Part 1: Ranger – current operations

Contents[1]

1 List of papers grouped by Key Knowledge Need

Development of a contaminant pathways conceptual model for Ranger mine

R van Dam & P Bayliss

Background

A conceptual model of contaminant pathways from the operational phase of Ranger uranium mine is being developed. The early development of the model was reported on by Finlayson and Bayliss (2003) and van Dam et al (2004). The primary purpose of the conceptual model will be 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 5.1.1), will serve as a communication tool for both scientists and traditional owners. The summary below focuses on progress towards the finalisation of the technical content of the model and an overview of the proposed involvement of, and communication with, traditional owners to complete the model.

Progress

To date, the development of the conceptual model has involved an internal technical expert panel approach to identify, and agree on, the relevant details within the following model elements:

- stressors (chemical, physico-chemical, radiological, biological)

and then for each stressor, its:

- sources
- transport mechanisms off-site
- affected environmental compartments
- receptor organisms
- routes of exposure
- types of effect
- measures of effect

Details within each of these elements have been compiled and constructed diagrammatically, as shown in figure 1. For each stressor transport pathway sub-model, the relevant linkages have been made that indicate how a stressor leaves the mine site, where in the environment it is distributed, what biota (including humans) could be exposed and affected, and what monitoring is in place, or needs to be in place, to detect effects.

As an example, the sub-model for inorganic toxicant transport via one of several specific surface water transport mechanisms is shown in figure 2. Amongst other pathways, inorganic toxicants can be transported off the mine site by surface water runoff and/or direct mine water discharges/overflows, and enter the nearby creeks. Depending on the nature of the toxicant, once in the creek it can partition into, and move between, various environmental compartments, most notably the water column, the sediment or the resident aquatic biota. Various biotic groups will be potentially exposed to the toxicant in these environmental compartments. For example, phytoplankton exposed directly from the water column, fish exposed directly from both the water column and from consuming prey that have taken-up and accumulated the toxicant, and benthic macroinvertebrates exposed directly from the sediment. In addition, people and other terrestrial animals can consume aquatic organisms that have taken-up and accumulated the toxicant. However, the extent to which the various exposure pathways are relevant and important in terms of resulting in adverse effects on receptor organisms depends on many biotic and abiotic factors (eg. relative contribution of the pathway to the total exposure of the receptor organism, sensitivity/tolerance of the receptor organism to the contaminant, environmental conditions that influence the bioavailability of the contaminant). To ensure that the Supervising Scientist can meet its primary objective of ensuring the protection of the people and the environment of the Alligator Rivers Region from the effects of uranium mining (termed as *assessment endpoints* in the conceptual model), various physico-chemical and biological indicators or *measurement endpoints* are assessed and monitored, as surrogates of the assessment endpoints. As can be seen in figure 2, the model indicates the relationship between the receptor organisms and the assessment endpoints (SSD objectives) and measurement endpoints (SSD monitoring and assessment programs).

Steps for completion

The following tasks will be undertaken in 2005–06 to complete the contaminant pathways conceptual model:

1 External technical stakeholder (ERA, EWLS, DBIRD) meeting to affirm and finalise technical aspects of the model;

2 Traditional Owner consultation: Other than risk assessment, one of the key functions of the conceptual model will be as a communication tool for outlining the types, pathways and relative risks of contaminants from Ranger. Suitability of the conceptual model as a communication tool for Traditional Owners will be assessed by incorporating their views and input through a workshop facilitated by the Aboriginal Communications Officer. Following the workshop an iterative process of consultation will be undertaken to ensure their input is correctly presented. It is anticipated that the final communication product for Traditional Owners will be presented in DVD format and a final graphical and narrative descriptions of the model.

When complete, the conceptual model will represent an iteration of previous models, having incorporated new knowledge and understanding of the relevant processes and issues. It will provide the framework for a quantitative comparison of risks (see summary for ARRTC KKN 5.1.1) and uncertainties for all stressor transport pathway sub-models. It will also enable historical and current activities and priorities to be evaluated, and future priorities to be determined. Further, it will act as a knowledge and communication management tool, within a risk management framework that links clearly to the ongoing management of mining operations.

References

Finlayson M & Bayliss P 2003. *Conceptual model of ecosystem processes and pathways for pollutant/propagule transport in the environment of the Alligator Rivers Region.* Discussion Paper prepared for the 11th meeting of ARRTC, 17–19 February 2003.

Van Dam R, Finlayson M & Bayliss P 2004. Progress on the development of a conceptual model of contaminant pathways from Ranger uranium mine. Internal Report 474, June, Supervising Scientist, Darwin. Unpublished paper.

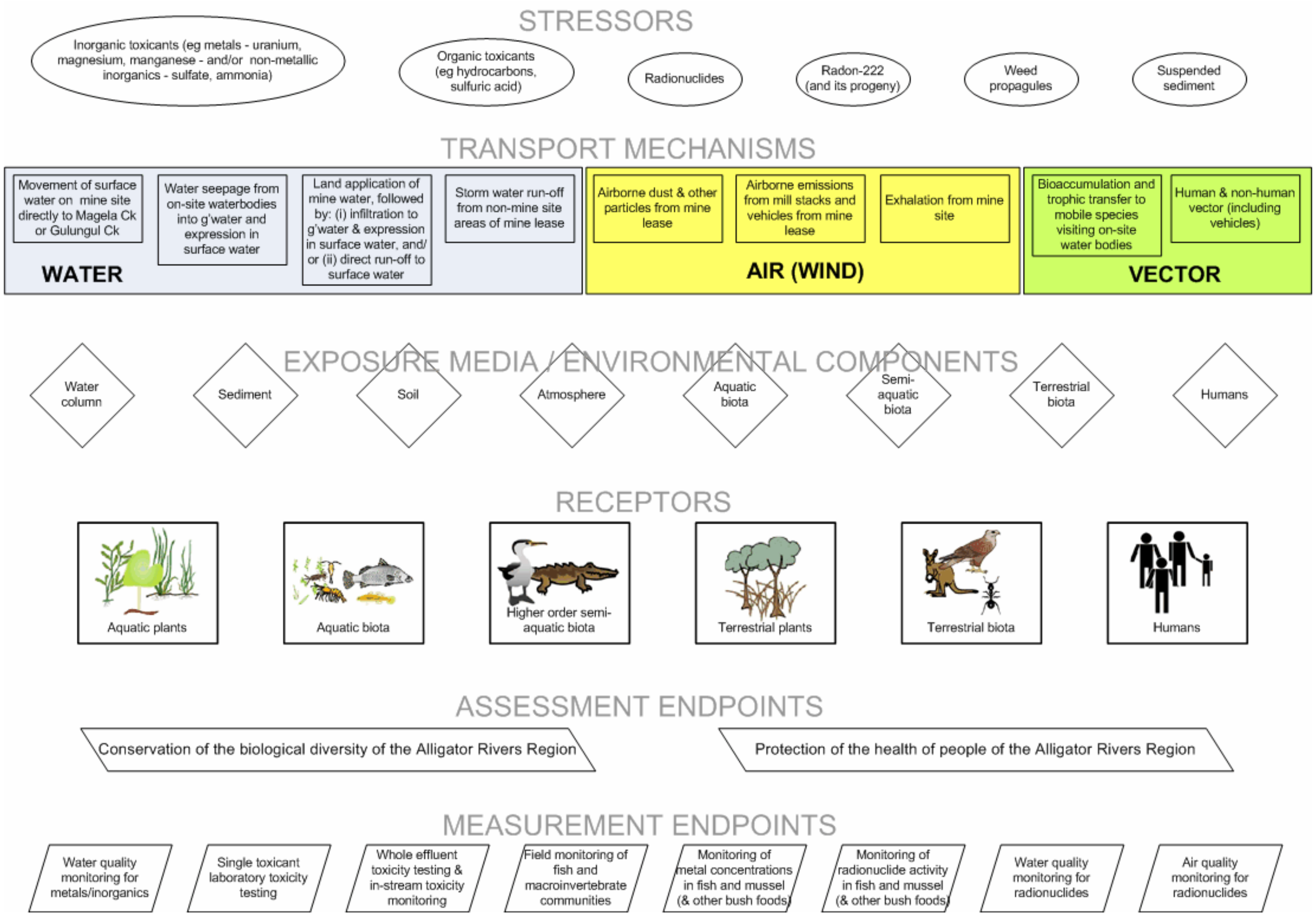


Figure 1 Basic elements of a contaminant pathways conceptual model for Ranger uranium mine

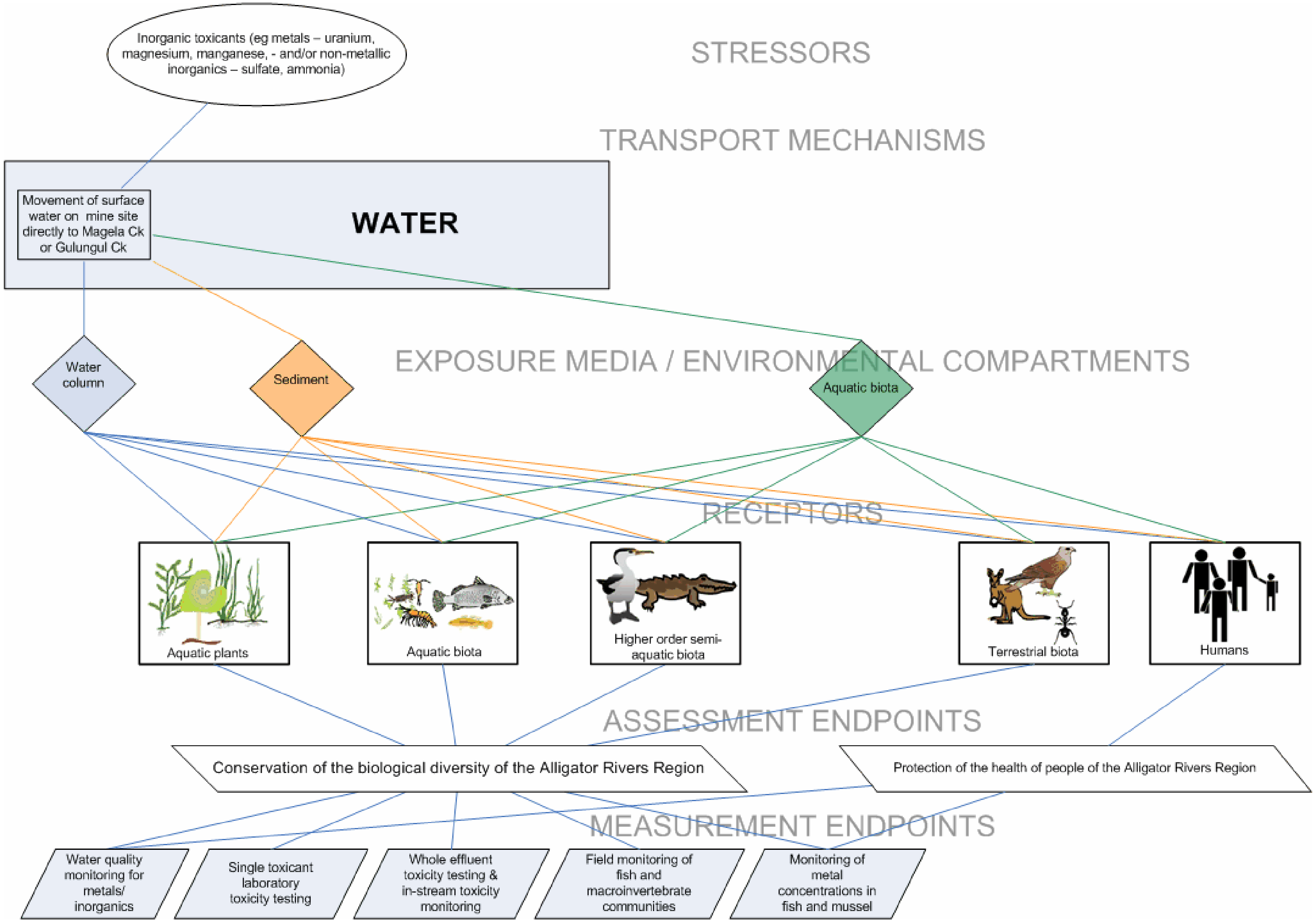


Figure 2 Conceptual model for transport of inorganic toxicants from Ranger uranium mine via a direct suface water to surface water pathway

Preliminary assessment of bioaccumulation and trophic transfer of key metals from Ranger mine

R van Dam, C Sauerland, K Turner, C Humphrey, B Ryan & A Bollhöfer

Background

Prior to the finalisation of the ARRTC Key Knowledge Needs (KKNs), a draft version of the KKN document (Draft 4.6.03) stated the following with respect to the uptake and trophic transfer of contaminants:

> ERISS has accumulated a considerable amount of ecotoxicological knowledge related to the effects of key contaminants on aquatic biota known to be present in the ARR. However, little of this information has been put into a management context, and has not been linked to the various biophysical pathways. For example, terrestrial environment & food sources have not been investigated. The possible transfer of contaminants such as uranium off site via the food chain should at least be assessed from a risk assessment perspective. What are possible pathways (e.g. waterbirds eating fish, sediment-dwelling invertebrates inhabiting RP1 and other close billabongs; terrestrial animals and birds eating plants and invertebrates inhabiting soil of land-application areas etc)?

Thus, in addition to the contaminant pathways conceptual model project summarised above, a project was initiated to review existing data to assess bioaccumulation and trophic transfer of metals associated with Ranger mine. The summary below focuses on the progress of this project during 2004–05 and presents a plan for its completion in 2005–06. The information summarised below was presented at the 8th International Conference on the Biogeochemistry of Trace Metals, in Adelaide, 3–7 April 2005 (van Dam et al 2005).

Progress and summary of findings to date

It should be noted that substantial bioaccumulation data for metals and radionuclides in aquatic and terrestrial biota have been collected over the past 20 years, albeit not necessarily as part of one program or a systematic series of programs. What was required for this project was a synthesis of the available data and an associated assessment of the ability to make conclusions about the ecological and human health risks of bioaccumulation and trophic transfer.

In planning the review and assessment, a 6 step process was developed:

1 Provide background on bioaccumulation and trophic transfer of metals and radionuclides;

2 Identify potential pathways and key metals for bioaccumulation and trophic transfer from mining at Ranger;

3 Collect and review available information (hard and electronic copy) from the ARR on concentrations/bioaccumulation/trophic transfer of key metals and radionuclides in aquatic (eg aquatic plants, invertebrates, fish), semi-aquatic (eg turtles, crocs, waterbirds) and terrestrial biota (eg lizards), and, if possible, relate to concentrations in the relevant environmental compartment(s) (eg water column, sediment, soil, vegetation/fruits);

4 Review national and international literature on bioaccumulation and trophic transfer (and subsequent effects) of metals and radionuclides identified from step 3 as being the most likely to bioaccumulate and/or biomagnify;

5 Where possible, identify ecological risks (and associated uncertainties) to the ARR of adverse effects arising from bioaccumulation/trophic transfer of key metals and radionuclides through comparison of exposure (steps 2 and 3) and effects (step 4); and

6 Identify information/research gaps.

Progress was made on all of the above steps, although the project was not completed (see below). The following key transport pathways were identified through the related project on the contaminant pathways conceptual model (see above):

- Direct movement/release of surface water on the mine site/lease directly to Magela Creek or Gulungul Creek (includes catchment runoff and release of retention pond waters);
- Seepage of water from on-site water bodies into groundwater and expression in surface water;
- Spray and flood irrigation of mine water to land, with subsequent infiltration to groundwater and expression in groundwater; and
- Biological uptake in mobile species inhabiting or visiting on-site water bodies.

The first three pathways represent the movement of on-site water to off-site natural surface waters and, ultimately, relate to the potential for bioaccumulation and biomagnification in biota exposed to highly diluted, relatively low concentrations of metals and radionuclides in the water column, or sediments of depositional zones. The fourth pathway is unique and relates primarily to the exposure of biota to waters that are moderately to highly contaminated, and the associated potential for metals and radionuclides to be accumulated as well as biomagnified and transported off-site via a biotic vector.

Metals of potential concern were identified based on historical assessments (eg Brown et al 1985, Noller 1991) and more recent information on ore and waste rock characteristics and process reagents. Given the pathways described above, it was clear that the two distinct exposure scenarios to consider for metal bioaccumulation/trophic transfer are those in:

- *off-site waterbodies:* the highly diluted surface waters of Magela Creek; and
- *on-site waterbodies:* the moderately to highly contaminated surface waters of the on-site water bodies at Ranger.

Moreover, given the high dilution of waste waters released to Magela Creek, it was clear that fewer metals will be of concern in the off-site waterbodies than the on-site waterbodies. Therefore, the list of metals of concern was screened, using two criteria, to identify key metals for this review and assessment, as summarised in table 1.

In total, over 100 published articles related to metals in the environment of the ARR have been located. Of these, approximately half are related to metal and/or radionuclide concentrations in biota. It needs to be emphasised that the vast majority of the data are yet to be screened for quality and used for the assessment. Data presented here are restricted to two studies only, one being the only study that has addressed the issue of trophic transfer of metals in on-site waterbodies (Batterham & Overall 1999) and another that addresses bioaccumulation of radionuclides and metals in the freshwater mussel (*Velesunio angasi*) downstream of Ranger (Ryan et al 2005).

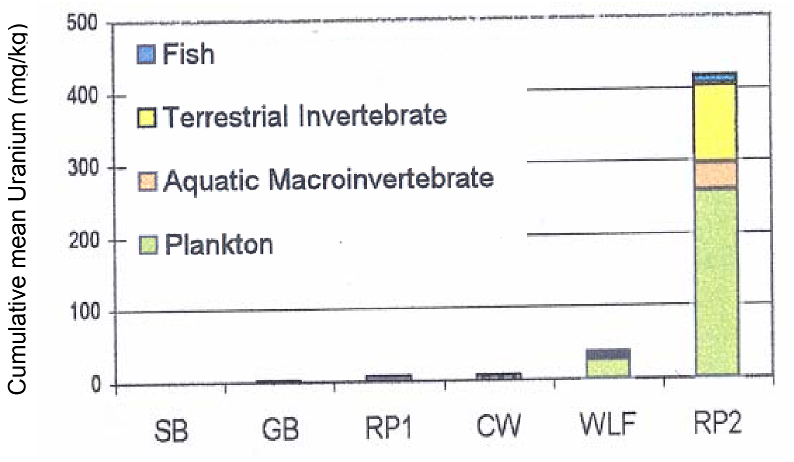
Table 1 Criteria for selecting key metals

Exposure scenario	Criteria for selection of key metals	Key metals
Exposure in ***on-site*** waterbodies (ie highly contaminated surface waters)	'Metals of concern' that have a moderate to high potential to bioaccumulate [1]	arsenic, barium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, selenium, silver, strontium, uranium, vanadium, zinc
Exposure in ***off-site*** downstream waterbodies (ie surface waters with highly diluted metal concentrations)	Key metals for *on-site* exposure scenario, for which concentrations in Magela Creek downstream of Ranger are elevated relative to concentrations upstream of Ranger [2]	manganese, uranium, zinc

1 *Information sources:* ATSDR Toxicological Profiles, IPCS Environmental Health Criteria + other specific references.

2 *Statistics:* Paired t-tests (one-tailed; α=0.05) + percentile approach.

Batterham and Overall (1999) sampled and measured tissue metal concentrations in four trophic groups from Ranger waterbodies representing a gradient of contamination. The results for uranium are shown in figure 1. In summary, bioaccumulation was evident at lower trophic levels (eg plankton), but not at higher trophic levels (eg fish), and this result was consistent among all the metals measured (data not shown). A reduction in tissue metal concentration with increasing trophic status is generally regarded as an indication that biomagnification is not occurring. This is not unexpected as there is very little evidence in the literature that inorganic metals biomagnify (Chapman et al 2003, McGeer et al 2004, Suedel et al 1994). However, there are numerous limitations associated with the work of Batterham and Overall (1999), including the fact that various key trophic pathways were not assessed (eg transfer of metals from aquatic flora and/or fauna to waterbirds and/or mammals). These pathways are important as they represent potential pathways for contaminants to be transported off the mine site.



