significant issue in 1999 and 2001. Refinement of sample processing methods and analytical procedures reduced the amount of contamination in 2002.

Concentrations of Ra in mussels are age-dependent (Figure 3) and also appear to be related to growth rates and location within a billabong. Mudginberri sediments are finer where the mussels are currently collected, and hence have higher ^{226}Ra values than collections made earlier in the billabong, and as compared to the sandy, coarser sediments in Sandy Billabong (Ryan et al 2005). The need to better characterise sediment is now recognised and more extensive and refined sediment sampling and size fractionation protocols will be used for future sampling (starting end of 2006–07 wet season).

When comparing data from a particular season and a particular within-billabong location (Figure 3), concentrations of Ra in mussels from Mudginberri Billabong are seen to be higher, age-for-age, than in mussels from Sandy Billabong. Naturally higher catchment concentrations of Ra in Magela Creek compared with Nourlangie Creek catchment combined with lower concentrations of Ca (Ca can act as an antagonist to the uptake of Ra by aquatic organisms) in Mudginberri Billabong waters compared with Sandy (see explanation below) are the likely cause.

Earlier studies have shown that the presence of calcium in water reduces the rate of radium uptake and is inversely proportional to radium levels present in freshwater mussels in the region. There is more calcium in Sandy Billabong water, sediments and mussels than in the respective Mudginberri samples during the same period (Ryan et al 2005). This may contribute to the higher levels of ^{226}Ra in Mudginberri mussels when compared to Sandy Billabong mussels.

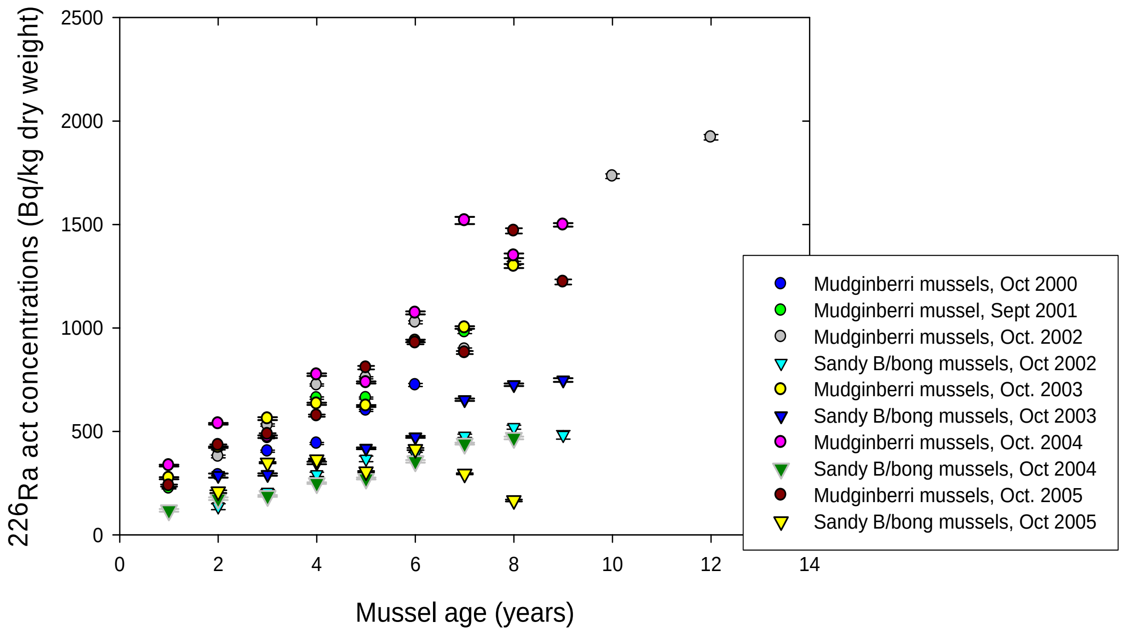

Figure 3 ^{226}Ra activity concentrations in the flesh of freshwater mussels collected from Mudginberri Billabong 2000-2005 and Sandy Billabong 2002–2005 (Ryan et al 2005)

Table 2 shows the current average 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 and Sandy billabong mussels (2000–2005). Even if it was assumed that the difference in doses between the two billabongs was exclusively mine-related (most unlikely, see below), the mine contribution would still amount to only 10 per cent of the public dose guideline limit (ICRP 1996).

Table 2 Average committed effective dose for Mudginberri and Sandy billabong mussels 2000–2005 (Ryan et al 2005)

Sample	Committed effective dose (mSv)
Average all collections from Mudginberri Billabong	0.239
Average all collections from Sandy Billabong	0.133

Currently the extent of mine site influence on radionuclide concentrations in mussels in Mudginberri Billabong is inferred (based on comparison of Ra concentrations in water between MCUS and MG009) rather than being quantified by direct measurement. In this context it should be noted that the concentrations of radionuclides in the sediments of Mudginberri Billabong (and consequently the concentrations of Ra in mussels) may be an historic consequence of the weathering of surface exposed orebodies (#1 and #3) through time, rather than a consequence of the 'recent' mining operation. Sampling of mussels in Magela Creek upstream of Ranger coupled with measurements of radionuclide concentrations in suspended sediments will be used in 2006–07 to further investigate the extent to which the Ranger site contributes to the total concentrations of elements in Mudginberri mussels.

Notwithstanding the above, the generally consistent relationship between age and Ra concentration observed for mussels between years and for each billabong (Figure 3) 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 3) will be explored as a means for quantifying any such future change.

A review of the bioaccumulation study of metals and radionuclides at Ranger, described above, was undertaken in October 2005. A number of recommendations and outcomes arose from the review (Jones 2005), including:

- The need to stream-line the bioaccumulation sampling program for freshwater mussels and fishes, by having uniform sample preparation and analysis protocols for ecosystem and human health requirements (implemented);
- Undertake a 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);
- The need to quantify the relationship between Ra in mussels and the filter-feeder-relevant < 63 µm sediment fraction in Mudginberri Billabong. Only total sediments have been analysed to date so it has not been possible to properly account for differences in Ra concentrations for mussels collected from different parts of the billabongs and to directly compare measured concentrations with those in the coarser sediment from Sandy Billabong (implemented);
- As described above, the need to initiate a sampling program of mussels resident in Magela upstream and downstream (before Mudginberri Billabong) of Ranger to address the issue of whether mine is contributing higher Ra and U in Mudginberri Billabong mussels compared to Sandy Billabong mussels (to commence in May 2007).