<0.001 0.002 0.07 0.15 3.5

Dry season uranium (mg/L)

Figure 1 Cumulative mean uranium concentration (mg/kg) in four trophic groups sampled from five Ranger waterbodies (GB – Georgetown Billabong; RP1 – Retention Pond 1; CW – Corridor Creek wetlands; WLF – RP1 constructed wetland filter; RP2 – Retention Pond 2) and one Nourlangie billabong (SB – Sandy Billabong; reference site, unaffected by mining). Typical dry season surface water uranium concentrations (mg/L) for the waterbodies during the study are indicated in italics. From Batterham and Overall (1999).

Ryan et al (2005) reported radionuclide and metal concentrations in freshwater mussels collected from Mudginberri Billabong, downstream of Ranger, from 2000 to 2003, and Sandy

Billabong, in the nearby Nourlangie catchment, in 2002 and 2003. Notwithstanding a number of qualifications regarding the sampling methods and locations over the four year sampling period for Mudginberri Billabong (see Ryan et al 2005 for details), a summary of the data for uranium is shown in figure 2. For both 2002 and 2003, tissue uranium concentrations were higher in mussels from the 'exposed' (Mudginberri) site compared to mussels from the 'unexposed' (Sandy) site (two-sample T tests, $P < 0.05$), despite a limited water quality dataset revealing little difference in surface water uranium concentration between the two billabongs. However, the difference in mussel tissue uranium concentration between the two sites is relatively small (ie ~1.5×), and, given the low temporal resolution (ie. two years comparative data only), it is not possible to be conclusive until the many additional datasets are evaluated. For example, it is worth noting that the uranium concentrations in Mudginberri Billabong mussels reported by Ryan et al (2005) were similar, if not slightly lower, than those reported 15 to 20 years earlier by Johnston et al (1987). Thus, even if future monitoring data affirm the apparent difference between the two sites, there will be a need to establish whether the difference relates to a catchment signal or a mining signal. In contrast to the results for uranium, tissue manganese and zinc concentrations in mussels from Mudginberri Billabong were not significantly higher than in mussels from Sandy Billabong (data not shown).

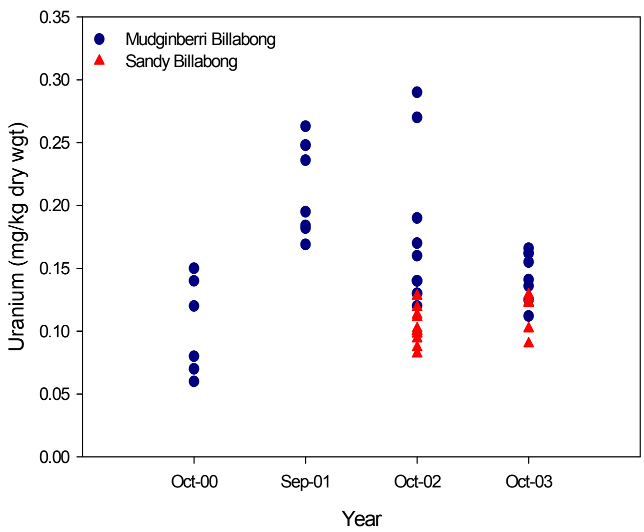


Figure 2 Tissue uranium concentrations (mg/kg dry weight) in freshwater mussels (*Velesunio angasi*) collected from Mudginberri Billabong from 2000 to 2003 and Sandy Billabong in 2002 and 2003. In general, each data point represents the uranium concentration for a composite sample of mussels from the same age (year) group. Data from Ryan et al (2005).

Preliminary conclusions

Although only a limited amount of data have been evaluated to date, a few preliminary conclusions can be made, as follows:

- The Ranger waterbodies trophic pathways study by Batterham and Overall (1999) and the majority of international literature on the issue of trophic transfer and biomagnification of metals, strongly indicate that, while trophic transfer of metals in aquatic environments does occur, biomagnification does not. Consequently, the transport of metals off site via a food chain pathway is unlikely to be a major transport mechanism.

- However, some data gaps exist for species at higher trophic levels, particularly herbivorous and piscivorous waterbirds;
- Although many data still remain to be analysed in relation to metals bioaccumulation in freshwater mussels (and other biota), there are indications of slightly elevated uranium concentrations in mussels downstream of Ranger relative to reference site mussels. However, there are uncertainties in the data, and there is a need for greater temporal resolution as well as an ability to establish whether any difference between the sites represents a catchment signal or a mining signal. In contrast, there is no evidence of elevated manganese or zinc in mussels downstream of Ranger.

Steps for completion

Due to high staff workloads and the need to better integrate the various related projects and/or reviews currently underway, this project has been subsumed into a broader program of works relating to the measurement and/or monitoring of radionuclides and metals in flora and fauna. The issue of trophic transfer, and off-site transport, of metals from on-site waterbodies will be addressed as part of the ongoing review and collation of data for the traditional bush foods project. This project has already developed and populated an Excel template and GIS for the multiple datasets for radionuclides (see summary for ARRTC KKN 2.5.1). The extent to which this is also done for key stable metals will depend on the level of priority assigned to the task as determined by the preliminary findings and discussions with ARRTC. At the very least, existing data of appropriate quality should be collated, with particular focus on data from on-site waterbodies. Further, in order to best answer the trophic transfer question raised by ARRTC (see *Background* to this project), the following steps could be undertaken:

1 Define/agree on 2–3 key trophic pathways for off-site transport, and within these, identify key trophic groups/species (animal and plant) to focus on species-specific data searching.

2 For the key trophic groups/species and key metals, collate and analyse existing data on tissue metal concentrations in biota on the mine area versus away from the mine (ie. exposed v reference), including consideration of associated water, sediment and/or soil concentrations.

3 Relate the above information to existing national/international literature on trophic transfer/ biomagnification of key metals (NB – much of this literature has already been summarised) including, if relevant, studies linking metal body burdens to adverse effects (more data searching is required here, although, overall, there is a general paucity of information on this topic).

4 Summarise site-specific ecological information on key species' feeding behaviour, feeding ground visitation, numbers potentially exposed/affected, etc.

5 Integrate the above information in a risk assessment context, including: ecological risks of individual key trophic pathways; relative risks of the key trophic pathways; and risks of the trophic pathways relative to other pathways (eg surface water). The latter is more an issue that will be addressed as the contaminant pathways conceptual model is populated.

The issue of bioaccumulation downstream of Ranger will be addressed as part of the ongoing (and currently under review) bioaccumulation monitoring program (see summary for ARRTC KKN 1.3.1).

References

Batterham R & Overall R 1999. Trophic pathways of heavy metals in Ranger water bodies. Internal Discussion Paper for ERA Ranger Mine. ERA Environmental Services Pty Ltd, Winnellie, NT.

Brown VM, Johnston A, Murray AS, Noller B & Riley GH 1985. Receiving water standards for the Alligator Rivers Region. In *Alligator Rivers Region Research Institute Annual Research Summary 1984–85*, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra, 111–119.

Chapman PM, Wang F, Janssen CR, Goulet RR & Kamunde CN 2003. Conducting ecological risk assessments of inorganic metals and metalloids: Current status. *Human and Ecological Risk Assessment* 9, 641–697.

Johnston A, Murray AS, Marten R, Martin P & Pettersson H 1987. *Bioaccumulation of radionuclides and stable metals in the freshwater mussel, Velesunio angasi*. Alligator Rivers Region Research Institute Annual Research Summary 1986–87, SSAR, AGPS, Canberra, ACT, 69–74.

McGeer J, Henningsen G, Lanno R, Fisher N, Sappington K & Drexler J 2004. *Issue paper on the bioavailability and bioaccumulation of metals*. Papers addressing scientific issues in the risk assessment of metals (submitted to the US Environmental Protection Agency). Eastern Research Group Inc, Lexington, MA.

Noller BN 1991. Non-radiological contaminants from uranium mining and milling at Ranger, Jabiru, Northern Territory, Australia. *Environmental Monitoring and Assessment* 19, 383–400.

Ryan B, Martin P, Humphrey C, Pidgeon R, Bollhöfer A, Fox T & Medley P 2005. Radionuclides and metals in fish and freshwater mussels from Mudginberri and Sandy Billabongs, Alligator Rivers Region, 2000–2003. Internal Report 498, November, Supervising Scientist, Darwin. Unpublished paper.

Suedel BC, Boraczek JA, Peddicord RK, Clifford PA & Dillon TM 1994. Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. *Reviews of Environmental Contamination and Toxicology* 136, 22–89.

van Dam R, Humphrey C, Sauerland C, Ryan B, Bollhöfer A and Wagner K 2005. A review and risk assessment of bioaccumulation and trophic transfer of key metals associated with mining in the Alligator Rivers Region, Northern Australia. In *8th international conference of ICOBTE*, Adelaide, 3–7 April 2005. Abstracts. Adelaide, CSIRO Land and Water, 590–600.

Influence of calcium on the ecotoxicity of magnesium: Implications for water quality trigger values

R van Dam, A Hogan, C McCullough[1], C Humphrey, S Nou[2] & M Douglas[3]

Background

Magnesium sulfate ($MgSO_4$), the dominant surface water contaminant associated with the ERA Ranger Mine, is generally considered to be of very low toxicity, Aquatic surveys around Ranger (O'Connor et al 1995) showed correlations between changes in macroinvertebrate community structure and increasing $MgSO_4$, prompting a full ecotoxicological investigation, including: identification of the dominant toxic ion; assessment of Mg toxicity in extremely soft local creek water (laboratory and field); and the influence of calcium (Ca) on Mg toxicity (laboratory), see Supervising Scientist 2002, 2003. Here, we present research quantifying the influence of Ca on Mg toxicity and the implications of this on a site-specific water quality trigger value for Mg. Although the majority of the work presented here was carried out during 2004–05, some additional data has been included to provide better context.

Progress and summary of findings to date

The three species (of six assessed) most sensitive to Mg in local Magela Creek water (duckweed, *Lemna aequinoctialis*; snail, *Amerianna cumingi*; green hydra, *Hydra viridissima*) were used to quantify the influence of Ca on Mg toxicity. The test species were exposed to a constant Mg concentration, being the concentration known to result in a 50% inhibition of response (eg population growth, reproduction) relative to unexposed (control) organisms, at increasing concentrations of Ca. Test results show that as Ca increases (ie as Mg:Ca decreases), Mg toxicity decreases for all three species (fig 1). For *Hydra* and *Lemna*, a full recovery was observed, whilst an approximate 80% recovery was observed for *Amerianna*. The differences in the extent of recovery between species is possibly due to different mechanisms of toxicity of Mg. Using the relationships from figure 1, a Mg:Ca ratio of 9:1 (ie that being the approximate 10% effect level for both *Hydra* and *Lemna*) was considered the best approximation of a ratio that minimises the likelihood of unacceptable Mg toxicity.

To complete the research, the toxicity of Mg was again fully characterised using six local freshwater species, but this time the Ca concentration was also manipulated such that the Mg:Ca ratio was maintained at the 'safe' ratio of 9:1. The summary results, compared to those obtained for Mg in the presence of only Magela Creek background Ca concentration (~0.2 mg/L), are shown in table 1. The IC_{15} values[4] presented represent maximum concentrations of Mg above which effects to the organisms might be considered ecologically unacceptable. For all species, Mg toxicity was markedly lower at the Mg:Ca ratio of 9:1.

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1 Edith Cowan University, Joondalup, WA

2 Formerly SSD; now Kakadu National Park, Parks Australia North, Department of the Environment & Heritage

3 Charles Darwin University, Darwin, NT.

4 See table 1 footnote.

However, the extent to which toxicity was reduced varied between species. For example, the cladoceran and snail were approximately only 2× less sensitive to Mg in the presence of Ca, whilst the gudgeon was well over 200× less sensitive. These differences in relative sensitivity are possibly due to different physiologies of the test species and different mechanisms of toxicity of Mg.

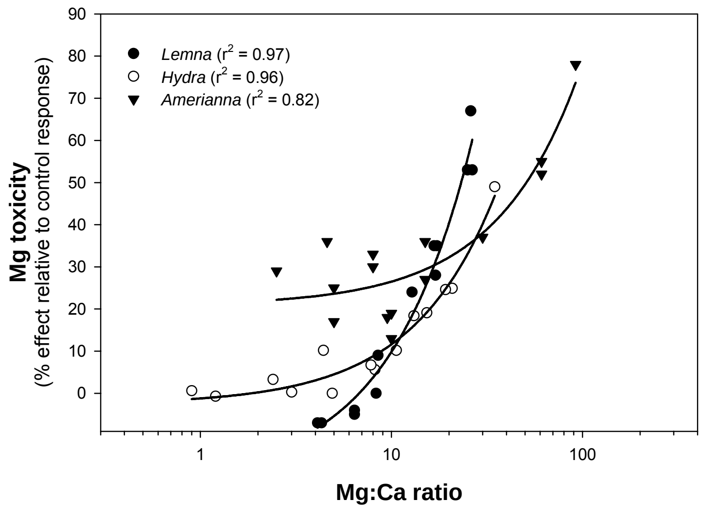

Figure 1 Effect of Ca (expressed as Mg:Ca ratio) on the toxicity of Mg to three species, when Mg concentration was held constant at the IC_{50} concentration (ie. 5 mg/L for *Lemna*, 10 mg/L for *Hydra*, 18 mg/L for *Amerianna*)

Table 1 Toxicity of Mg to six tropical freshwater species (a) without additional Ca and (b) with Ca added to maintain a Mg:Ca ratio of 9:1

		Mg toxicity (mg/L; expressed as IC_{15} values)[1]	
Species	**Endpoint (acute/chronic)**	**A. Without additional Ca[2]**	**B. At Mg:Ca ratio of 9:1[3]**
Green alga (*Chlorella* sp.)	72-h cell division rate (chronic)	93	1830[4]
Duckweed (*Lemna aequinoctialis*)	96-h plant growth (chronic)	2.2	155
Cladoceran (*Moindaphnia macleayi*)	3-brood reproduction (chronic)	24	37[4]
Snail (*Amerianna cumingi*)	96-h reproduction (chronic)	4.1	10
Green hydra (*Hydra viridissima*)	96-h population growth (chronic)	4.2	329
Purple-spotted gudgeon (*Mogurnda mogurnda*)	96-h survival (acute)	16[5]	4300[4,5]

1 IC_{15}: The concentration resulting in a 15% inhibition of response (eg reproduction) relative to unexposed (ie. control) organisms.

2 Except where noted, data for each species represent geometric means of IC_{15} values from 3 independent toxicity tests.

3 Except where noted, data for each species represent geometric means of IC_{15} values from 2 independent toxicity tests .

4 Data for each species represent values from 1 toxicity test. These data should be considered as interim only.

5 Data for *M. mogurnda* represent LC_5 values; the LC_5 being the concentration resulting in 5% mortality relative to unexposed (ie. control) organisms. This more conservative value is required because the endpoint for this test represents an acute, lethal response.

Using the BurrliOZ[5] species sensitivity distribution approach, a site-specific water quality trigger value for Mg in Magela Creek was derived. Given that Magela Creek lies largely within the World Heritage and Ramsar listed Kakadu National Park, a high level of protection is required. The derived water quality trigger value for Mg will protect at least 99% of species. Based on the Burr Type III distribution (black fitted line in figure 2), the concentration of Mg that should protect at least 99% of species is approximately 1 mg/L when Ca concentration is maintained at Magela Creek background Ca concentration, and approximately 4 mg/L[6] when Ca concentration is manipulated to maintain the Mg:Ca ratio at 9:1.

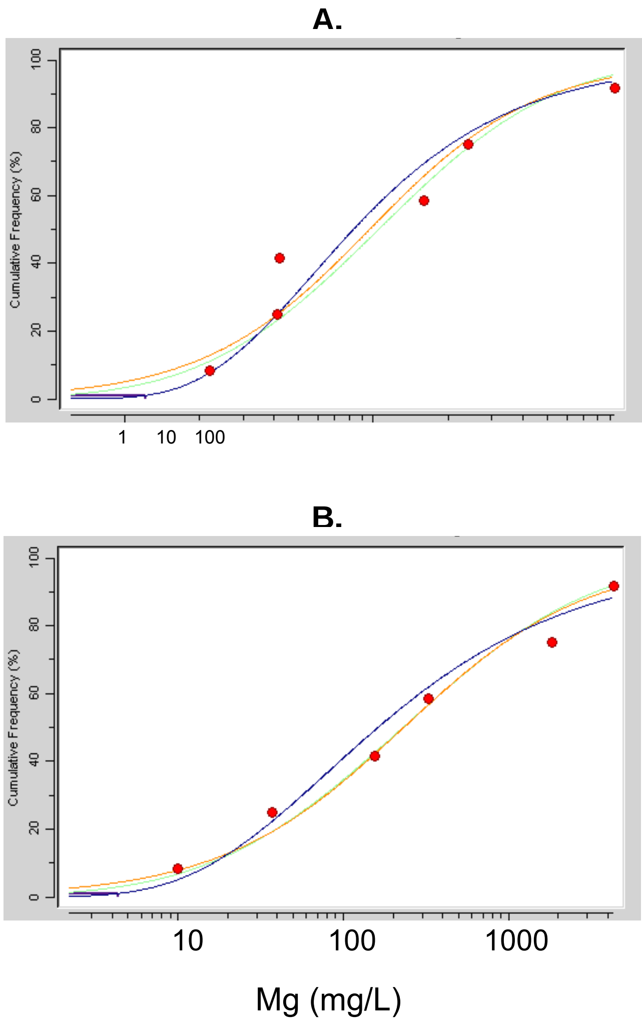

Figure 2 BurrliOZ species sensitivity distributions for Mg toxicity at (A) natural Magela Creek backgound Ca concentration (ie. ~0.2 mg/L) and (B) a constant Mg:Ca ratio of 9:1 (Note the different *x* axis scales). Data points represent the IC_{15} toxicity values for each species (see table 1) and are plotted as the cumulative frequency. The Burr Type III distribution is represented by the black fitted curve, and is the distribution used to calculate the trigger values for Mg (as recommended by ANZECC & ARMCANZ 2000).

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5 BurrliOZ is a statistical software package that was specifically developed for the ANZECC & ARMCANZ (2000) Water Quality Guidelines to calculate water quality trigger values.

6 This must be considered an interim value because the full toxicity data set is yet to be completed for several species (see table 1).

The above four-fold difference in the two trigger values is of significance to the management of discharged waste water at Ranger, because the discharged waters contain elevated Ca as well as elevated Mg. Thus, the Ca in Ranger waste waters is able to provide a protective function against potential Mg toxicity, and this needs to be taken into account when developing a site-specific trigger value. Therefore, if the Mg:Ca ratio in Magela Creek downstream of Ranger is maintained at or below 9:1, Mg concentrations of up to 4 mg/L should present very low risk to the local aquatic biota. To illustrate the low risk to aquatic biota to date, based on actual water quality data, figure 3 shows cumulative frequency distributions for Mg and the Mg:Ca ratio as measured at the monitoring point downstream of Ranger, from 1985 to 2005. The Mg trigger value has been exceeded only approximately 0.5% of the time, whilst the Mg:Ca ratio of 9:1 has not been exceeded. Thus, these data, although not finalised, indicate there is negligible risk to the aquatic biota of Magela Creek from Mg in surface water as a result of current mining operations. Had the Mg trigger value of ~1 mg/L, which does not account for the ameliorative effect of Ca on Mg toxicity, been applied, the risk to aquatic biota would have been significantly overestimated. Although yet to be finalised, the process of deriving the trigger value will almost certainly require consideration of both the Mg concentration and the Mg:Ca ratio.

Steps for completion

All that remains to complete this project is to finish several more Mg toxicity tests at the Mg:Ca ratio of 9:1. At the time of preparation of this summary, an additional test needed to be completed for *Chlorella* sp., *M. macleayi* and *M. mogurnda*. Following this, the Mg toxicity, and associated, research will be submitted for publication as one or two papers in an international peer-reviewed journal. Also, the results were presented at the bi-annual conference of the Australasian Society for Ecotoxicology, in Melbourne, 23–28 September 2005 (van Dam et al 2005).

References

ANZECC & ARMCANZ 2000. *Australian and New Zealand guidelines for fresh and marine water quality. National Water Quality Management Strategy Paper No 4*. Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

O'Connor R, Humphrey C, Dostine P, Lynch C & Spiers A 1996. A survey of aquatic macroinvertebrates in lentic waterbodies of Magela and Nourlangie Creek catchments, Alligator Rivers Region, NT. Internal report 225, Supervising Scientist, Canberra. Unpublished paper.

Supervising Scientist 2002. *Annual report 2001–2002*. Supervising Scientist, Darwin NT.

Supervising Scientist 2003. *Annual report 2002–2003*. Supervising Scientist, Darwin NT.

van Dam R, McCullough C, Hogan A, Humphrey C, Nou S & Douglas M 2005. How toxic is magnesium sulfate? Implications for the mining industry. Paper presented at Biennial Conference of the Australasian Society for Ecotoxicology, Melbourne, Victoria, 24–28 September.

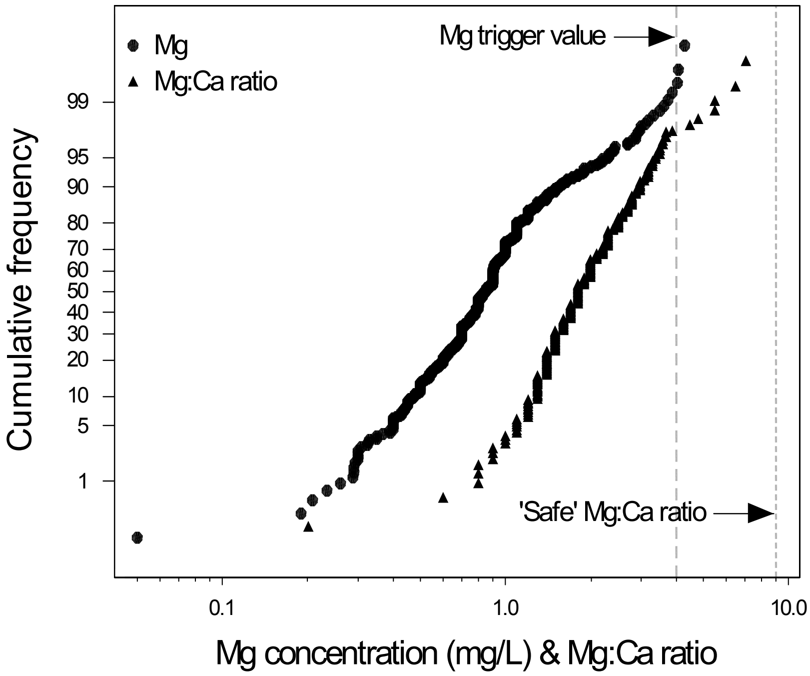

Figure 3 Cumulative probability distributions for Mg concentration and Mg:Ca ratio in Magela Creek downstream of Ranger, from 1985–2005. The vertical broken lines represent, from left to right, the Mg trigger value of 4 mg/L (when the Mg:Ca ratio is maintained at or below 9:1) and the 'safe' Mg:Ca ratio of 9:1

Chronic toxicity of uranium to the tropical duckweed, *Lemna aequinoctialis*

R van Dam, S Nou[1] & A Hogan

Background

As uranium is one of the primary toxicants associated with the water released from Ranger, a receiving water Trigger Value (TV) has been derived specifically for Magela Creek according to the ANZECC and ARMCANZ Guidelines for Fresh and Marine Water Quality (ANZECC & ARMCANZ 2000). The site-specific TV of 6 µg/L is classified as being of 'high reliability' due to the use for its derivation of chronic toxicity data from five local species representing four taxonomic groups (see Hogan et al 2005). Although not detailed here, there are several reasons to support the need to further strengthen the uranium chronic toxicity dataset. However, the worth of generating uranium chronic toxicity data for additional species must be weighed against the normally large effort and cost required to develop new toxicity tests. The recent refinement of a growth inhibition test using the tropical duckweed, *Lemna aequinoctialis,* and the development of a reproduction test using the pulmonate gastropod, *Amerianna cumingi*, provided an opportunity to increase the chronic toxicity dataset for uranium with minimal additional developmental effort and cost. When data for these species are added to the existing suite of species that have been assessed, uranium toxicity data will exist for seven species from six taxonomic groups, a marked improvement on the current dataset. The site-specific uranium toxicity tests using the *L. aequinoctialis* growth inhibition test are summarised in this report.

Aims

The aims of this study were to (i) determine the minimum concentrations of nutrients (nitrate and phosphate) required to be added to natural Magela Creek water (NMCW) to ensure acceptable growth of *L. aequinoctialis* in control treatments, and, using the revised protocol, (ii) assess the chronic toxicity of uranium to *L. aequinoctialis* in NMCW.

Methods

The *L. aequinoctialis* 96 h growth inhibition toxicity test procedure is described in full by Riethmuller et al (2003). For this study, two series of experiments were carried out.

Nutrient optimisation tests: Two experiments were undertaken to determine the minimum concentrations of nitrate (NO_3) and phosphate (PO_4) that need to be added to NMCW in order to obtain acceptable *L. aequinoctialis* growth (ie four-fold increase in initial frond numbers after 96 h). The nutrients were added to each treatment at a NO_3:PO_4 ratio of 10:1 (N:P ≈7:1) at the following concentrations (NO_3/PO_4):

Test 1: 0/0, 0.1/0.01, 0.3/0.03, 1/0.1, 3/0.3, 10/1, 30/3 and 100/10 mg/L

Test 2: 0/0, 0.3/0.03, 1/0.1, 3/0.3, 10/1 and 30/3 mg/L

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[1] Kakadu National Park, Parks Australia North, Department of the Environment and Heritage

One-way analysis of variance (ANOVA) followed by a Fisher's pairwise comparison was used to identify significant differences between treatments ($\alpha = 0.05$).

Uranium chronic toxicity tests: One rangefinder and three definitive experiments were undertaken to characterise the effect of uranium exposure on the growth of *L. aequinoctialis* in NMCW. Concentrations of uranium assessed in each of the three definitive tests were:

Test 1: 0, 50, 100, 250, 500, 750, 1000 µg/L

Test 2: 0, 250, 500, 750, 1000, 1500, 2000, 3000 µg/L

Test 3: 0, 50, 100, 250, 500, 750, 1000, 1500 µg/L

Analysis of covariance (ANCOVA) was used to determine whether the concentration-response relationships for the three definitive uranium toxicity tests were significantly different ($\alpha = 0.05$), and to determine the validity of pooling the datasets for further statistical analysis. If valid, the data were pooled and a one-tailed Bonferoni T-test was used to determine the Lowest-Observed-Effect-Concentration (LOEC) and the No-Observed-Effect-Concentration (NOEC) of uranium ($\alpha = 0.05$). In addition, non-linear regression was used to characterise the concentration-response relationship and to calculate inhibition concentrations (IC_x) of uranium corresponding to a 10, 25 and 50% inhibition of response compared to the control response (ie. IC_{10}, IC_{25} and IC_{50}).

Results and discussion

Nutrient optimisation tests: The minimum NO_3/PO_4 combination at which plant growth exceeded the control acceptability criterion of a four-fold increase in frond number was 3 mg/L NO_3 and 0.3 mg/L PO_4 (equivalent to 0.7 mg/L N and 0.1 mg/L P) (figure 1). Plant growth at this NO_3/PO_4 combination was not significantly different to growth at any of the higher concentrations ($P > 0.05$). At lower concentrations, plant growth was significantly lower ($P < 0.05$), and well below the minimum acceptable control criterion. These nutrient concentrations were selected for addition to NMCW for the uranium chronic toxicity tests.

Uranium chronic toxicity tests: ANCOVA indicated that the concentration–response relationships for each of the three definitive tests were not significantly different (F = 0.895, df = 2, $P = 0.430$), and as such, the data were pooled for further analysis and interpretation. The resultant concentration–response relationship is shown in figure 2. Based on this dataset, the LOEC and NOEC were 247 µg/L and 216 µg/L, respectively. At the LOEC concentration, there was an approximate 20% reduction in plant growth relative to controls. Plant growth decreased to around 30–40% of the control up to a concentration of around 700 µg/L, but plateaued thereafter. The concentration–response relationship was best described by a 4 parameter logistic distribution ($r^2 = 0.97$, n = 19, $P < 0.001$; see figure 2). Using this model, the IC_{10}, IC_{25} and IC_{50} (95% confidence intervals) were calculated to be 187 (130–228) µg/L, 291 (256–325) µg/L and 504 (449–572) µg/L, respectively.

L. aequinoctialis exhibited similar sensitivity to uranium as the green hydra, *Hydra viridissima* (NOEC = 183 µg/L; Hogan et al 2005). It is noteworthy that plant growth, albeit around 3 fold lower than in controls, was still observed at uranium concentrations in excess of 1000 µg/L. In comparison, complete inhibition of population growth (and death) of *H. viridissima* occurs at only 250–300 µg uranium/L (ARRRI 1988). Thus, *L. aequinoctialis* is able to maintain some growth at concentrations of uranium that are lethal to other species that have been assessed. A similar ability was reported for *L. aequinoctialis* exposed to the herbicide tebuthiuron, in this instance in comparison to the response of the green alga, *Chlorella* sp. (van Dam et al 2004).

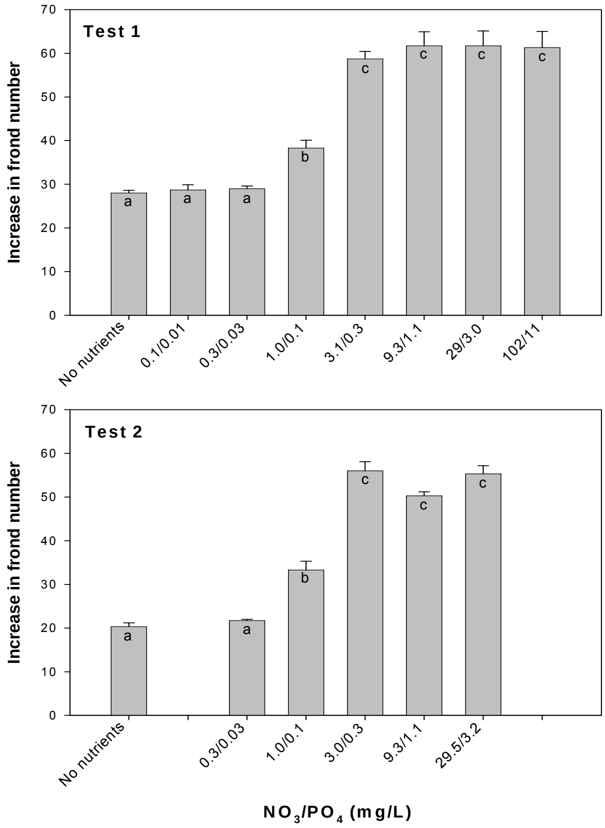


Figure 1 Effect of various combinations of nitrate (NO_3) and phosphate (PO_4) concentrations (measured) on growth of *L. aequinoctialis* over 96 h. Results for each treatment are expressed as the mean (±SEM) of three replicates. Treatments with a letter in common are not significantly different from each other (P>0.05).0

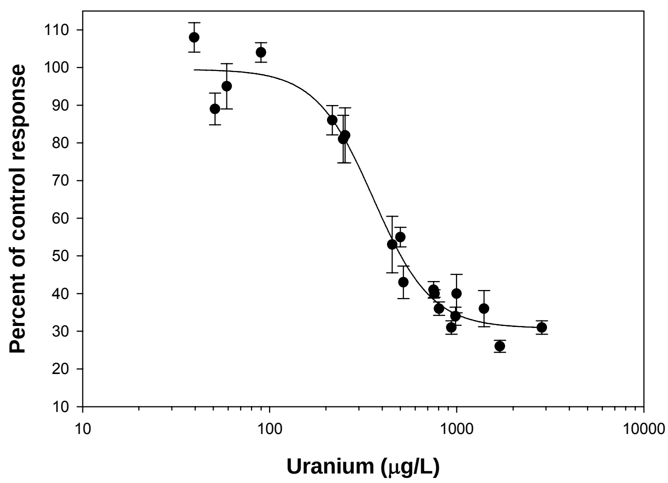


Figure 2 Response of *L. aequinoctialis* growth to uranium (measured) based on the pooled dataset and relative to the pooled control response of 55.4 ± 1.4 fronds (mean ± SEM; n = 9). The fitted curve represents a 4 parameter logistic model (r^2 = 0.97, n = 19, P < 0.0001).

If the *L. aequinoctialis* NOEC of 216 µg/L (or IC_{10} of 187 µg/L) is added to the existing uranium chronic toxicity dataset, the resultant 99% protection trigger value for uranium increases from 6 µg/L to around 9 µg/L (based on a log-logistic distribution). Not surprisingly, given the small sample size of six, the uncertainty remains high, with the 95% confidence interval being around 1–90 µg/L. A revised trigger value will not be formally published until the uranium toxicity experiments for the snail, *A. cumingi*, have been completed.

Steps for completion

An internal report of the *L. aequinoctialis* testing is almost complete. This research, and that to be completed in 2005-06 for the snail, will be combined and submitted for publication in an international peer-reviewed journal, most likely in 2006.

References

ANZECC & ARMCANZ 2000. *Australian and New Zealand guidelines for fresh and marine water quality. National Water Quality Management Strategy Paper No 4*. Australian and New Zealand Environment and Conservation Council & Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

ARRRI 1988. *Alligator Rivers Region Research Institute: Annual research summary 1987–1988*. Supervising Scientist for the Alligator Rivers Region, Australian Government Publishing Service, Canberra.

Hogan AC, van Dam RA, Markich SJ & Camilleri C 2005. Chronic toxicity of uranium to a tropical green alga (*Chlorella* sp.) in natural waters and the influence of dissolved organic carbon. *Aquatic Toxicology* 75, 343–353.

Riethmuller N, Camilleri C, Franklin N, Hogan AC, King A, Koch A, Markich SJ, Turley C & van Dam R 2003. *Ecotoxicological testing protocols for Australian tropical freshwater ecosystems*. Supervising Scientist Report 173, Supervising Scientist, Darwin NT.

van Dam RA, Camilleri C, Turley C, Binet M & Stauber J 2004. Chronic toxicity of the herbicide tebuthiuron to the tropical green alga *Chlorella* sp. and the duckweed *Lemna aequinoctialis*. *Australasian Journal of Ecotoxicology* 10, 97–104.

Development of a reference toxicity testing program for *Chlorella* sp. and *Hydra viridissima*

A Hogan, R van Dam & S Nou[1]

Background

The ***eriss*** ecotoxicology laboratory generally has good procedures in place to ensure quality of results. However, one key gap in the laboratory's QA/QC program is a regular reference toxicity testing program for the routine toxicity testing species. This gap was identified by a review of the ecotoxicology research program by van Dam (2004) and also by Dr Jenny Stauber at ARRTC's 14th meeting. Consequently, it was agreed by ARRTC that a formal project be initiated to develop a reference toxicity testing program. This project was initiated in 2004, with the focus being on the green alga, *Chlorella* sp, and the green hydra, *Hydra viridissima*, and is proposed to be continued in 2005. The reference toxicant being used is uranium. This summary outlines the progress made during 2004–05.

Progress

Two reference toxicity tests (using synthetic Magela Creek water – SMCW) for *Chlorella* sp and three for *H. viridissima* were completed in 2004–05, as scheduled. Given the existence of four and three reference toxicity test results for *H. viridissima* and *Chlorella* sp, respectively, the 04–05 testing enabled the construction of reference toxicant control charts for each species (given that a minimum of five test results are required in order to construct a control chart; Environment Canada 1990). The relevant uranium toxicity data are shown in table 1. The controls charts are shown in figure 1.

For *Chlorella* sp, the running mean IC_{50} is currently around 65 µg U/L. To date, no test results have deviated beyond the *warning limits* (ie two standard deviations above or below the running mean), indicating a consistent experimental technique and response of the test species to uranium over time. However, the last two reference toxicity tests were invalid due to unacceptably low control growth. This response has not been observed in recent *Chlorella* sp. tests run in natural Magela Creek natural water (NMCW). Measures being undertaken to investigate the cause of the response include: sending the algal culture to CSIRO (Sydney) for treatment to return it to an axenic state; sending samples of SMCW stock solutions for chemical analysis, and renewing nutrient stock solutions. The success of these measures will be known in September.

For *H. viridissima*, the running mean IC_{50} is currently around 103 µg U/L. To date, no test results have deviated beyond the warning limits, indicating a consistent experimental technique and response of the test species to uranium over time.

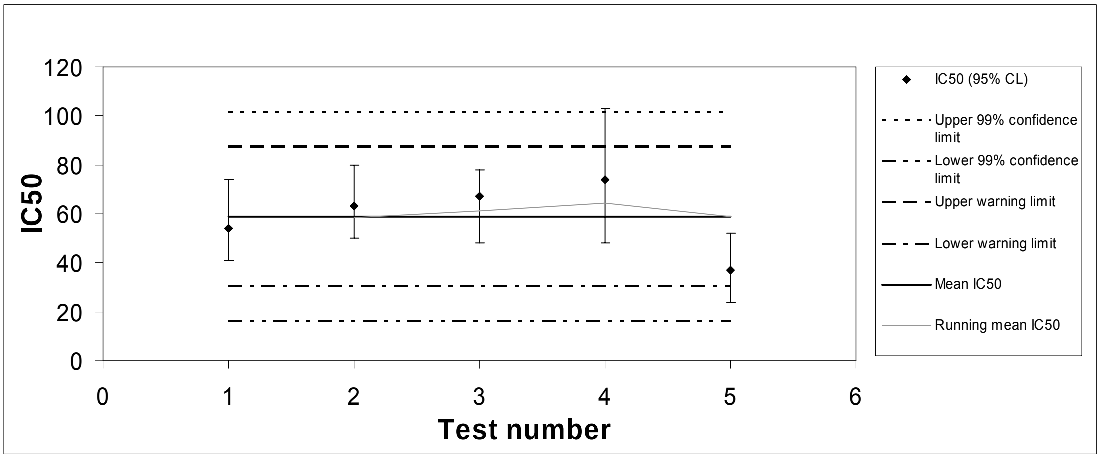
[1] Formerly SSD; now Kakadu National Park, Parks Australia North, Department of the Environment & Heritage

Table 1 Uranium reference toxicity test results for *chlorella* sp. and *hydra viridissima*

Species	Endpoint	Test No.	IC_{50} (µg/L)	Reference
Green alga (*Chlorella* sp)	72-h cell division rate	1	54 (41–74)[1]	Franklin et al (1998)
		2	63 (50–80)	Franklin et al (1998)
		3	67 (48–78)	Franklin et al (1998)
		4	74 (48–103)	Hogan et al (2005)
		5	37 (24–52)	This study
		6	Invalid test[2]	This study
		7	Invalid test[2]	This study
Green hydra (*Hydra viridissima*)	96-h population growth	1	133 (99–158)	Riethmuller et al (2000)
		2	108 (100–142)	Riethmuller et al (2000)
		3	117 (CL not calculable)	Riethmuller et al (2000)
		4	85 (68–93)	This study
		5	80 (67–94)	This study
		6	94 (78–113)	This study

1 Values in parentheses represent 95% confidence intervals.

2 Invalid *Chlorella* sp. tests were due to poor control growth. See main text for discussion and steps to rectify this issue.