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Monitoring using macroinvertebrate community structure

C Humphrey, J Hanley & C Camilleri

Macroinvertebrate communities have been sampled from a number of sites in Magela Creek at the end of significant wet season flows, each year from 1988 to the present. The design and methodology have been gradually refined over this period to both improve the ability to confidently attribute any observed changes to mining impact and to carry out the work more efficiently. The most significant refinement that took place in the study occurred in 1994 when there was a reduction from ten sites sampled in Magela Creek to just three, as well as commencement of sampling at reference sites in three additional streams not thought to be impacted by mining. Since 1994, there have also been three changes to sampling and sample processing methods (Humphrey & Pidgeon 1998).

As described in the previous Research Summary (2004–2005, SSR189), the refined (1994) design for this macroinvertebrate study is now based on the principle of gathering macroinvertebrate samples from sites in Magela and Gulungul Creeks upstream and downstream of Ranger (Gulungul Creek now no longer regarded as a reference stream), and also from similar paired upstream and downstream sites in two adjacent 'control' streams (Baroalba and Nourlangie Creeks) that are generally unaffected by any mining activity (Map 3). Thus the design of this study is a balanced one comprising two 'exposed' streams and two control streams.

Five replicate samples (each 0.31 m^2 in area) were collected from macrophyte-edge habitat at each site, using a Surber sampler, at the end of each wet season (between April and May, 2006). 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 identical macroinvertebrate communities while a value of 'one' indicates totally dissimilar communities, sharing no common taxa.

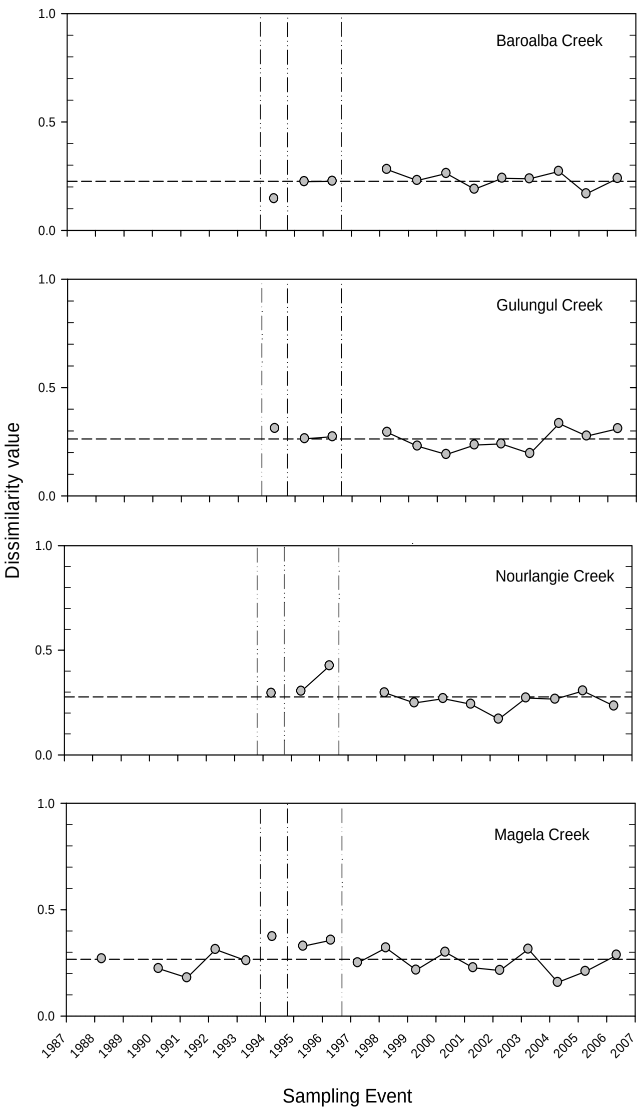
Research elsewhere in the Alligator Rivers Region (eg Faith et al 1995) has shown significantly 'higher' dissimilarity values for locations upstream and downstream of point sources of disturbance compared with values recorded in the pre-disturbance, baseline period in a stream prior to the start of an impact, and in still-undisturbed control streams. The higher dissimilarity is a consequence of the 'altered' (disturbed) macroinvertebrate community structure downstream of such point sources.

Analysis of the full macroinvertebrate data set from 1988 to 2006 has been completed and results are shown in Figure 1. This figure plots the paired-site dissimilarity values using family-level (log-transformed) data, for the two Magela catchment streams and two Nourlangie catchment (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 for Magela and Gulungul Creeks upon which to assess whether or not significant changes have occurred as a consequence of mining. Notwithstanding this, 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, singlefactor ANOVA shows no significant difference in the mean dissimilarities between the two treatment groups, 'control' versus 'potentially disturbed' streams.

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 2006), together with all other control sites sampled for the same period. Because the data-points associated with these two sites are interspersed amongst the points representing the control sites, this indicates that these 'exposed' sites have macroinvertebrate communities that are not dissimilar to those occurring at control sites.

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 uranium mine for the period 1988 to 2006. 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.