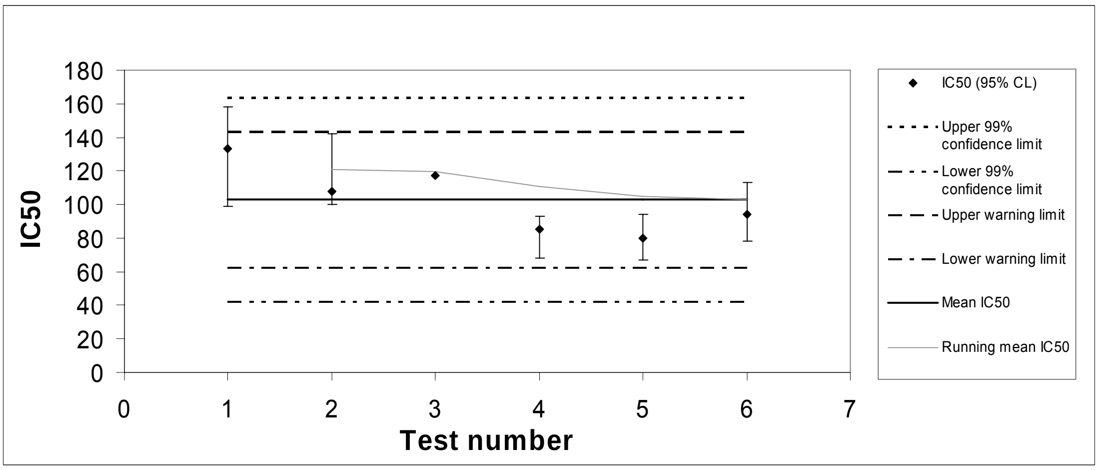


Figure 1 Control charts showing reference toxicity test data for (top) *Chlorella* sp and (bottom) *Hydra viridissima*

Steps for completion

The reference toxicity testing program is an ongoing program. Hence, it is expected that approximately four reference toxicity tests for both the above two species will again be completed in 2005–06. Further, 2005–06 will focus on the establishment of reference toxicity testing programs for the duckweed, *Lemna aequinoctialis*, and the purple-spotted gudgeon, *Mogurnda mogurnda*.

References

Environment Canada 1990. *Guidance document on control of toxicity test precision using reference toxicants.* Environmental Protection Series 1/RM/12, Minister of Supply and Services Canada, Ottawa, Ontario, Canada.

Franklin N, Stauber J, Markich S & Lim R 1998. *A new tropical algal test to assess the toxicity of metals in freshwaters*. Supervising Scientist Report 133, Supervising Scientist, Canberra.

Hogan AC, van Dam RA, Markich SJ & Camilleri C 2005. Chronic toxicity of uranium to the tropical green alga *Chlorella* sp. in natural waters and the influence of organic carbon. *Aquatic Toxicology* 75, 343–353.

Riethmuller N, Markich S, Parry D & van Dam R 2000. *The effect of true water hardness and alkalinity on the toxicity of Cu and U to two tropical Australian freshwater organisms*. Supervising Scientist Report 155, Supervising Scientist, Canberra.

van Dam R 2004. *A review of the **eriss** Ecotoxicology Program*. Supervising Scientist Report 182, Supervising Scientist, Darwin NT.

Revision of the ecotoxicology laboratory manual

A Hogan

Background

A comprehensive manual of all routine laboratory procedures has been a central reference document in the ecotoxicology laboratory since the late 1980s. While being designed as a dynamic document that undergoes regular minor updates, the relocation of the laboratory to Darwin required more significant amendments. This was also recognised as an opportunity to review the order, style and format of the manual to make it more user-friendly and up to date.

Progress on the laboratory manual revision was slow for the first two years after the relocation, with staff only managing to spend a small amount of time on it between projects. The benefits of being able to devote solid blocks of time to this work was recognised and the revision was made a formal project on the 2004–2005 workplan.

Progress

Major revisions of all the original chapters have been completed. A chapter on the new aquaculture water system is at a final draft stage. A new chapter on *Lemna* maintenance and small sections on the new methods for taking water chemistry samples and care and maintenance of the incubators still need to be written. Some graphics still need to be added and pasted in and more photographs throughout the document would be useful. The order of chapters is still to be assessed to improve the flow of the document and final formatting will need to be undertaken.

Steps for completion

Laboratory staff will complete the water chemistry, incubator and aquaculture water system sections and insert final graphics during October. All ecotoxicology staff will then meet to identify any information gaps and to decide on the final order of chapters. Laboratory staff will then check for continuity between chapters and standardise the formatting throughout the document (possibly with assistance from our publications officer).

Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Bollhöfer, D Elphick[1] & SA Atkins

Introduction

The International Commission on Radiation Protection (ICRP 1991) recommends that the annual dose received from practices such as uranium mining and milling should not exceed 1 milli Sievert (mSv) per year. The ICRP furthermore states in paragraph 6.2.1 of Publication 77 (ICRP 1997) that 'To allow for exposures to multiple sources, the maximum value of the constraint used in the optimisation of protection for a single source should be less than 1 mSv in a year. A value of no more than about 0.3 mSv in a year would be appropriate.' This dose is on top of the radiation dose received naturally, which averages to approximately 2 mSv per year in Australia, but typically varies between 1–10 mSv per year (UNSCEAR 2000).

Ranger is the main potential anthropogenic source of radiation exposure to the community in the Alligator Rivers Region. During the operational phase of a uranium mine there are two potential exposure pathways to the general public. The inhalation pathway, which is a result of dispersion of radionuclides from the mine site into the air, and the ingestion pathway, which is caused by the uptake of radionuclides into bushfoods from the Magela Creek system downstream of Ranger.

Methods

The Supervising Scientist monitors the two airborne pathways:

- Radioactivity trapped in or on dust (or long lived alpha activity, LLAA)
- Radon decay products (RDP).

Dust samples are collected monthly on 47 mm glass fibre filters for periods of 7–10 days, using a solar powered low volume dust sampler. After sample collection and allowing for the decay of short lived radionuclides (usually 5–7 days), LLAA is measured on a *Daybreak* gross alpha counter, calibrated using a certified ^{230}Th source.

Radon decay products are measured using a portable continuous radon and thoron working level monitoring system. This system pumps air through a filter and measures the subsequent decay of RDP collected on the filter with a built-in gross alpha detector counting all alpha energies above 1 MeV. The system is deployed for 24 continuous hours per month and data are downloaded from the unit's memory after a 3.5 hour decay analysis subsequent to the sampling cycle.

Results

The main areas of habitation are Jabiru, Mudginberri and Jabiru East, consequently monitoring focuses on those three population centres in the region. Airborne RDP and LLAA concentrations are measured monthly and the results are compared with Energy Resources of Australia's (ERA) quarterly and annual atmospheric monitoring results.

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[1] Formerly SSD; now Fisheries Division, NT Department of Primary Industry, Fisheries and Mines

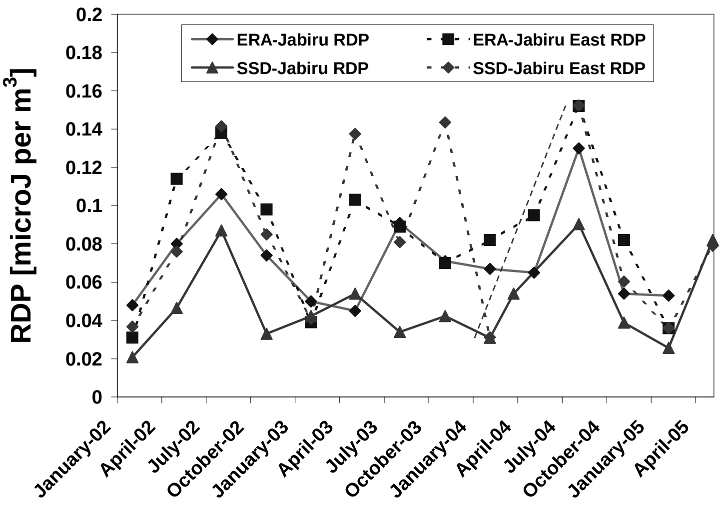

Figure 1 Radon decay product concentration measured by SSD and ERA in Jabiru and Jabiru East from 2002 to 2005

Figure 1 shows Jabiru and Jabiru East RDP data and a comparison with ERA data from January 2002 to April 2005. Differences in sampling time and location are most likely the cause of the differences in RDP concentrations observed at Jabiru, with ERA values being slightly higher than values measured by ***eriss***. The annual exposure due to the inhalation of radioactivity trapped in or on dust for people working in Jabiru East and living in Jabiru has been shown to be trivial and is less than 1% of the public dose limit (Bollhöfer et al 2005).

Table 1 shows the average annual doses received from the inhalation of RDP calculated from ***eriss*** and ERA (ERA, 2005) data at Jabiru. This is assuming an occupancy of 8760 hrs (1 year) and a dose conversion factor for the public of 0.0011mSv per µJ·h/m^3. Mine derived annual doses from the inhalation of radon progeny are shown as well, as calculated by ERA using a wind correlation model developed by ***eriss***, which correlates wind direction with airborne radon decay product concentration.

Table 1 Average annual RDP concentration measured by ***eriss*** and ERA (*italics*) at Jabiru East and Jabiru and doses received from the inhalation of RDP at Jabiru

		2002	2003	2004
RDP concentration [µJ/m^3]	Jabiru East	0.085; *0.095*	0.101; *0.075*	0.095; *0.103*
	Jabiru	0.047; *0.077*	0.043; *0.065*	0.063; *0.079*
Total annual dose [mSv] Jabiru		0.45; *0.74*	0.41; *0.63*	0.61; *0.76*
Mine derived dose [mSv] at Jabiru		*0.03*	*0.011*	*0.014*

Although there were no activities reported at the Jabiluka mine site, the population group that may in theory receive a radiation dose due to future activities at Jabiluka is the 60 or so inhabitants of Mudginberri, a small community approximately 10 km south of Jabiluka. At Four Gates Rd radon station, a few kilometres west of Mudginberri, the Supervising Scientist has a permanent atmospheric research and monitoring station. RDP and LLAA concentrations are measured there on a monthly basis. In addition, radon gas is continuously measured at the station with radon data being recorded every 30 minutes.

Figure 2 shows airborne RDP and LLAA concentrations measured at Four Gates Rd radon station by ***eriss***. Concentrations are very small and comparable with natural background levels (UNSCEAR 1993). The average airborne radionuclide concentrations measured in 2004 would translate into an annual total effective dose, including natural background, of 0.46 mSv from RDP and 0.013 mSv from LLAA. Only a small fraction of these doses would be due to mine derived radionuclides.

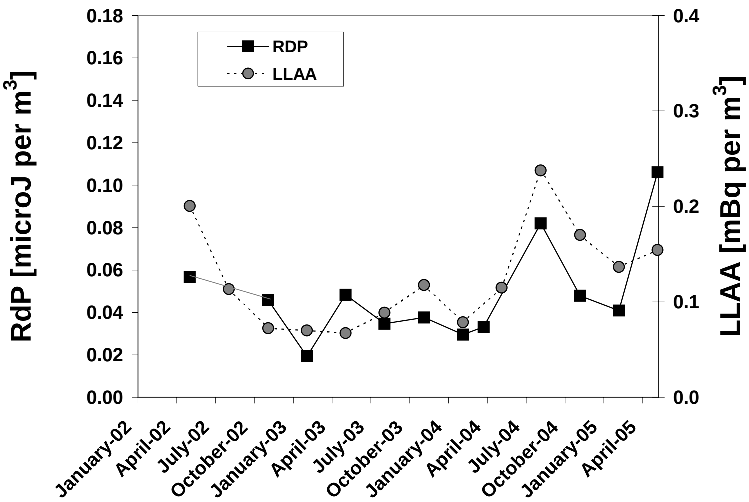

Figure 2 Radon decay product (RDP) and long lived alpha activity (LLAA) concentrations measured at ***eriss***'s Mudginberri Four Gates Rd radon station

References

Bollhöfer A, Honeybun R, Rosman K & Martin P 2005. The lead isotopic composition of dust in the vicinity of a uranium mine in northern Australia and its use for radiation dose assessment. *The Science of the Total Environment.* In press.

ERA 2005. Radiation Protection and Atmospheric Monitoring program. In *Annual Report for the year ending 31 December 2004*. Energy Resources of Australia Ltd, Ranger Mine.

ICRP 1991. *1990 Recommendations of the International Commission on Radiological Protection*. Publication 60 of the International Commission on Radiological Protection. Pergamon Press, Oxford.

ICRP 1997. *Radiological protection policy for the disposal of radioactive waste*. Publication 77 of the International Commission on Radiological Protection. Pergamon Press, Oxford.

UNSCEAR 2000. *Sources and effects of ionizing radiation*. Report to the General Assembly, with scientific annexes, United Nations Scientific Committee on the Effects of Atomic Radiation, New York.

Monitoring of radionuclides in groundwater at Ranger

B Ryan

Introduction

The aim of the Ranger groundwater program is to investigate the dispersion of contaminants through the groundwater pathway, both from a monitoring and hydrogeological modelling perspective.

Aquifers, both shallow and deep, and sources of contaminants need to be characterised, both geochemically and hydrogeologically, to enable the prediction of the dispersion of these contaminants. The monitoring program will also have to be continued during and following the rehabilitation of the mine to assess the success of the rehabilitation and the integrity of the pits as tailings repositories.

Progress to date

RUM bore water samples were collected by Department of Business, Industry and Resource Development for ***eriss*** in 2004 and aliquots prepared for radioisotope analysis. ICPMS-OES analysis of all archived groundwater samples were 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 began in late 2004 and are not yet complete. During the 2004/05 financial year emphasis was given on the Nabarlek groundwater samples that had been collected and archived from 1996–2004.

Analyses and interpretation of the long-term groundwater data for uranium isotopes and radium will be undertaken in 2005–06 after all samples have been analysed, with particular attention being placed on assessing groundwater movement and any sources of contamination.

Radon concentrations in air in the Alligator Rivers Region

A Bollhöfer & P Martin[1]

Introduction

The radon network in the Alligator Rivers Region (ARR) was designed to provide detailed time-series radon concentration and meteorological data at various locations within the ARR over a time frame of several years:

- to provide information on the influence of past and present uranium mining and milling activities on radon concentrations in air in the ARR;
- to establish a baseline database to assist in assessment of any future mining activities;
- to provide datasets useful in assessing the effects of various factors on radon transport in the atmosphere. In particular to calibrate and verify radon transport models.

Four radon stations were used earlier in this project, with three stations placed throughout the ARR and on the Ranger mine lease, respectively, collecting data for periods of one year. One station is collecting data permanently at Four Gates Road radon station in the vicinity of Mudginberri as a control and monitoring site. The radon monitor at this site has now been collecting data for more than five years.

Methods

The sampling equipment used is described in detail in Tims (2001), Bollhöfer et al (2004) and Martin et al (2004). Briefly, air is drawn through a delay line of ~15 litres at a flow rate of approximately 5 litres per minute to remove thoron and then passes through a class P2 particulate filter to remove radon and thoron decay products. It then enters the tank with a volume of 100 litres where radon decays and radon progeny are circulated through and collected on a second filter mounted approximately 7 mm from a ZnS coated light guide, attached to a photomultiplier (PM) tube. The PM tube detects the light pulses generated in the scintillating ZnS coating by the α-particles emitted by the radon progeny. Pulses are fed into a discriminator and then directly into a data logger. Data are read into the long-term memory of a data logger every 30 minutes. The logger also records the internal tank pressure, the flow rate, the internal and external blower fan currents, the PM tube voltage, the internal battery voltages and the temperature. Data loggers are download approximately every two months.

Results

The Mudginberri Four Gates Road radon station radon dataset has been updated up to January 2005. A comprehensive QA/QC on the dataset to allow journal publication, is being conducted. A paper on the radon network in the ARR (Martin et al 2004) and an internal report on Nabarlek radon concentrations (Bollhöfer et al 2004) have been published.

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[1] Formerly SSD; now Agency's Laboratories Seibersdorf, IAEA, A-1400, Vienna, Austria.

In figure 1 the daily averages of radon concentration measured at the Mudginberri Four Gates Rd radon station are plotted versus the date throughout the last 5 years, together with a weekly running average to the data. The data show the typical seasonal variations with higher radon concentrations measured during the dry season and lower values during the wet. The long-term average radon concentration at Mudginberri Four Gates Road radon station is 10 Bq·m^{-3}, which is a value typical for outdoor environmental radon concentrations (UNSCEAR 2000).

Figure 1 highlights the variation occurring in airborne radon concentration caused by meteorological conditions. These variations may explain discrepancies between ERA and ***eriss***'s radon progeny monitoring data at Jabiru (East), which are occasionally observed as radon progeny monitoring is performed one day per month only, at different days of the month. With our dataset, maximum day-to-day differences in radon concentration have been estimated to be more than 25 Bq·m^{-3}. This results in a maximum difference in average daily doses from the inhalation of radon progeny at Mudginberri of up to 2 μSv.

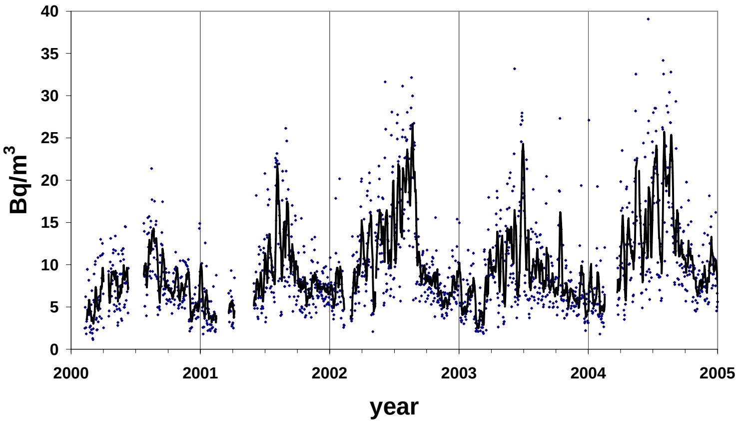

Figure 1 Daily averages of airborne radon concentration and weekly running average (solid line) measured at the Mudginberri Four Gates Road radon station plotted versus the date

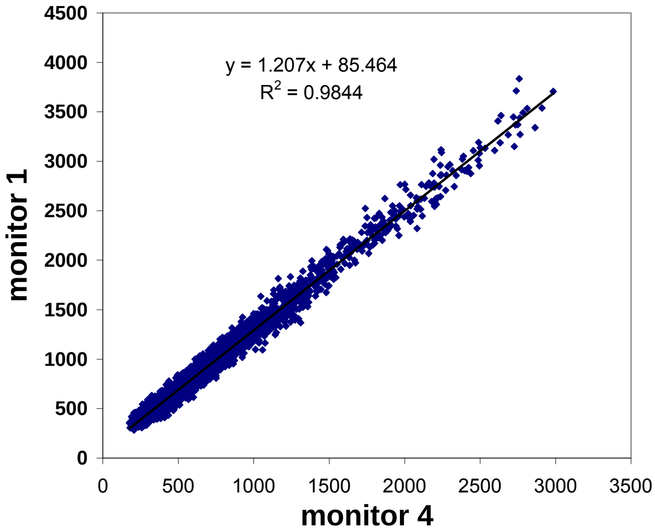

Figure 2 Net counts per 30 minutes of monitor 1 plotted against monitor 4

This year, an intercomparison between radon monitors 1 and 4 at Mudginberri Four Gates Road radon station was run for three consecutive months, from 17 January 2005 to 27 April 2005. Monitor 1 has been calibrated against a known radon concentration earlier in 2002 in ARPANSA's radon chamber in Melbourne (Bollhöfer et al 2004). The comparison shows that monitor 1 is about 20% more efficient than monitor 4, in agreement with calibration constants determined earlier in the laboratory using a Pylon radon source. These constants amounted to 0.018 ± 0.001 (monitor 1) and 0.020 ± 0.003 (monitor 4) Bq/m^3 per cts/30 min, respectively (Bollhöfer et al 2004). Monitor 1 has been used for cross calibration of the instruments since the ARPANSA calibration.