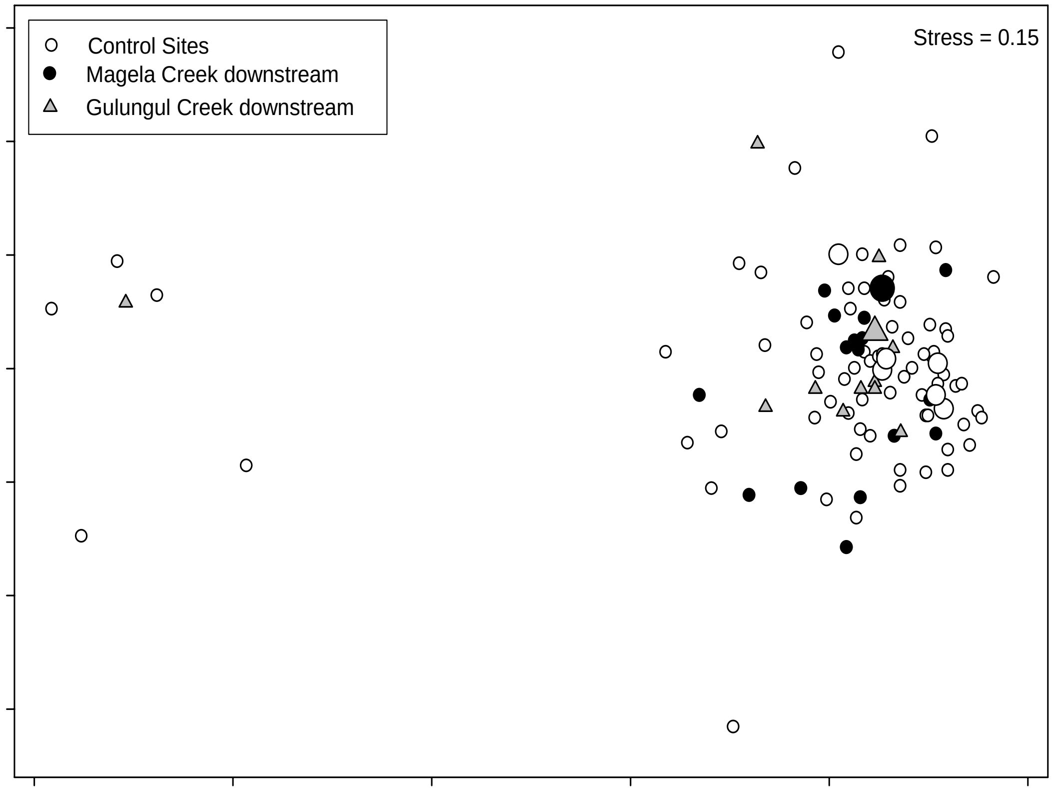


Figure 2 Ordination plot of macroinvertebrate communities sampled from sites in several streams in the vicinity of the Ranger mine for the period 1988 to 2006. Data from Magela and Gulungul Creeks for 2006 are indicated by the enlarged symbols.

Collectively, these results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining in the period 1994 to 2006, 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. The results from this billabong study and those acquired in future years from the same sites may also serve an important biological monitoring role.

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Monitoring using fish community structure

C Humphrey & D Buckle

Sampling of fish communities in billabongs is conducted between late April and the end of June each year. Two types of data are gathered, using non-destructive sampling methods:

1. Visual observation data from two deep channel billabongs: Mudginberri Billabong on Magela Creek about 12 km downstream of Ranger ('exposure' billabong, 1989–present); and Sandy Billabong on Nourlangie Creek (control billabong, independent catchment, 1994–present).
2. Data from 'pop-nets' set in shallow weedy lowland billabongs, in various combinations of billabong exposure types, located in Magela and Nourlangie Creek catchments, from 1994 to the present.

Shallow lowland billabongs

Fish in shallow billabongs were not sampled in 2006 given the extensive (annual) database already available and the results from power analysis which indicated that not sampling after the 2005-06 wet season would not significantly reduce the power of the historical data set.

This program of work was reviewed, along with the broader biological monitoring program, in early October 2006. The outcomes of this review will be reported in next year's Annual Research Summary.

Channel billabongs

The extent of similarity between fish communities in Mudginberri Billabong (impact site downstream of Ranger) and Sandy Billabong (control site in the Nourlangie catchment) (Map 3) was determined using multivariate dissimilarity indices. Calculated for each annual sampling occasion, the dissimilarity index is a measure of the extent to which fish communities of the two sites differ from one another. A value of 'zero' indicates identical fish communities while a value of 100% indicates totally dissimilar communities, sharing no common species. A significant change or trend in the dissimilarity values over time could imply mining impact. A plot of the dissimilarity values from 1994 to the present is shown in Figure 1.

Dissimilarity indices may also be mapped in an ordination to depict the relationship of the community sampled at any one site and sampling occasion with all other possible samples. Shifts in fish community structure have been observed in both billabongs over time (Figure 2). While the sites do not faithfully 'track' one another from year to year, the patterns and extent of 'meandering' are not too dissimilar. In the last four years, community structure in both sites has become more similar (annual data points in Figure 2 for the two sites are closer together), and this community structure has also become more similar to that found at the commencement of the paired-site study in 1994.

In the Supervising Scientist Annual Report for 2003–2004, a significant decline was noted in the paired-site dissimilarity measures over time. This decline has continued (Pearson's correlation R = -0.70, P<0.05) with the value reported in 2006 the lowest yet recorded (Figure 1). The decline is primarily attributed to the particularly high abundances in the early years of the study of chequered rainbowfish (*Melanotaenia splendida inornata*) and to a lesser extent glassfish (*Ambassis* spp) in Mudginberri Billabong, 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 more 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.

Further work will be needed to elucidate the cause of the decreasing dissimilarity of fish communities between Sandy and Mudginberri Billabongs. The continued decline has been less influenced by chequered rainbowfish and glassfish in the latter years, suggesting more subtle changes in community structure are also occurring.