References

Bollhöfer A, Martin P, Tims S & Ryan B 2004. High sensitivity airborne radon concentration measurements in the Alligator River Region: rehabilitated Nabarlek uranium mine. Internal Report 469, Supervising Scientist, Darwin. Unpublished paper.

Martin P, Tims S, Ryan B & Bollhöfer A 2004. A radon and meteorological measurement network for the Alligator Rivers Region, Australia. *Journal of Environmental Radioactivity* 76, 35–49.Tims S 2001. Operation and calibration of the ***eriss*** radon detectors. Internal Report 359, Supervising Scientist, Darwin. Unpublished paper.

UNSCEAR 2000. *Sources and effects of ionizing radiation.* Report to the General Assembly, with scientific annexes. United Nations Scientific Committee on the Effects of Atomic Radiation, New York.

Introduction to SSD's stream monitoring program for Ranger, 2004–05

C Humphrey

Based upon its research program, SSD has implemented an integrated chemical, physical and biological control regime to ensure protection of the aquatic ecosystems of the Alligator Rivers Region from the operation of mines in the region. Since 2001, routine monitoring and ecotoxicity programs have been employed for environmental assessment of aquatic ecosystems.

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

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

- *Water chemistry* – physical and chemical indicators, including pH, electrical conductivity, suspended solids, uranium, magnesium, manganese and sulfate (weekly sampling during the wet season) and radium (fortnightly);
- *Creekside monitoring* of reproduction in freshwater snails and survival of fish fry (four-day tests conducted at fortnightly intervals during the wet season);
- *Bioaccumulation* – concentrations of chemicals (including radionuclides) in the tissues of freshwater mussels and fish at strategic locations downstream to detect far-field effects including those arising from any potential deposition of mine wastes in sediments (mussels sampled every late-dry season, fish sampled biannually in the late dry season).

(ii) Assessment of 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).

Bioaccumulation studies are currently under review and progress is reported separately in this report (Sauerland et al 2006). Results for water chemistry, creekside monitoring, macroinvertebrate (Magela and Gulungul creek sites only) and fish community studies conducted during the 2004–05 wet and early dry seasons, and macroinvertebrate studies carried out in 2004 are reported in the ensuing papers of this SSR.

Reference

Sauerland C, Ryan B, Humphrey C & Jones DR 2006. Review of the bioaccumulation monitoring program. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 60–62.

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

M Iles

Magela Creek

The first water chemistry samples for the Supervising Scientist's 2004–05 wet season surface water monitoring program were collected from the Magela Creek downstream statutory compliance point, gauging station 009, on 21 December 2004, one day after the commencement of flow past the site. Weekly spot-sampling continued throughout the wet season with the last of the routine monitoring samples collected on 25 May 2005, the week before flow past the downstream compliance point ceased.

All indicators remained within limits/guidelines[1] throughout the 2004–05 wet season. The measured values are indicative of the pattern of improved quality seen in the past three wet seasons, exemplified in the uranium results of figure 1.

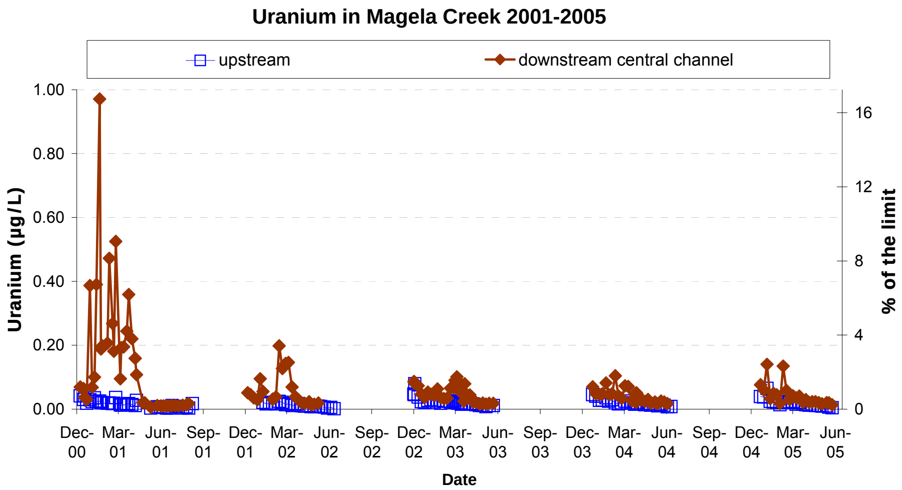

Figure 1 Uranium concentrations in Magela Creek since the 2000–01 wet season (SSD data)

The upstream and downstream key water quality data from both the SSD and ERA programs are summarised in table 1 with uranium concentrations shown in figure 2.

There appears to be good agreement between the datasets. Uranium, magnesium and sulfate wet season median concentrations from both datasets were higher downstream of the mine but the concentrations were very low and not of environmental concern. For the season, uranium was less than 3% of the limit and mostly less than 1% (figure 2). Electrical conductivity (EC), whose guideline value provides a management control for the 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 each dataset.

[1] The terms 'limit', 'guideline' and 'objective' are defined in the Supervising Scientist Annual Report for 2004–05

The water quality objectives set to protect the aquatic ecosystems downstream of the mine (provided in the Supervising Scientist annual report for 2004–05) were achieved during the 2004–05 wet season. Available biological monitoring data (ensuing papers) also indicate that the environment remained protected throughout the season.

Table 1 Summary of Magela Creek 2004–05 wet season water quality upstream and downstream of Ranger

Parameter	Guideline or Limit	Organisation	Median		Range	
			Upstream	Downstream	Upstream	Downstream
pH	5.0 – 6.9	SSD	6.3	6.3	5.4 – 6.7	5.8 – 6.7
		ERA	6.1	6.2	5.7 – 6.5	5.8 – 6.4
EC (µS/cm)	43	SSD	13	17	5.9 – 17	7.3 – 23
		ERA	12	14	7.9 – 18	8.4 – 19
Turbidity (NTU)	26	SSD	2.2	2.6	0.9 – 15	0.9 – 12
		ERA	2.	3.	<1 – 25	<1 – 5
Sulfate‡ (mg/L)	Limited by EC	SSD	0.2	0.7	0.1 – 0.3	0.2 – 1.8
		ERA	0.2	1.0	<0.1 – 0.8	0.4 – 3.2
Magnesium‡ (mg/L)	Limited by EC	SSD	0.5	0.8	0.2 – 0.8	0.3 – 1.0
		ERA	0.6	0.9	0.4 – 0.8	0.4 – 1.3
Manganese‡ (µg/L)	26	SSD	5.0	5.3	3.3 – 8	3.0 – 22
		ERA	6.0	5.7	2.7 – 13	1.5 – 17
Uranium‡ (µg/L)	6	SSD	0.015	0.031	0.004 – 0.065	0.014 – 0.145
		ERA	0.015	0.035	<0.005 – 0.048	0.018 – 0.174

ERA data taken from the ERA Weekly Water Quality Report 5 July 2005;

‡ dissolved (<0.45 µm);

A compliance limit applies to uranium; management guidelines apply to all other parameters shown.

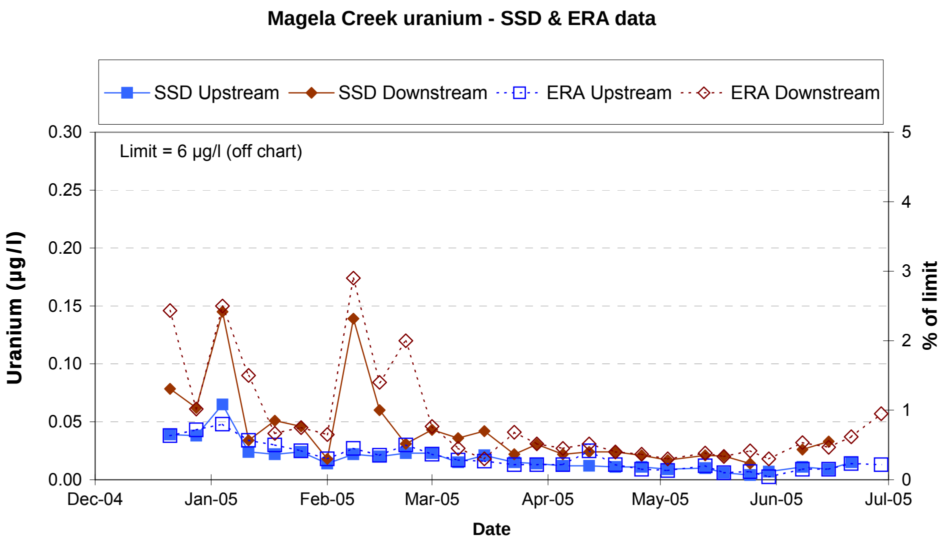

Figure 2 Uranium concentrations measured in Magela Creek by SSD and ERA during the 2004–05 wet season

Gulungul Creek

The first water chemistry samples for the Supervising Scientist's 2004–05 wet season surface water monitoring program were collected from Gulungul Creek on 23 December 2004, the first day of flow in the creek for the wet season. Weekly spot-sampling continued throughout the season with the last of the routine monitoring samples collected on 8 June 2005, during the week in which flow at the downstream site ceased.

The overall water quality and seasonal trends were comparable to those seen in previous years. This is demonstrated by the uranium concentrations shown in figure 3.

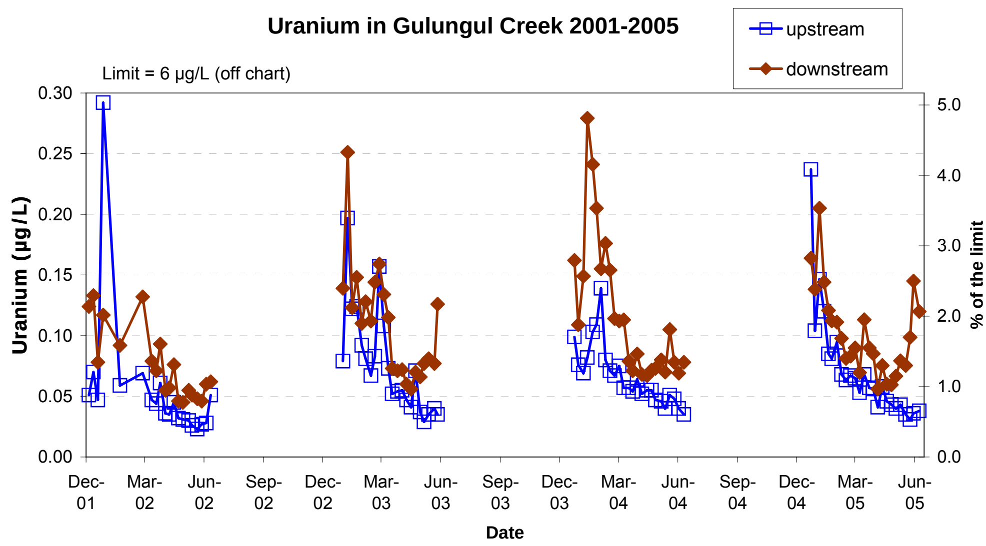

Figure 3 Uranium concentrations in Gulungul Creek since the 2000–01 wet season (SSD data)

The upstream and downstream key water quality data from both the SSD and ERA programs are summarised in table 2 with uranium concentrations shown in figure 4. There appears to be good agreement between the two datasets. Small differences could be attributable to different sampling times.

Table 2 Summary of Gulunugul Creek 2004–05 wet season water quality upstream and downstream of Ranger

		Median		Range	
Parameter	Company	Upstream	Downstream	Upstream	Downstream
pH	SSD	6.6	6.6	5.7 – 6.9	5.8 – 6.9
	ERA	6.3	6.3	8.9 – 6.8	6.0 – 6.8
EC (µS/cm)	SSD	17	21	9.0 – 26	9.4 – 28
	ERA	14	18	10 – 21	13 – 23
Turbidity	SSD	1.4	1.8	<0.5 – 4.0	0.6 – 6.7
(NTU)	ERA	1.	2.	<1 – 20	<1 – 7
Sulfate‡	SSD	0.2	0.5	<0.1 – 0.9	0.1 – 1.4
(mg/L)	ERA	0.2	0.6	0.1 – 1.0	0.1 – 2.3
Magnesium‡	SSD	0.9	1.1	0.4 – 1.6	0.4 – 1.5
(mg/L)	ERA	1.0	1.2	0.7 – 1.6	0.8 – 1.4
Manganese‡	SSD	2.7	3.0	1.2 – 8.0	0.7 – 15
(µg/L)	ERA	2.1	3.0	1.1 – 9.5	0.8 – 7.8
Uranium‡ *	SSD	0.058	0.093	0.031 – 0.237	0.058 – 0.212
(µg/L)	ERA	0.069	0.110	0.034 – 0.253	0.063 – 0.249

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

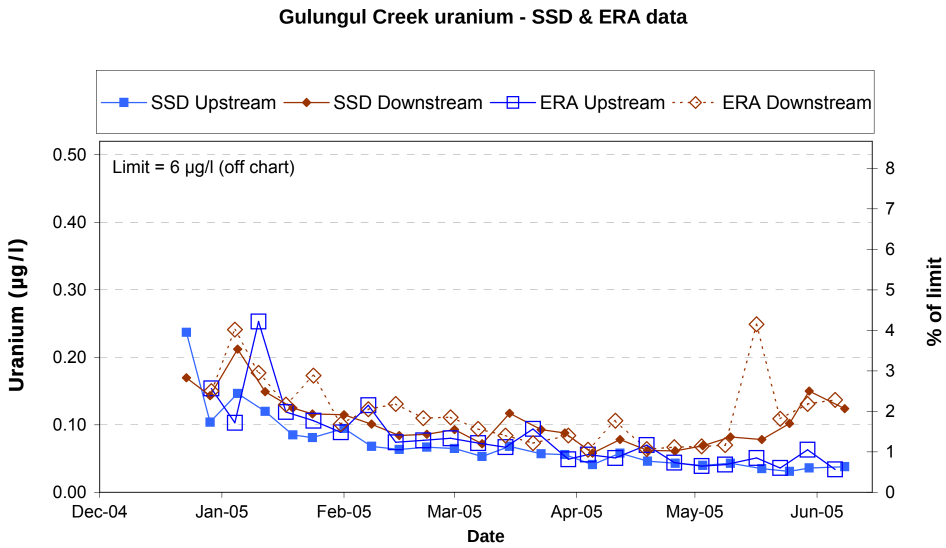

Figure 4 Uranium concentrations measured in Gulungul Creek by SSD and ERA during the 2004–05 wet season

Uranium concentrations in Gulungul Creek are naturally higher than those in Magela Creek (due to different geochemistry and hydrology influences in the respective catchments). Like Magela Creek, the concentrations at the Gulungul Creek downstream site are slightly higher than those at the upstream site. However, the uranium concentrations in Gulungul Creek were well below the limit throughout the season, with the concentration at both the upstream and downstream sites ranging between about 1% and 4% of the limit (figure 3).

Sulfate concentrations in Gulungul Creek were similar to those seen in previous years, with the downstream concentrations generally higher than those upstream but still well below levels of environmental concern[2]. The small difference in electrical conductivity (EC) between the upstream and downstream sites shows that the sulfate increase has little effect on the overall solute levels downstream of the mine. The EC trend at both upstream and downstream sites closely follows that of magnesium. Magnesium, manganese, pH and turbidity were similar at the upstream and downstream sites indicating that these variables are not, or only slightly, influenced by the mine.

[2] Toxicity tests on local *Hydra* and *Lemna* species demonstrated that SO_4 (as Na_2SO_4) exhibits very little, if any toxicity to these species below 200 mg L^{-1}.

Creekside monitoring in Magela Creek

C Humphrey, D Buckle & R Luxon

In this form of monitoring, effects of Ranger mine wastewater dispersion 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, some 5 km downstream of the mine. At each of the two sites, duplicate pumps in the creek feed water separately to: (i) in the case of snails, duplicate containers respectively, each container holding replicate (8) snail pairs; and (ii) in the case of fish, triplicate containers respectively, each container holding ten larval fish. 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 other week during the wet season. Tests usually commence in December and cease in early April, 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 1A for snail egg production, and in figure 1B 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 website (http://www.deh.gov.au/ssd/monitoring/magela-bio.html).

Eight creekside tests were conducted in the 2004–05 wet season (late December 2004 to early April 2005.) Only five tests were conducted using larval fish, there being too few fish larvae available to conduct the fourth, sixth and seventh tests.

Amongst snail tests, egg production at upstream and downstream sites was similar across all tests conducted for the wet season (figure 1A). Using the data shown in figure 1A, 'difference' values for 2004–05 were compared with those from previous years. (The difference data shown and subsequently used in statistical analyses are those for valid tests only.) No significant difference was found ($P>0.05$).

The lack of fish larvae available in the fourth, sixth and seventh tests was a result of high mortality in the developing eggs held in broodstock waters at the Jabiru Field Station laboratories. Broodstock waters are taken from Magela Creek upstream of Ranger. Broodstock waters collected for these (failed) tests generally coincided with periods when flow in Magela Creek was unusually low. It is possible that during these periods, high water temperatures and insolation are conducive to the build up of micro-organisms that are harmful to fish larval development. This issue is being investigated, together with possible solutions to prevent future occurrences of such test failures.

Across all fish tests, larval fish survival at upstream and downstream sites was similar (figure 1B), apart from reduced survival at the upstream site relative to the downstream site during the third creekside test in particular (an observation commonly noted in previous years for this test species).