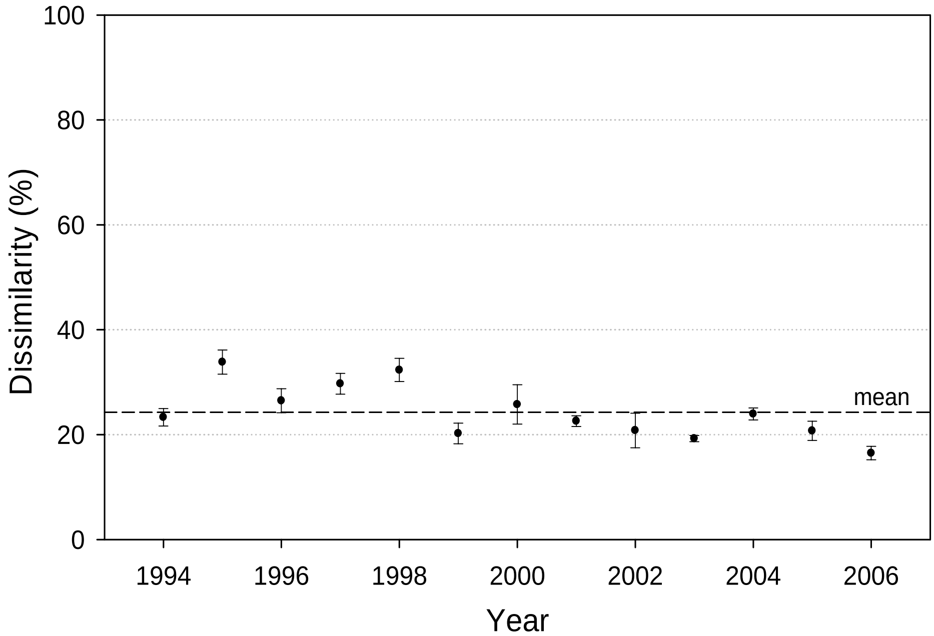

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') billabongs in the vicinity of the Ranger uranium mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

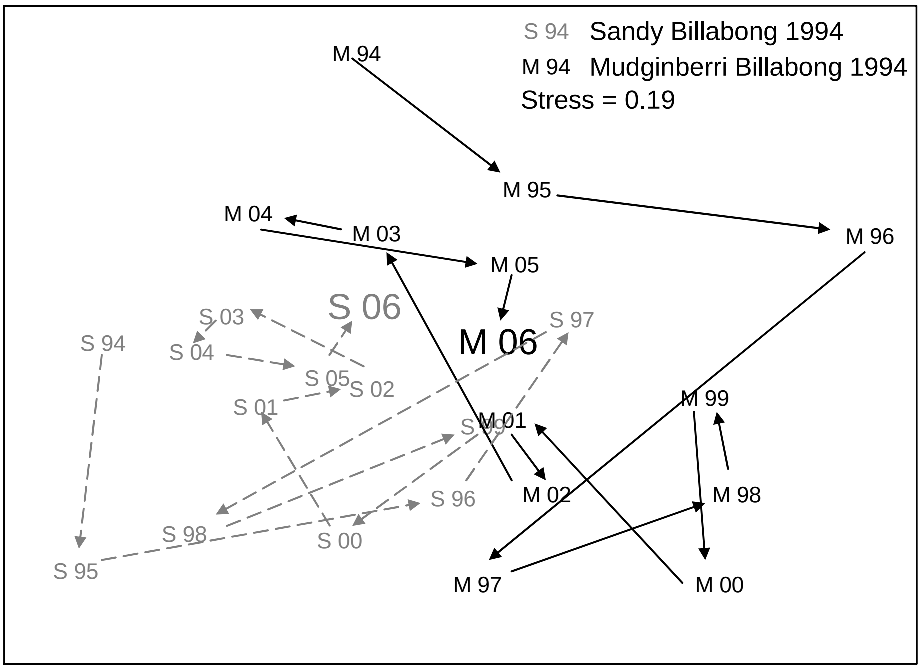

Figure 2 Ordination plot of fish communities sampled from two channel billabongs in the vicinity of the Ranger uranium mine for the period 1994 to 2006. Lines follow the trajectory of sites over time. The two-dimensional MDS was generated using Log(X+1) data. Data were averaged prior to generating Bray-Curtis dissimilarity matrix.

Monitoring support tasks

C Humphrey

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 was reported to ARRTC in late February 2005. At that time, it was proposed to ARRTC that publication of the high-level protocols in the SSR series would be completed in 2006. Delays in publication have been experienced due to:

1 Competing priorities (including rehabilitation research); but most significantly

2 The need to document, as a priority, operational manuals pertaining to each of the protocols. This task was rated a high priority because of the risks associated with loss of corporate knowledge should critical staff leave the SSD. This task has diverted staff time away from completion of the high level protocols.

Work on the operational manuals and high-level protocols is well advanced and is anticipated that they will be completed in time for the commencement of the respective monitoring techniques in the upcoming 2006–07 wet season.

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; and

   c. competing resources in so far as possible increased intensity of new monitoring approaches and rehabilitation research

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

3 Wishes of stakeholders, including local landowners

Preliminary outcomes of the review were presented to the 18th meeting of ARRTC (October 2006) while the proceedings are currently being prepared as a Supervising Scientist Internal Report.

Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer, P Medley & C Sauerland

Introduction

Since 2001 the monitoring techniques developed by SSD for environmental assessment of aquatic ecosystems have been implemented by way of a routine monitoring program. This includes the measurement of physical, chemical and radiological indicators in Magela and Gulungul Creeks, upstream and downstream of the Ranger mine (Supervising Scientist, 2004).

Surface water samples in the vicinity of the Ranger and Jabiluka project areas are regularly monitored for their ^{226}Ra activity concentrations to assess if there have been any changes in ^{226}Ra activity concentration downstream of the mine sites, and hence in the potential risk of increased exposure to radiation via the biophysical pathway due to mining-related activities. Water samples are collected weekly in Magela Creek and monthly in Ngarradj (Swift) Creek at sites both upstream and downstream of the project areas according to the ***eriss*** surface water monitoring protocol (Sauerland & Iles 2005).

Methods

All Ngarradj samples, and Magela Creek samples from every second week, are analysed for total ^{226}Ra (ie, combined filtered and particulate fractions) in ***eriss***'s radiochemical laboratories following a method described in Medley et al (2005).

The remaining fortnightly samples for Magela Creek are combined into '*wet season composite samples*', one for the upstream site and one for the downstream site.