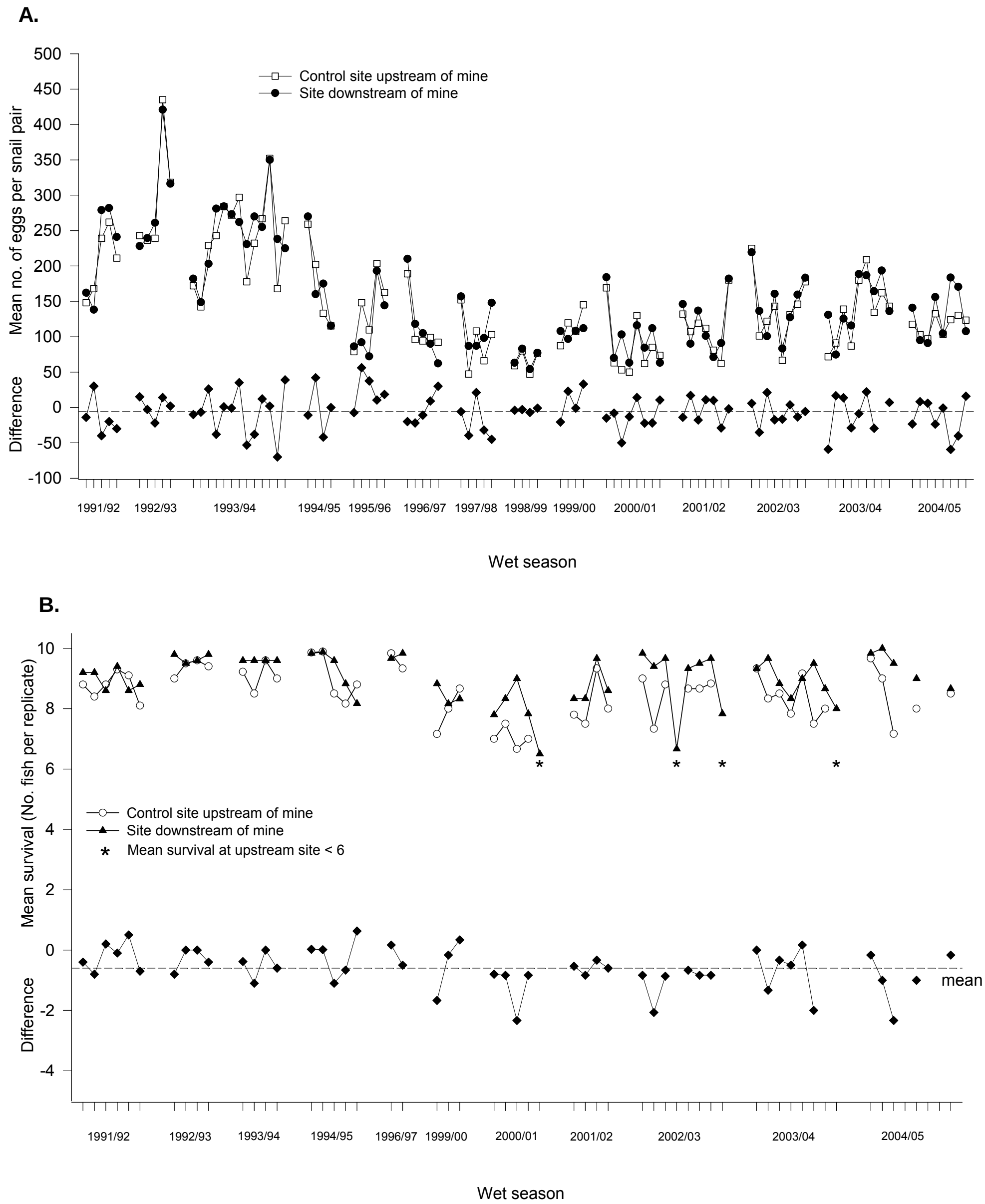

Figure 1 Creekside monitoring results for: A. freshwater snail egg production, and B. larval black-banded rainbowfish survival, for wet seasons between 1992 and 2005

In the Supervising Scientist annual report for 2003–2004, is was noted that fish survival 'difference' (upstream-downstream) data for the two periods 1991/92–96/97 and 1999/00–03/04 were significantly different from one another as a consequence of the reduced larval fish survival at the upstream control site in the period 1999/00–02/03. (Possible causes are discussed in the Supervising Scientist annual report for 2002–2003.)

With the inclusion of 2004/05 data, the same significant difference, 1991/92–96/97 versus 1999/00–03/04 (P=0.004), was observed. When 'difference' results for 2004/05 were compared separately with results for the two time periods (1991/92–96/97 and 1999/00–03/04), then:

- a significant difference was found in the comparison 1991/92–96/97 versus 2004/05 ($P<0.05$), but
- no significant difference was found in the comparison 1999/00–03/04 versus 2004/05 ($P>0.05$).

This result indicates that larval fish survival at the downstream relative to upstream (control) site in Magela Creek during 2004/05 was consistent with the same relative survival rates observed in the previous five wet seasons.

From the collective creekside results, it was concluded that there were no adverse effects of dispersed Ranger mine waste waters to Magela Creek on either of the creekside test species over the 2004–05 wet season.

Monitoring using macroinvertebrate community structure

C Humphrey, J Hanley, C Camilleri & A Cameron[1]

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 meet the needs of cost efficiency and improved ability to confidently attribute any observed changes to mining impact. 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 sites in three additional control streams. Since 1994, there have also been three changes to sampling and sample processing methods.

The refined (1994) design for this macroinvertebrate study was based on the principle of gathering macroinvertebrate samples from sites in Magela Creek upstream and downstream of Ranger, and also from similar paired upstream and downstream sites in three adjacent 'control' streams that are generally unaffected by any mining activity. In recent years, it has become evident that Gulungul Creek has been receiving some small quantities of mine contaminants from Ranger. Given its doubtful role as a true control stream, it is more appropriate now to consider this stream in the same category as Magela Creek, that is, 'exposed'. The design of this study, therefore, is now a balanced one comprising two 'exposed' streams and two control streams.

Samples were collected from each site at the end of each wet season (between April and May). For each sampling occasion and for each pair of sites for a particular stream, 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 both the pre-disturbance, baseline period and in undisturbed control streams; the higher dissimilarity is a consequence of the 'altered' (disturbed) macroinvertebrate community structure at (the) site(s) downstream of such point sources.

Analysis of the full macroinvertebrate data set from 1988 to 2004, and data from the paired sites in the two 'exposed' streams, Magela and Gulungul creeks for 2005, 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 upon which to assess whether or not significant changes have occurred as a consequence of mining. Notwithstanding, the plots show that the mean

[1] Formerly SSD; now Ecowise Environmental P/L, Melbourne

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, single-factor ANOVA shows no significant difference in the mean dissimilarities between the two treatment groups, 'control' versus 'potentially disturbed' streams.

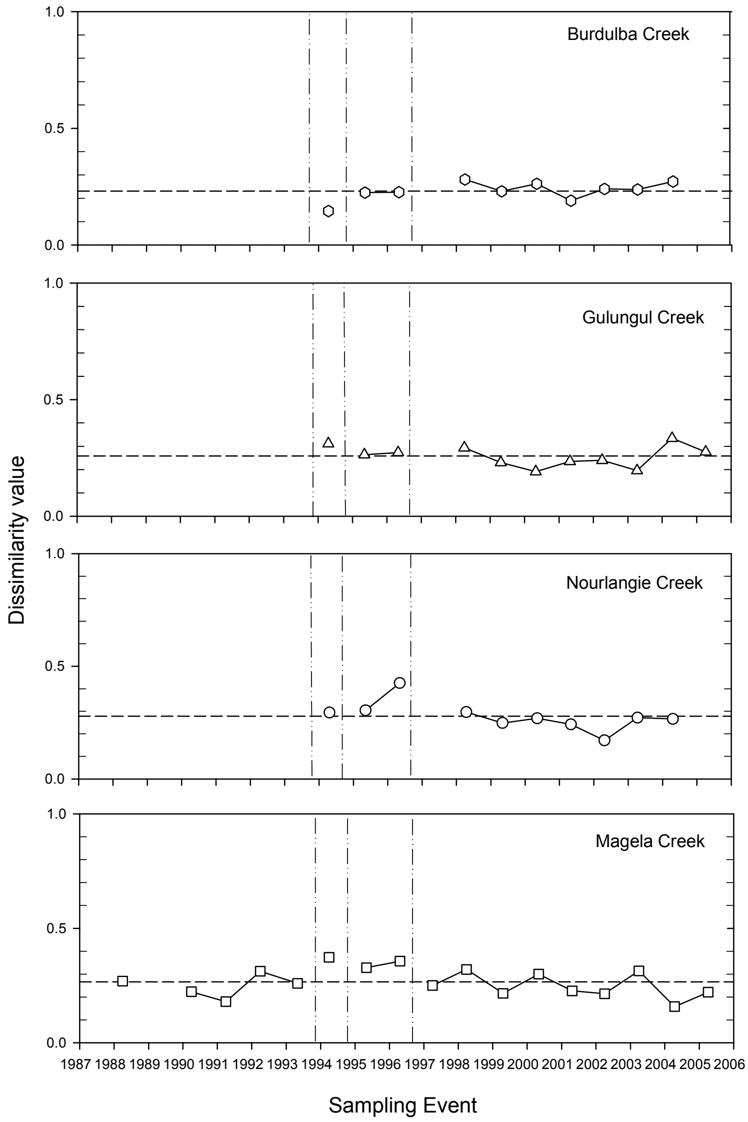

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger uranium mine for the period 1988 to 2005. The dashed vertical lines delineate periods for which a different sampling and/or sample processing method was used. Dashed horizontal lines indicate mean dissimilarity across years.

Processing of samples collected from the two control streams in 2005, Burdulba and Nourlangie, had not been completed by the time this report was prepared.

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 macroinvertebrate data gathered from, the sites sampled in Magela and Gulungul creeks downstream of Ranger for each year of study, together with all other control sites sampled over the same time period. Because the data-points associated with these two sites are interspersed amongst the points representing the control sites (including data from 2004 and 2005), this indicates that these 'exposed' sites have macroinvertebrate communities that are not dissimilar to those occurring at control sites.

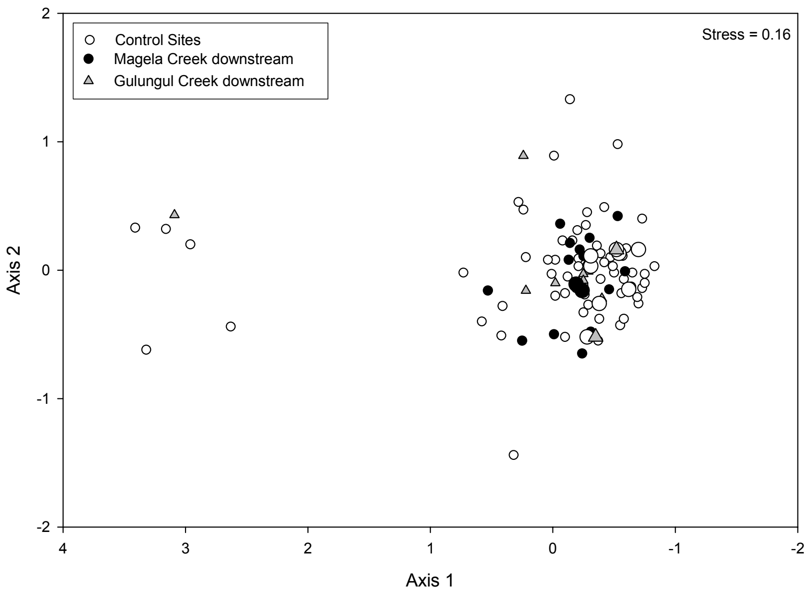

Figure 2 Ordination plot of macroinvertebrate communities sampled from sites in several streams in the vicinity of the Ranger mine for the period 1988 to 2005. Data from 2004 and 2005 are indicated by the enlarged symbols.

Collectively, these results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining in the period 1994 to 2005, at least, are not sufficient to have adversely affected macroinvertebrate communities.

Reference

Faith DP, Dostine PL & Humphrey CL 1995. Detection of mining impacts on aquatic macroinvertebrate communities: Results of a disturbance experiment and the design of a multivariate BACIP monitoring programme at Coronation Hill, Northern Territory. *Australian Journal of Ecology* 20, 167–180.

Monitoring using fish community structure

R Pidgeon & C Humphrey

Sampling of fish communities in billabongs is conducted in late April to the end of June of 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, and directly exposed to any released mine waters ('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, from 1994 to the present:
   - 'Directly exposed' billabongs in Magela Creek adjacent to and downstream of Ranger mine. These sites are directly exposed to contaminated surface flows from the minesite;
   - 'Indirectly exposed' billabongs in Magela Creek downstream of the mine. Whilst not directly receiving mine waste waters, the sites can receive contaminated creek water, indirectly, by back flow ('pseudo' controls);
   - 'Control' billabongs in Wirnmuyirr Creek (Winmurra) (a floodplain tributary of Magela Creek), Nourlangie Creek (Sandy and Buba) and East Alligator River (Cathedral) (true controls). These sites cannot receive contaminated water as they are in different catchments with no mining activity. (In previous annual reporting *viz* the 'Supervising Scientist annual reports' (2003–04 and earlier), Winmurra Billabong has been regarded as an 'indirectly exposed' site.)

The design for both approaches is amenable to the comparisons: (i) directly exposed billabong(s) versus control billabong(s) from independent catchments (Nourlangie Creek, East Alligator River, Wirnmuyirr Creek); and/or (ii) directly exposed billabongs versus indirectly exposed billabongs in Magela Creek, recognising that this second approach is confounded by possible movement of fish between the two lowland billabong types in the same stream system.

Since mining activities commenced at Ranger in 1979, changes unrelated to mining have occurred in stream catchments that, if not well understood, have the potential to confound conclusions drawn about the environmental impact of mining. These changes may be associated with natural climatic events or phases, or may be due directly or indirectly to invasive species removed from, introduced to, or which have increased their range throughout, the region. Hence an important ongoing task is to gain a sound understanding of the dynamics and factors affecting populations and communities of the key biota in streams adjacent to ARR minesites. In this way, changes in metrics used to summarise responses of these monitoring organisms can be correctly attributed to mining or non-mining-related causes. Alongside monitoring results for the respective studies, two such investigative studies are reported in ensuing papers of this report, focusing in fish communities sampled in deeper channel (Humphrey et al 2006) and shallow lowland billabongs (Pidgeon et al 2006).

References

Humphrey C, Buckle D & Pidgeon R 2006. Fish communities in channel billabongs. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 48–53.

Pidgeon R, Luxon R & Buckle D 2006. Fish communities in shallow lowland billabongs. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 54–57.

Fish communities in channel billabongs

C Humphrey, D Buckle & R Pidgeon

Introduction

In this sampling technique, visual observations are made upon fish communities inhabiting the littoral zones of deep, sandy-bottomed channel billabongs. Five sites are sampled in each of Mudginberri and Sandy billabongs, each site surveyed repeatedly (five times) along a 50 m transect set parallel to the shore. Typically, the transect is set immediately adjacent to steep banks with dense, over-hanging or submerged pandanus palms. Observations are made through the front of a boat with custom-made, clear, perspex-viewing dome.

The basic design entails the simple pairwise comparison of fish community data between Mudginberri (directly exposed) and Sandy (control) billabongs using multivariate dissimilarity indices. These indices and rationale for their use are explained in an accompanying monitoring paper contained in this SSR (Humphrey et al 2006). While data for Mudginberri Billabong have been gathered since 1989, only the results since 1994 are shown here, the period from which the additional control billabong (Sandy) has been sampled. A plot of the paired-site dissimilarity values using log-transformed data, 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) and 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 three years, there has been a return in both sites to a reasonably common community structure that is also similar to that found at the commencement of the paired-site study in 1994.

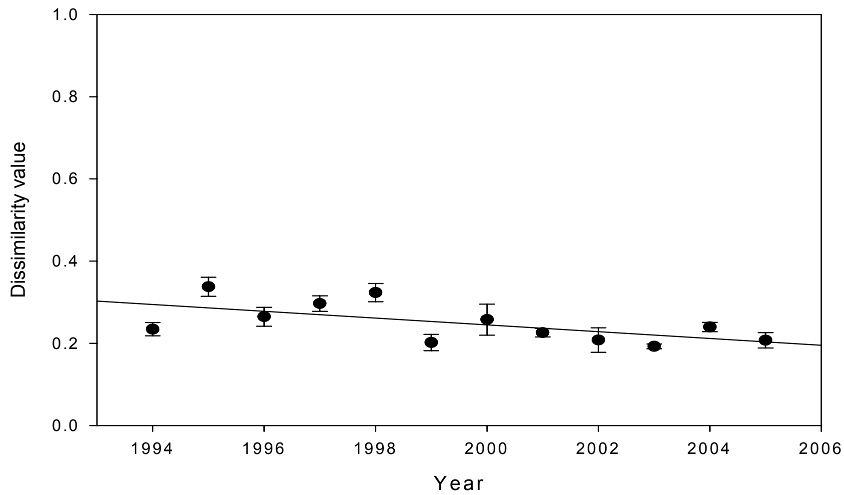

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') billabongs in the vicinity of the Ranger uranium mine over time. Values are means (± standard error) of the 5 possible (randomly-selected) pairwise comparisons of transect data between the two billabongs, while line about the means illustrates the fitted regression (R^2 = 0.2, P = 0.0003).

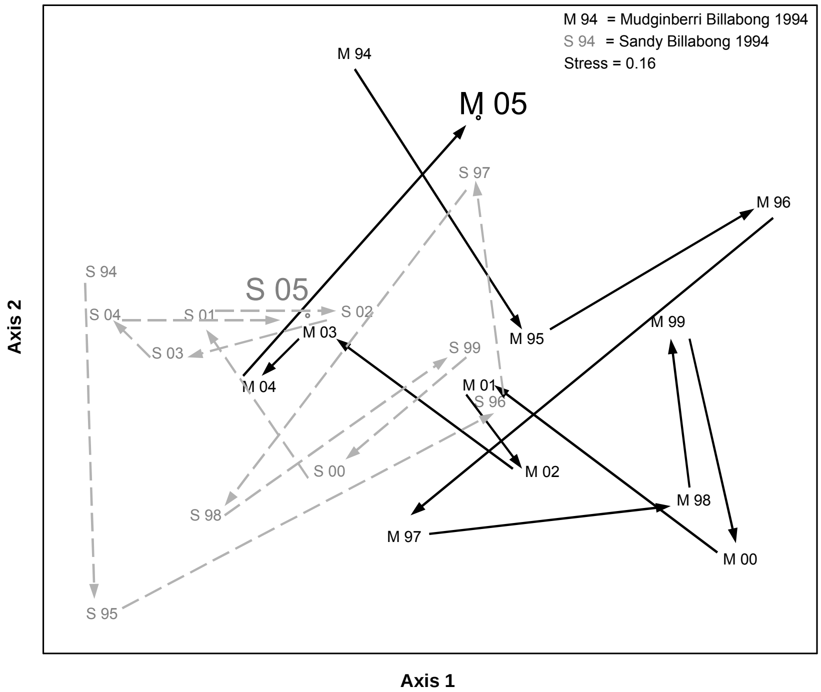

Figure 2 Ordination plot of fish communities sampled from two channel billabongs in the vicinity of the Ranger uranium mine for the period 1994 to 2005. Lines follow the trajectory of sites over time.

In the Supervising Scientist annual report for 2003–2004, a significant decline was noted in the paired-site dissimilarity measures over time (figure 1). Large discrepancies in the abundances of numerically-dominant fish species between billabongs are particularly influential in inflating dissimilarity measures. In the 2003–2004 annual report, the decline in the dissimilarity measure was attributed to the particularly high abundances of chequered rainbowfish (*Melanotaenia splendida inornata*) and to a lesser extent glassfish (*Ambassis* spp) in Mudginberri Billabong in the early years of the study, relative to Sandy Billabong. This result has subsequently been confirmed. Thus, in table 1, the influence of these numerically-dominant fish species in channel billabongs upon correlation and regression results for the dissimilarity versus time relationship is shown. Removal of each species, particularly chequered rainbowfish, reduces the significance of the decline in the dissimilarity measure. Removal of both fish species results in a non-significant result ($P>0.05$).

Table 1 Influence of numerically-dominant fish species in channel billabongs upon correlation and regression results for the paired-site dissimilarity value, and time

	Correlation (ρ)	Regression parameters	
		R^2	P
All taxa	-0.45	0.20	0.0003
Glassfish species removed	-0.36	0.13	0.005
Chequered rainbowfish removed	-0.31	0.07	0.02
Both taxa removed	-0.17	0.03	0.20

The relative abundances of chequered rainbowfishes and glassfish in both Mudginberri and Sandy billabongs are plotted as a time series in figure 3. (Sampling in Mudginberri commenced in 1989 while sampling in Sandy did not commence until 1994.) In 1996, the visual-observation sampling method was conducted relatively early in the wet-dry recessional flow period in Mudginberri Billabong, and the high counts of both fish species observed in this year reflect fish migrating upstream through the billabong. (Normally sampling commences later in the season, after significant fish migration has ceased.)