Results

Magela Creek

Figure 1 shows the results of the ^{226}Ra monitoring from 2001–2005. The levels of ^{226}Ra are very low in Magela Creek. A paired two-tailed t-test indicates that the ^{226}Ra activity concentrations are not significantly different between the Magela creek downstream and upstream sites for the 4 wet seasons. The total ^{226}Ra activity concentrations at the upstream site can at times be higher than at the downstream site. However a ^{226}Ra activity concentration of 8.8 mBq/L^{-1} for the upstream site on the 15th of February 2005 was probably due to contamination along the sampling and processing chain of that particular sample.

Table 1 shows the median and standard deviations of the means for individual wet seasons for the entire study period. 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 2004–05 the ^{226}Ra activity concentrations of the '*wet season composite sample*' from Magela Creek both upstream and downstream, was 2.4 mBq/L.

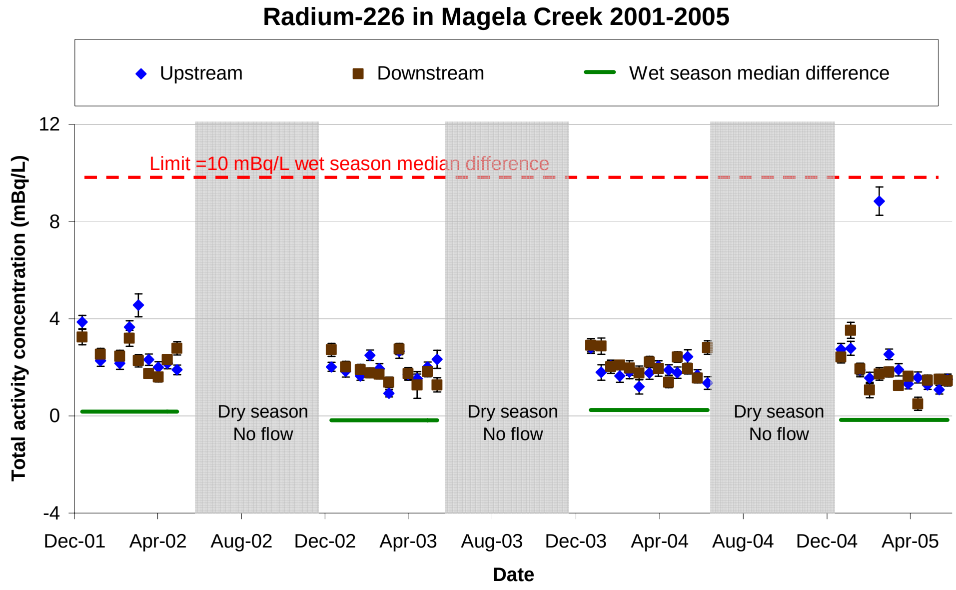

Figure 1 Time series of total radium-226 activity concentrations in Magela Creek (2001 to 2005)

A limit for ^{226}Ra concentration in surface waters downstream of Ranger has been defined for human radiological protection purposes (Klessa 2001) and is based on the potential dose received from the ingestion of ^{226}Ra in the freshwater mussel *Velesunio angasi* (Martin et al 1998). The upstream median value is subtracted from the median at the downstream site (Sauerland et al 2005) – the wet season median difference (shown in Figure 1) – and should not exceed10 mBq/L.

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

Magela creek		All years	2001–02	2002–03	2003–04	2004–05
Median and standard deviation of the mean	upstream	1.9 (± 1.2)	2.3 (± 1.1)	2.0 (± 0.5)	1.8 (± 0.4)	1.7 (± 2.1)
	downstream	1.9 (± 0.6)	2.5 (± 0.6)	1.8 (± 0.5)	2.0 (± 0.5)	1.6 (± 0.7)
Wet season median difference		0.0	0.2	- 0.2	0.2	- 0.2
Ngarradj		All years	2001-02	2002-03	2003-04	2004-05
Median and standard deviation of the mean	upstream	1.2 (± 0.5)	1.2 (± 0.6)	1.4 (± 0.6)	1.1 (± 0.4)	1.3 (± 0.3)
	downstream	1.2 (± 2.0)	3.0 (± 2.8)	1.1 (± 1.5)	0.9 (± 0.9)	1.0 (± 0.6)
Wet season median difference		0.0	1.8*	- 0.3	- 0.2	-0.3

* note that the error of this number is greater than 1.8

The wet season median difference for the 2001–05 wet seasons is approximately zero. The available data for the four sampling seasons indicate that ^{226}Ra levels in Magela Creek are due to the natural occurance of radium in the environment and that mine origin radium has not caused any impact on human health.

Ngarradj Creek

The very low ^{226}Ra activity concentrations in Ngarradj Creek are shown in Figure 2 below. Although there were significant differences observed during the first two wet seasons between the upstream and the downstream monitoring site, Figure 2 shows that ^{226}Ra activity concentrations at the Ngarradj downstream site have been similar to those at the upstream site since December 2003, coinciding with the inception of the long-term care and maintenance phase at Jabiluka in the 2003 dry season. A paired two-tailed t-test of the data from the last two wet seasons indicates that there is no significant difference between upstream and downstream ^{226}Ra activity concentrations at the 95 % confidence level.