A feature of the abundance plots of figure 3 is the decline in rainbowfish in Mudginberri Billabong since 1989. Omitting anomalous 1996 data (see above), this decline is shown to be significant in regression analysis (R^2 = 0.547, p = 0.001). The decline in this fish species in Mudginberri, relative to Sandy, is the main reason for the corresponding decline in the paired-site dissimilarity measure (table 1). For this reason, possible causes of this decline were examined. Potential correlates of rainbowfish abundance were sought from water quality (natural and related to wet season wastewater discharges from Ranger mine) and quantity (stream discharge) variables.

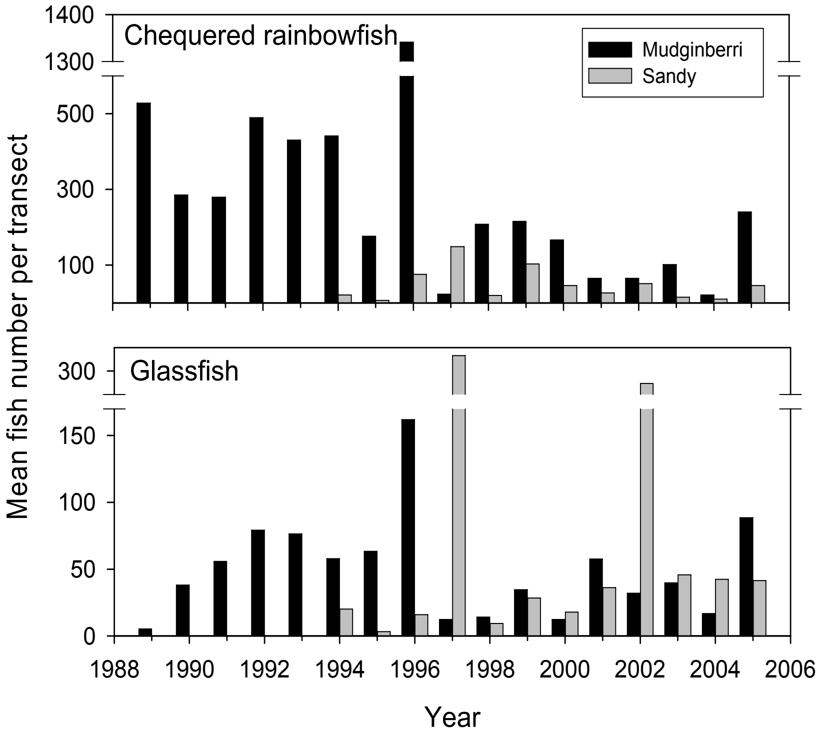

Figure 3 Relative abundances of chequered rainbowfish and glassfish in channel billabongs over time

In addition, rainbowfishes (and glassfishes) observe very significant migrations in Magela Creek after wet season spawning and recruitment on the (downstream) floodplain (ARRRI Annual Research Summary 1987–88). Thus, the abundances of these fishes in Mudginberri at the time of annual sampling reflect, to a large extent, breeding and recruitment success on Magela floodplain. (This migration phenomenon appears to be less pronounced in Nourlangie Creek (***eriss*** unpublished data), possibly explaining to some extent the smaller magnitude and/or variability in numbers of the same two fish species in Sandy Billabong.) Therefore, correlates that could explain changes to floodplain conditions over time, as these affect rainbowfish breeding and recruitment success, were also sought.

Key environmental correlates of rainbowfish abundance and decline in Mudginberri Billabong over time are shown in figure 4. These results may be summarised as follows.

Water quality associated with Ranger mine wastewater discharges

The dominant contaminants associated with Ranger mine wastewater discharges to Magela Creek are uranium, magnesium and sulphate. Uranium (U) data derived from Magela Creek prior to 2000 are unreliable, due to contamination and instrumentation problems (ie poor detection limits). Therefore, magnesium (Mg) data were analysed as a reasonably reliable, surrogate measure of mine wastewater contaminant concentrations in Magela Creek. Because Mg concentrations are naturally and inversely correlated with stream discharge in Magela Creek, the net input of Mg from Ranger was derived, this being the difference in median wet season concentration between downstream (compliance site) and upstream (control) locations. The plot of net wet season Mg concentration and corresponding rainbowfish abundance in Mudginberri Billabong for that wet season is provided in figure 4C. No significant relationship is observed. This is not surprising: concentrations of U and Mg in Magela Creek arising from mine wastewater discharges are at least two orders of magnitude lower than those known to adversely affect larval fishes, including in the case of U, chequered rainbowfish (eg Supervising Scientist annual report 2003–04, section 3.4.1 and Supervising Scientist annual report 2004–05, chapter 3, ecotoxicity research highlight).

Stream discharge and natural water quality

Since the commencement of this study in 1989, there has been a general increase in wet season rainfall and associated stream discharge in Magela Creek. Consequently, total wet season discharge in the creek is significantly and negatively correlated with rainbowfish numbers in Mudginberri Billabong (figure 4A). How higher discharge *per se* would result in lower fish numbers is not clear. However, greater discharge volumes result in greater dilution of wet season surface waters and their solute concentrations. Median wet season values of electrical conductivity (EC) of Magela Creek waters upstream of Ranger were used as a surrogate measure of solute concentrations. Not surprisingly, median wet season EC in Magela Creek is significantly and positively correlated with rainbowfish numbers (figure 4B).

Magela Creek surface waters are extremely soft and poorly-buffered, factors accentuated at high wet season flows in the creek. It is possible that early life stages of rainbowfishes are stressed under these conditions (which include relatively high acidity) and lack essential minerals for growth and development. There is some experimental evidence to support this. The fry of both chequered rainbowfish (ARRRI Annual Research Summary 1987–88) and the congener, the black-lined rainbowfish used in the SSD's creekside testing program (section 2.2.3 of the Supervising Scientist annual report for 2004–05), exhibit reduced survival when exposed to creek waters during high flow events.

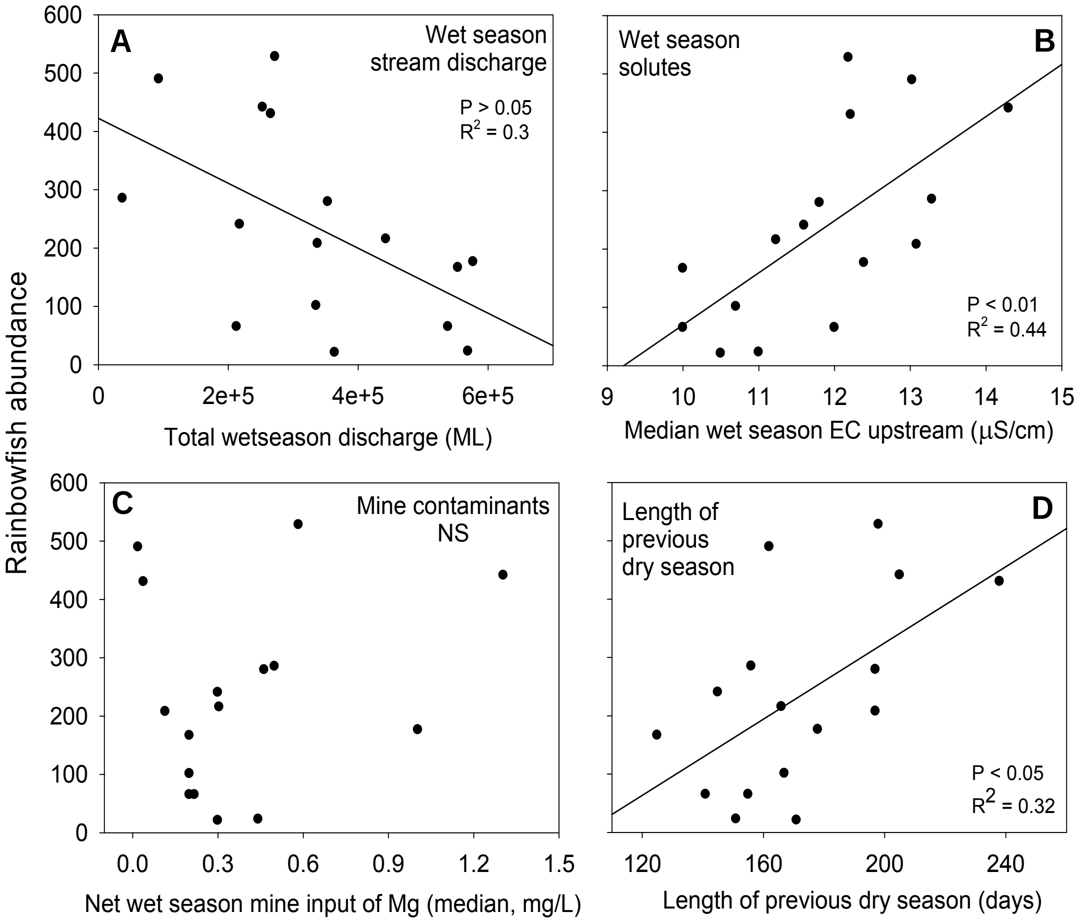

Figure 4 Environmental correlates of rainbowfish abundance in Mudginberri Billabong, 1989–2005

Conditions on Magela floodplain

Changes to vegetation communities on Magela floodplain, the main breeding area and recruitment source for chequered rainbowfish in Magela Creek, may have adversely affected rainbowfish populations. In particular, a number of grass species are rapidly expanding in range and densities on the floodplain, due partly to removal of feral buffalo that once grazed on these grasses and acted as a form of control. A particularly aggressive species is the exotic para grass (*Urochloa mutica*) which is currently expanding its area of coverage at 14% p.a. Without management it could dominate the floodplain in 15 to 20 years. With the presence of satellite patches of para grass, this could be a conservative time frame (Bayliss et al 2006). The recent rapid expansion of para grass on Magela floodplain corresponds to the period of decline of chequered rainbowfish in Mudginberri Billabong. It is quite possible that the expansion of this and other exotic and native grasses (especially native *Hymenachne*) has had some adverse effects on recruitment of rainbowfish.

Impacts of both para grass and native *Hymenachne* on floodplain biota were studied by Douglas et al (2002). Stands of both these grasses contained fewer fish species and lower fish abundance than areas of more open vegetation dominated by wild rice (*Oryza meridionalis*). Consequently, any increase in the area covered by either para grass or hymenachne could have adverse effects on the recruitment of fish that utilise floodplain habitats in the wet season into dry season refuges in floodplain billabongs and upstream channel billabongs.

Unfortunately, there are no comparable data on possible grass expansion for the floodplain of Nourlangie Creek downstream of Sandy Billabong.

A feature of floodplain hydrology that may also affect fish breeding success and recruitment is the period of drying of the floodplain prior to annual re-wetting. The so-termed 'flood-pulse' theory predicts and shows that primary and secondary production of floodplains is dependant upon the degree of seasonal drying and inundation. In particular, chemical cycling and nutrient release are dependent upon, and enhanced by, sediment and soil drying prior to wet season inundation. For Magela Creek, a shorter previous-dry-season and reduced drying of the floodplain may reduce ensuing wet season production on the floodplain nursery zone. The relationship between length of previous dry season (ie from Magela Creek cease-to-flow to commencement of flow, downstream of Ranger) and Mudginberri rainbowfish numbers is, in fact, significant (figure 4D), lending some support to this hypothesis.

It is worth noting that the expansion of grasses on Magela floodplain will have lead to greater soil water retention and thereby, and independently, accentuated any reduction in floodplain drying observed since this monitoring study commenced.

Summary

The decline in rainbowfish numbers in Mudginberri Billabong over the period 1989 to 2005 does not appear to be related to any change in water quality associated with mine wastewater discharges from Ranger. Over time with further monitoring and analysis, it may be possible to distinguish and identify natural stream water quality, discharge and/or floodplain habitat factors responsible for changes to fish populations in Magela Creek billabongs. These causal factors may then be modelled to account for variation in monitoring response variable(s).

References

Bayliss P, van Dam R, Boyden J & Walden D 2006. Ecological risk assessment of Magela floodplain to differentiate mining and non-mining impacts. In ***eriss** research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 172–185.

Douglas MM, Bunn SE, Pidgeon RJW, Davies PM, Barrow P, O'Connor RA & Winning M 2002. Weed management and the biodiversity and ecological processes of tropical wetlands. Final report to Environment Australia and Land and Water Australia.

Humphrey C, Hanley J, Camilleri C & Cameron A 2006. Monitoring using macroinvertebrate community structure. In ***eriss** research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 43–45.

Fish communities in shallow lowland billabongs

R Pidgeon, R Luxon & D Buckle

The design of this monitoring technique is described in an accompanying introductory paper contained in this report (Pidgeon & Humphrey 2006).

Results for 2004–05

The abundance of fish captured by pop-net sampling in shallow billabongs for the period 1994 to 2005 is shown in figure 1.

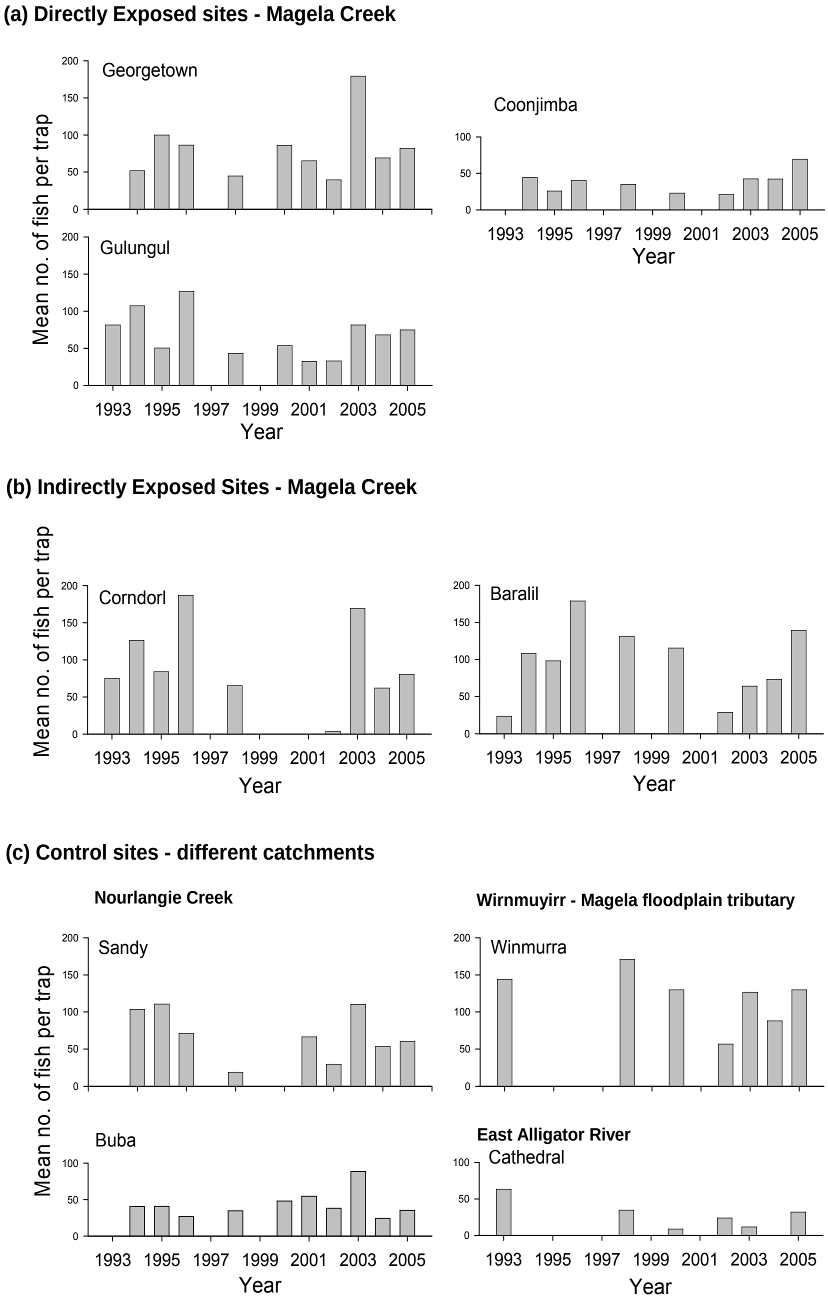

Figure 1 Relative abundance of fish in billabongs with different degrees of exposure to contaminants from Ranger uranium mine, 1994–2005. Not all sites have been sampled each year. Sampling in Djalkmara Billabong was discontinued after mining at Pit 3 closed its connection to Magela Creek.

Abundances in 2005 were similar to, or slightly greater than in 2004. The changes in fish abundances in 2005 were, with one exception, all within the range of natural variation observed for each location since sampling commenced in 1994. Fish abundance in Coonjimba Billabong was the highest so far recorded for this site.

The number of fish species recorded in each billabong has varied only slightly over the 1994 to 2005 period by comparison with fish abundance (data not shown here).

Multivariate ordination was used to compare the fish community structures in the three exposure types (figure 2). The area enclosed by data points for each billabong in years prior to 2005 is shown as a polygon to indicate their 'natural' location in the ordination space. Whilst some sites overlap considerably, the data points for different sites tend to occupy different areas of the ordination indicating differences in community structure. Formal statistical testing of the ordination patterns shows significant differences among some sites and amongst the three exposure types (ANOSIM, R= 0.37–0.42, P = 0.001).

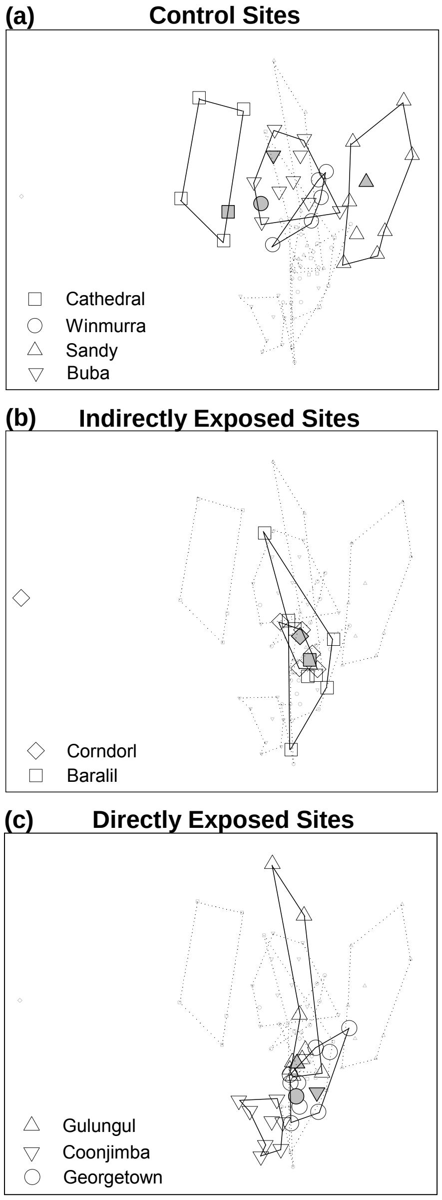

Figure 2 Ordination plots of fish communities in shallow billabongs near Ranger uranium mine with different potential exposure to contamination from mine wastes, for 1994 to 2005. Ordination calculated for 2 dimensions, stress = 0.18. The areas enclosing points for years 1993 to 2004 for each billabong are shown as polygons. (Only the relevant polygons are highlighted.) The position of data for 2005 is indicated separately as filled symbols for comparison with previous years.

Nevertheless, the most important feature of the ordination pattern is that the relative position of sites has remained quite constant over time. The reasonably-well defined locations of each of the billabongs in the ordination over the 11 years of sampling provide a useful basis for detecting and assessing change. In particular, departure from the natural patterns in community structure in exposed sites could indicate adverse effects of mining activity. The potential of this detection procedure was indicated by an outlying data point for Corndorl Billabong in 2002 that was related to an abnormal cover of the exotic floating weed, *Salvinia molesta* (Supervising Scientist annual report, 2002–2003).