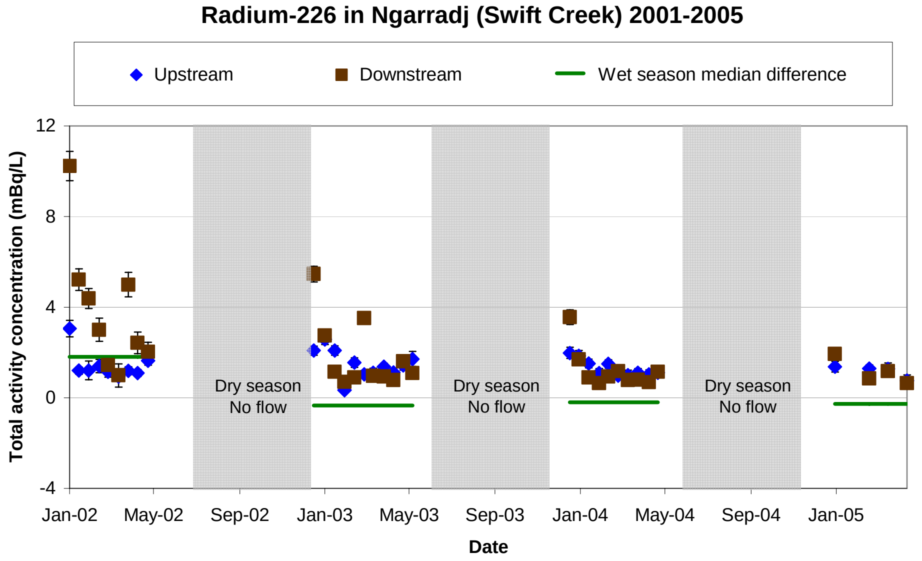

Figure 2 Radium-226 concentrations measured in Ngarradj from 2001 to 2005

Steps for completion

^{226}Ra activity concentration data for the 2005–06 wet season are currently being analysed and compiled and will be reported when complete. The monitoring of ^{226}Ra in Magela Creek and Ngarradj will continue in the 2006–07 wet season.

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Surface water transport of uranium in the Gulungul catchment

K Mellor[1], A Bollhöfer, C Sauerland & D Parry[1]

Introduction

Gulungul Creek lies to the west of Ranger mine and flows north to join Magela Creek, a tributary of the East Alligator River. Part of the mine's infrastructure, notably the tailings dam, lies partially within the Gulungul Creek catchment (Figure 1). Flow in the creek occurs mostly in the wet season, during which time it is made up of the main channel and numerous side channels and tributaries, three of which flow from areas possibly influenced by the Ranger mine.

Since 2001 the water quality monitoring program for Gulungul Creek upstream and downstream of the Ranger mine has included the measurement of physical and chemical indicators, including pH, electrical conductivity, suspended solids, sulphate and uranium concentration, (Supervising Scientist 2004).

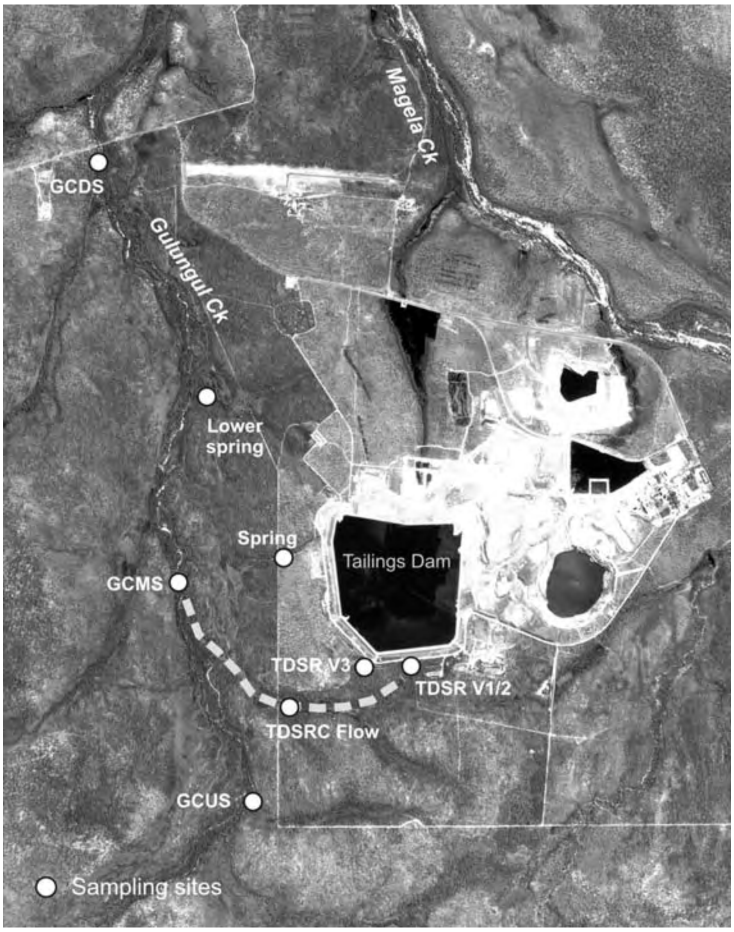

Figure 1 The Gulungul catchment and project sampling sites

[1] Charles Darwin University, Darwin

In January 2004, higher concentrations of uranium were found at Gulungul Creek downstream (GCDS) compared to Gulungul Creek upstream (GCUS). This difference in concentrations was greater than in previous years, and coincided with lower pH values, higher EC values and higher sulphate concentrations at the downstream site. Although the uranium concentrations were more than one order of magnitude less than the 99% ecosystem protection limit value of 6 µg/L established for Magela Creek, they were comparable to the focus trigger value for GS009 in Magela Creek. An investigation was therefore initiated to identify the source and mechanism of the increase.

Methods

In addition to the routine monitoring samples that were collected during the 2004–05 wet season, several field trips were conducted to collect surface water samples from elsewhere in the catchment for chemical and radionuclide analysis. Samples were collected from the upstream (GCUS), downstream (GCDS) and midstream (GCMS) sites and from several locations in the vicinity of GCMS, ie downstream at 'GCMS – 10 m', 'GCMS – 50 m' and upstream at 'GCMS + 50 m' and 'GCMS + 150 m'. Additional samples were taken from V-notches (TDSRV 1–3) located upstream of the tailings dam southern road culvert (TDSRC), the overland flow from TDSRC (TDSRC flow), a spring tributary flow (Spring) and from a swampy area produced by another suspected spring (Lower Spring) (Figure 1). Samples were collected in (i) February 2005, several days after a four day period of heavy rain that resulted in the flooding of the creek; (ii) in March 2005, towards the middle of the wet season when the creek was reasonably full; and (iii) in May 2005, towards the end of the wet season when water flow was much diminished and sampled tributaries had dried up.

Heavy metal and uranium concentrations were measured via ICPMS. Activity ratios of uranium isotopes were measured via alpha spectrometry to determine whether there was a difference in upstream and downstream activity ratios that may enable discrete contributing sources to be identified (Ivanovich & Harmon 1982). Uranium activities were extremely low and the standard chemical pre-concentration procedures needed to be modified to increase the chemical recovery of uranium from solution prior to isotope analysis.

Results

Uranium concentration

Figure 2 shows the routine weekly uranium monitoring results for GCUS and GCDS, the difference between downstream and upstream uranium concentrations, and the catchment rainfall measured at Jabiru Airport during the 2003–2004 and 2004–2005 wet seasons.

Differences between downstream and upstream uranium concentrations in Gulungul Creek during the first part of the 2004–05 wet season were less pronounced than in the previous wet season. Although there is no direct correlation between rainfall and uranium concentration over these wet seasons, it appears that the cumulative effect of heavy rain influences the difference in uranium concentration measured upstream and downstream of the mine. Rainfall was reasonable heavy and constant during December-January (582 mm in 25 days) leading up to the uranium increase at GCDS in the 2003–04 wet season. In the 2004–05 wet season rainfall was lower and less frequent, apart from two large rain events in early January and February.

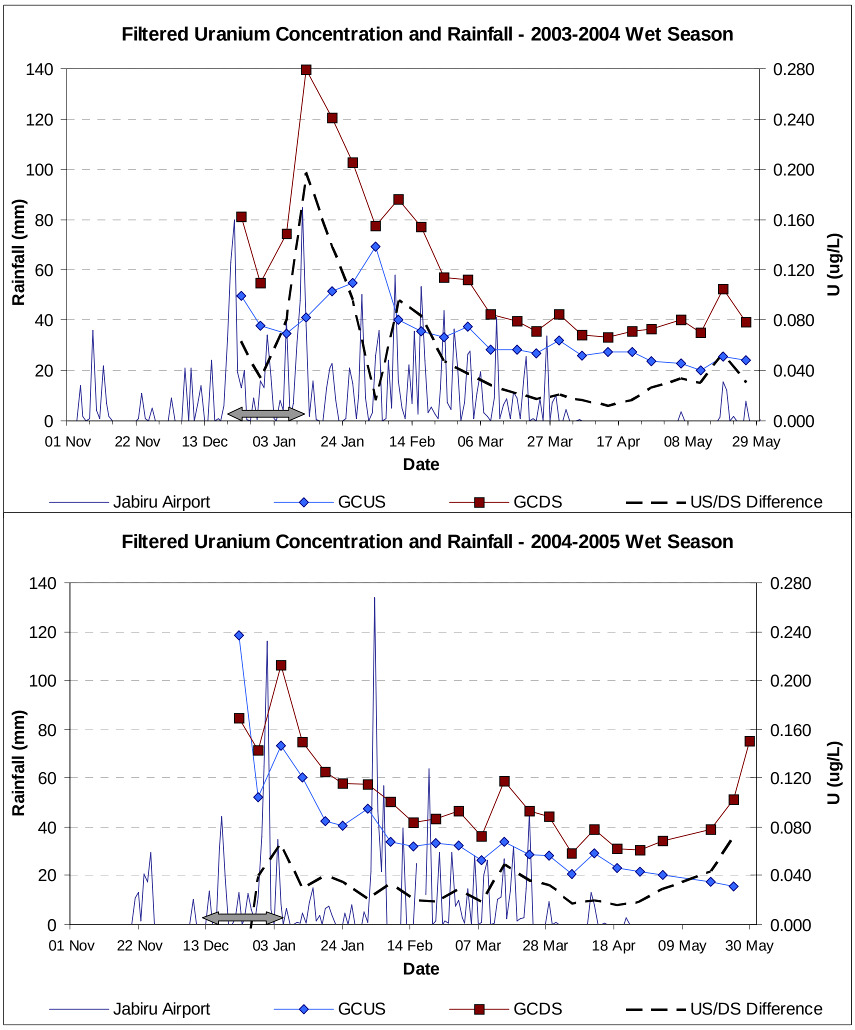

Figure 2 Uranium concentration and rainfall in the Gulungul Creek catchment

Table 1 shows the spatial distribution and the per cent increases in uranium concentration along Gulungul Creek measured in 2005. The data indicate that during the peak of the wet season (February, March), much of the increase in uranium concentration between GCUS and GCDS had already occurred at GCMS. The disparity in uranium input over time indicates that uranium input into the creek may be dependent on specific hydrological conditions. This is supported by the 2005–06 wet season data (Figure 3), which exhibit a uranium spike even higher than in 2003–04, and a similar rainfall pattern: the increase in uranium is preceeded by a period of relatively heavy, constant rain (515 mm in 25 days).

Table 1 Percent increase in filtered uranium concentration in Gulungul Creek in 2005 (dd/mm)

Site	Distance (m)	Filtered U (µg/L)			Increase from GCUS to GCDS (%)			Increase above GCUS (%)		
		08/02	18/03	10/05	08/02	18/03	10/05	08/02	18/03	10/05
GCUS	0	0.068	0.060	0.045	0	0	0	0	0	0
GCMS +150m	1950	ns	ns	0.052	-	-	21	-	-	17
GCMS +50m	2050	0.087	0.093	0.053	58	74	23	28	56	19
GCMS -10m	2215	ns	0.105	0.054	-	100	26	-	76	22
GCMS -50m	2255	ns	ns	0.055	-	-	27	-	-	23
GCDS	6520	0.101	0.105	0.082	100	100	100	48	76	85

ns: not sampled

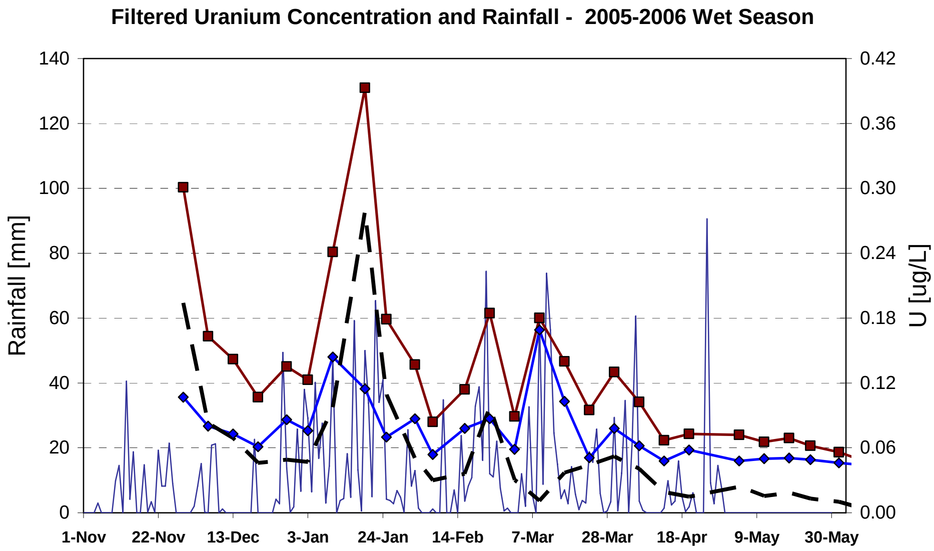

Figure 3 Uranium concentration and rainfall in Gulungul Creek, 2005–06 wet season

Uranium isotope activity ratios

The $^{234}U/^{238}U$ activity ratios measured in Gulungul Creek samples from 2004 and 2005 range from 1.26–1.69 for the upstream site, and from 1.12–1.45 for the downstream site. Although changes in upstream and downstream uranium concentrations do not appear to directly influence the magnitude of the ratios measured at each of these locations, the midstream ratios in samples collected in 2005 do exhibit such an influence. Figure 4 shows the inverse concentration plots for samples taken at GCUS and GCMS. Whereas GCUS uranium activity ratios are relatively constant at approximately 1.4, the midstream site exhibits a signature indicating mixing of two sources with endmembers represented by the upstream ratio and a ratio approaching 1. A $^{234}U/^{238}U$ activity ratio close to 1 is found for uranium in the flow from the tailings dam southern road culvert (TDSRC) that enters Gulungul Creek in the vicinity of GCMS, and represents a ratio typical for a mine-related source (Iles et al 2002).

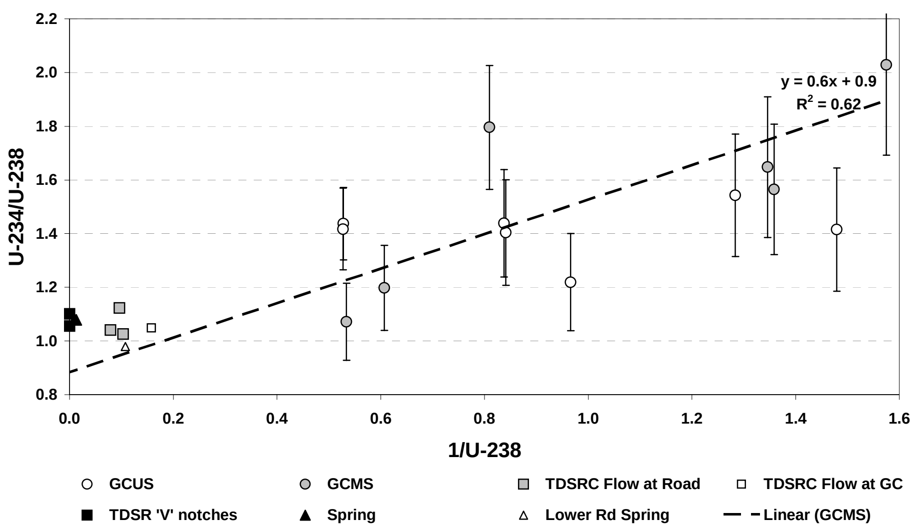

Figure 4 Uranium activity ratio data for 2005 plotted versus the inverse total uranium-238 activity concentration (L/mBq) measured at GCUS, GCMS and various possible sources in the catchment

Soil sampling

To further investigate the cause for the increase in uranium concentration measured in the creek, soil samples were taken in the dry season along the tributary indicated by the dashed line in Figure 1. Uranium concentrations in the soils decrease by almost 3 orders of magnitude from the tributary source close to the tailings dam, to where it enters Gulungul Creek, indicating significant attenuation of uranium, and other metals including manganese, copper and nickel. Leaching experiments show that much of this uranium and other heavy metals is able to be readily leached from the dry soils into water.

The previously identified large areas of black soils in the Gulungul catchment (Crossing 2002, Klessa & Welch 2003) thus appear to be acting as a sink and a source of contaminants from one season to the next. During the initial phase of the following wet season the metals leached out of the black soils in the vicinity of GCMS are flushed down into Gulungul Creek and cause elevated concentrations of uranium and other heavy metals, and a decrease in uranium isotope activity ratios in the vicinity of GCMS.

Summary

There is an input of U and other heavy metals into Gulungul Creek between the upstream and midstream sites, which produces consistently lower $^{234}U/^{238}U$ activity ratios at the downstream site. The contaminating uranium exhibits a $^{234}U/^{238}U$ activity ratio of ~1 and may originate from areas of black soils between the tailings dam and GCMS, which contain readily available metals. The metals are leached from dry soils at the start of the wet season, then accumulate and flush down into the creek under specific hydrological conditions. Uranium loads and discharges from this source required to shift uranium activity ratios in Gulungul Creek from a maximum of 1.40 at GCUS to 1.20 at GCDS (March 18, 2005) have been estimated at 665 µg/s and 1740 L/s, respectively, which is less than ¼ of the GCUS discharge, and is considered a realistic possibility. Future work will focus on event based sampling at GCMS and subsequent measurement of heavy metals, uranium and uranium activity ratios.

Acknowledgments

Thanks to Peter Medley, Jared Sellwood and Dene Moliere for their support of the laboratory and field work, and to Paul Martin and Arthur Johnston for initial discussion about formulating the project.

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