The ordination results for 2005 lie within, or very close to, these natural positions in the ordination space and indicate there was no discernible effect of mining activity on fish communities. This was supported by the temporal pattern of multivariate dissimilarities between control and direct exposure sites (figure 3). There were no significant differences among years. The slight trend of increase over time is not statistically significant, nor did closer examination of the numerically-dominant fish species indicate any trends nor divergence in abundances of fish between direct exposed and control sites (cf channel billabong analysis, described above).

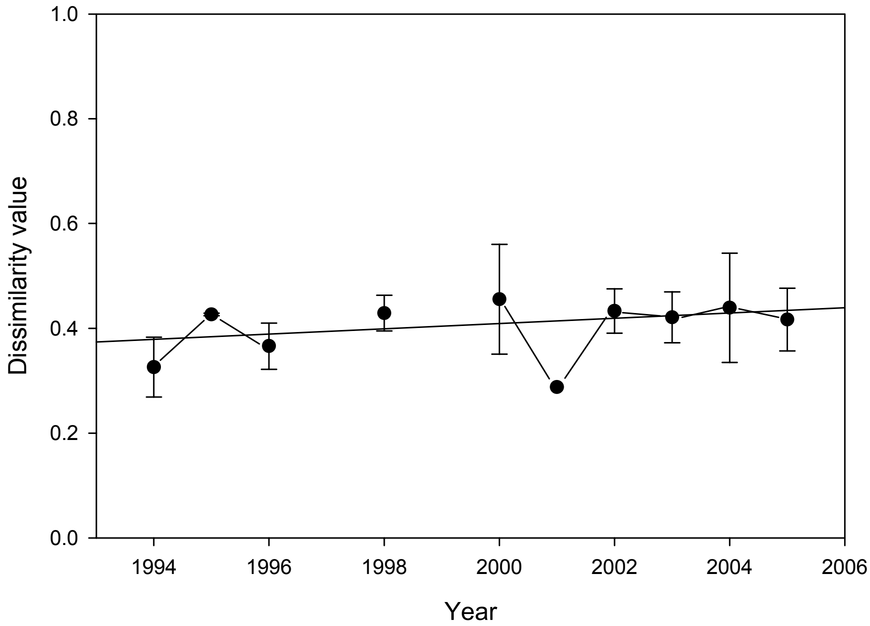

Figure 3. Paired dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in 'directly exposed' Magela and 'control' Nourlangie and Magela billabongs in the vicinity of the Ranger uranium mine over time. Values are means (± standard error) of the (up to) 3 possible pairwise billabong comparisons: Coonjimba vs Buba, Gulungul vs Winmurra, and Georgetown vs Sandy billabongs. Line about the means illustrates the fitted, but non-significant, regression.

The small variation in dissimilarity values for fish communities in channel billabongs and the similarity of 2005 fish communities in shallow and channel billabongs to the respective communities found in previous years indicate there is no evidence of any adverse effects of mine waste waters arising from the Ranger minesite on fish communities of Magela Creek.

Potential impacts of cane toads

It is now three years since cane toads (*Bufo marinus*) invaded the area of billabong sampling sites on Nourlangie Creek and two years since they appeared at Magela Creek billabong sites. Possible effects of cane toads on fish communities are examined in figure 4 by comparing numbers of fish in the two catchments before and after cane toad arrival. The fish were grouped into three trophic guilds with different potential risk from the presence of toxic toad life stages: 'carnivores' (spangled grunter, mouth almighty, sleepy cod); 'benthic omnivores' (four species of eel-tailed catfish); and 'microphagic omnivores' (glassfish, rainbowfish, hardyheads). There is no evidence of a decline in any of the trophic groups following toad arrival. Consequently, it is concluded that there has been no measurable impact from cane toads on billabong fish communities. This result contributes valuable information to national assessments of risk to biodiversity posed by this invasive species. It also indicates that this invasive species is unlikely to confound assessments of mining impact at Ranger and supports the continued use of this monitoring technique for this role.

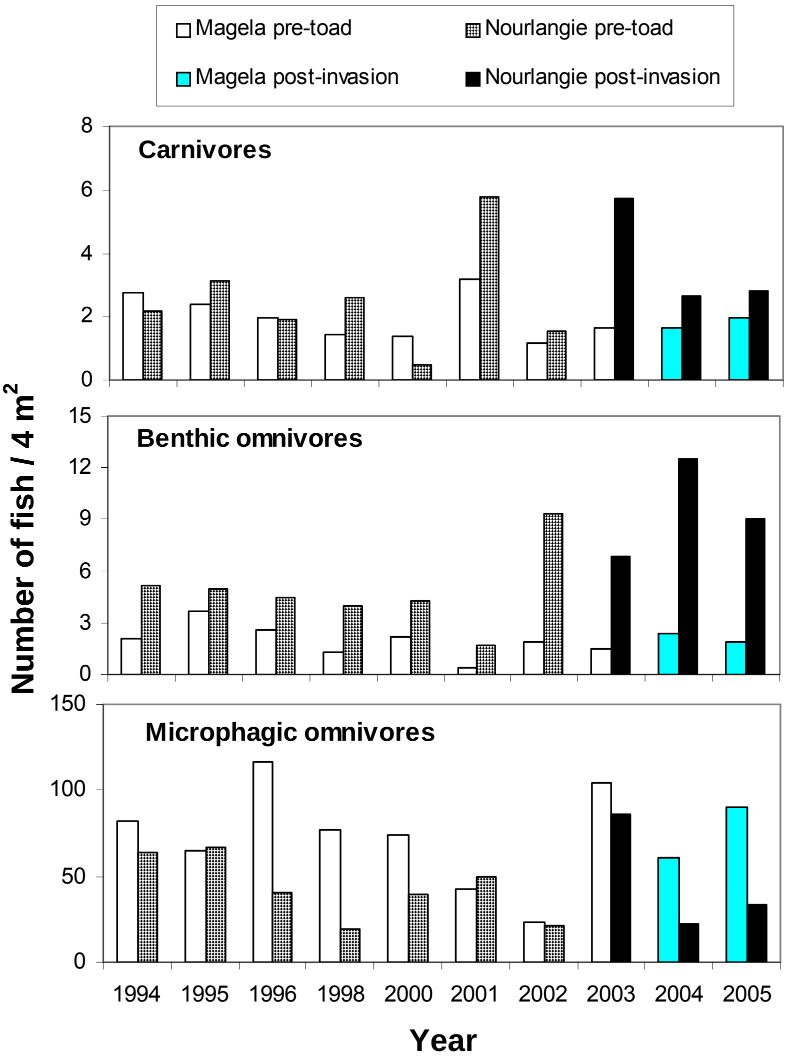

Figure 4 Temporal patterns of abundance of fish in shallow billabongs before and after the arrival of cane toads in two catchments, Nourlangie Creek and Magela Creek

Reference

Pidgeon R & Humphrey C 2006. Monitoring using fish community structure. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 46–47.

Publication of protocols for SSD's stream monitoring program in Magela Creek and Quality Control and Quality Assurance of SSD's stream monitoring program

C Humphrey

Background

Protocols for SSD's stream monitoring program are being prepared for publication. Progress in preparing these protocols was reported to ARRTC in late February 2005. Key summary issues provided to ARRTC included:

1. Six of eight protocols for environmental monitoring of streams associated with the Ranger mine site have been prepared for publication in the SSR series. A corresponding operational manual – working documents with full and more complete details of methods and procedures – is being prepared for each of the protocols.
2. Two protocols associated with bioaccumulation (in freshwater mussels and fishes) continue to be drafted following a review of the bioaccumulation program (see Sauerland et al 2006).
3. Data analysis aspects of the protocols are also currently under revision with improvements to data analysis approaches to be incorporated into the protocols prior to publication of the SSR.
4. Considering items 2 and 3, the protocols will be published in the first half of 2006.
5. QA/QC aspects of the protocols were reported to ARRTC separately (Humphrey et al 2005).

Regarding item 5, ARRTC was provided with the general principles applied to documenting QA/QC steps in SSD's environmental monitoring protocols and in implementing the steps in the routine conduct of the monitoring program. A risk analysis of critical steps in each protocol is being undertaken, the outcome of which will determine the level of auditing, internal and external, as well as training to devote to steps of the protocols. By way of example, such a risk analysis, combined with precedent and best-practice elsewhere, was undertaken for two of SSD's monitoring protocols and presented to ARRTC. The committee agreed that (i) QA/QC for SSD's monitoring program be developed according to this logic, (ii) the balance of internal and external QA/QC and training identified in the exemplary risk analyses was appropriate, and (iii) following publication of the protocols, QA/QC findings would be reported annually in an SSR of monitoring results and, as such, are embedded in the protocols, both in implementation and as an annual reporting outcome.

Progress to date

Statistical advice from CDU (Dr Keith McGuinness) has been sought for sections of the protocols dealing with design and analysis. This advice is being incorporated into the final drafting of protocols.

Reference

Humphrey C, Sauerland C & Cameron A 2005. QA/QC of the SSD's stream monitoring program. Report to Alligator Rivers Region Technical Committee,15th Meeting (28 February – 1 March 2005), Agenda Item 4.1.2.

Sauerland C, Ryan B, Humphrey C & Jones D 2006. Review of the bioaccumulation monitoring program. In ***eriss*** *research summary 2004–2005*. eds Evans KG, Rovis-Hermann J, Webb A & Jones DR, Supervising Scientist Report 189, Supervising Scientist, Darwin NT, 60–62.

Review of the bioaccumulation monitoring program

C Sauerland, B Ryan, C Humphrey & D Jones

Background

Detection of increases of metal and radionuclide concentrations in mussels and fish in Mudginberri Billabong downstream of Ranger could provide early warning of uptake, and hence bioavailability, of mine dispersed wastes. Measurements in fish and mussel tissues have been made since 1970. Since 2002, bioaccumulation data have also been gathered from a control site (Sandy Billabong).

A review of the bioaccumulation study of metals and radionuclides at Ranger is currently being undertaken with the aim of developing, as far as possible, a common metals and radionuclide sampling and analysis program, to avoid any unnecessary duplication between programs. As part of the review, all available bioaccumulation data for Ranger (mussels and fish, both billabongs) are being compiled and reported. The review includes an assessment of the data quality, and an assessment of whether any changes in metal/radionuclide concentrations are mining-related and culminates with the production of suitable protocols for continued monitoring.

Progress to date

The bioaccumulation of radionuclides and metals in freshwater mussels from Mudginberri and Sandy billabongs between 2000–2003 was reported by Ryan et al (2005a). A second report on the bioaccumulation of radionuclides and metals in freshwater mussels of the upper South Alligator River (Ryan et al 2005b) provides a dataset for comparison from another region in the Alligator Rivers Region.

All available bioaccumulation data for metals in fish and mussels collected between 1970–2003 from Mudginberri and Sandy billabongs by different organisations have been compiled and summarised in an inventory. These results will be reported shortly. As an example, figure 1 illustrates that uranium concentrations measured in the flesh of the forktail catfish (*Arius leptaspis*) and the freshwater mussel *Velesunio angasi* are generally consistent across different organisations, are generally low and similar for both billabongs, and have not significantly increased since mining started 1980 until 2003. The quality of SSD fish data currently being assessed against an extensive set of quality control samples taken in 2003 (replicate, blank and spiked samples; interlaboratory comparison).

Preliminary results from an additional and related review, which was conducted as part of the trophic transfer study (various biota), were reported by van Dam et al (2005) (see summary for ARRTC KKN 1.2.1). The current review and the aquatic pathway section of the trophic transfer study will be merged.

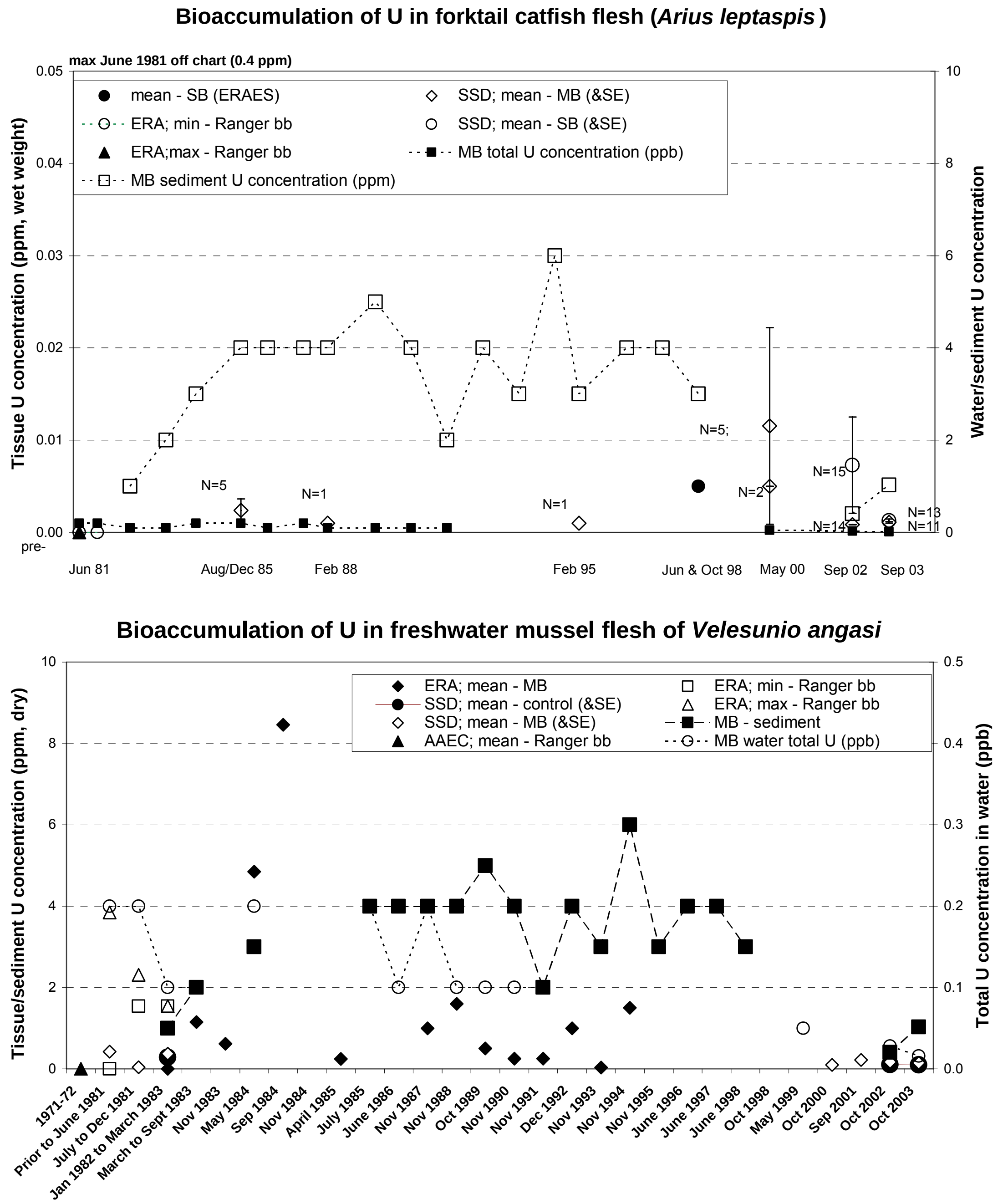

Figure 1 Mean or minimum and maximum values of uranium concentrations in flesh of the forktail catfish and the freshwater mussel *Velesunio angasi* collected in Sandy Billabong (SB), Mudginberri Billabong (MB) or other billabongs on the Ranger lease (Ranger bb) between 1980–2003 by the Supervising Scientist Division (SSD), Energy Resources of Australia (ERA) and in 1971–1972 by the Australian Nuclear Science and Technology Organisation (previously AAEC), and uranium concentrations in water and sediment. SE = standard error and N = number of samples.

A small workshop was conducted in September 2005 to report on progress made towards meeting the objectives of the bioaccumulation review. The outcomes of the workshop are currently being compiled for reporting. Summary items arising from the review included:

- Metal and radionuclide body burdens in mussels and fishes from Mudginberri Billabong are generally low, with no evidence of trends through time.
- Average metal concentrations of fish and mussel flesh samples are well below the maximum allowable levels for human consumption (Commonwealth of Australia 2005).

Copper and zinc average concentrations in fish tissues are above the generally expected levels (Commonwealth of Australia 2001) at both the 'impact' and control site reflecting the naturally higher metal and radionuclide levels in the Alligator Rivers Region

- Concentrations of metals and radionuclides in mussels from Mudginberri Billabong appear to be related to age, growth rates and location (viz sediment quality). The need to better characterise sediment quality was recognised for future sampling.
- Metal concentrations in fish are not related to fish size or weight but may vary by orders of magnitude dependent on the type of organ. Calcium, magnesium, manganese, strontium, barium, lead and uranium are preferentially incorporated into bones while antimony, selenium, mercury, tin and zinc are highest in liver samples.
- Concentrations of radionuclides and a number of metals in mussels from Mudginberri Billabong are generally higher than mussels from Sandy Billabong. Naturally higher catchment concentrations of constituent elements in Magela Creek compared with Nourlangie Creek catchment are likely to be the cause, together with lower concentrations of calcium in Mudginberri Billabong waters compared with Sandy (likely to contribute higher radium concentrations at least, in Mudginberri mussels).
- A stream-lined bioaccumulation sampling program was devised for freshwater mussels and fish covering radionuclides and metals. Targeted species are *Arius leptaspis* and *Velesunio angasi* as they are the high-level bioaccumulators and their monitoring data set is the most extensive.
- The higher concentrations of radionuclides and a number of metals in mussels from Mudginberri Billabong compared with mussels from Sandy Billabong warrants additional sampling of mussels in Magela Creek upstream of Ranger to determine the extent to which the Ranger mine contributes to the total concentrations of elements in Mudginberri mussels.

References

Commonwealth of Australia 2001. *Generally Expected Levels (GELs) for Metal Contaminants, Additional guidelines to maximum levels in Standard 1.4.1*, Published by Anstat Pty Ltd, Canberra, Australia.

Commonwealth of Australia 2005. *Australia and New Zealand Food Standards Code: Standard 1.4.1: Contaminants and natural toxicants, incorporating amendments up to and including Amendment 81*, Published by Anstat Pty Ltd, Canberra, Australia.

Ryan B, Martin P, Humphrey C, Pidgeon R, Bollhöfer A, Fox T & Medley P 2005a. Radionuclides and metals in fish and freshwater mussels from Mudginberri and Sandy Billabongs, Alligator Rivers Region, 2000–2003. Internal Report 498, November, Supervising Scientist, Darwin. Unpublished paper.

Ryan B, Martin P, Humphrey C, Iles M, Bollhöfer A & Fox T 2005b. Radionuclides and metals in freshwater mussels of the upper South Alligator River. Internal Report 487, February, Supervising Scientist, Darwin. Unpublished paper.

van Dam R, Humphrey C, Sauerland C, Ryan B, Bollhöfer A & Wagner K 2005. A review and risk assessment of bioaccumulation and trophic transfer of key metals associated with mining in the Alligator Rivers Region, Northern Australia. In 8^{th} *international conference of ICOBTE*, Adelaide, 3–7 April 2005. Abstracts Adelaide, CSIRO Land and Water, 590–600.

Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

C Sauerland, P Medley & J Sellwood

Introduction

Surface water samples in the vicinity of the Ranger and Jabiluka project areas are regularly monitored for radium-226 (^{226}Ra) to assess a change, if any, in the ^{226}Ra activity concentration downstream of the mine sites and the potential risk of increased exposure to radiation via the biophysical pathway due to mining activities.

A limit of 10 mBq/L as increase above natural background in total ^{226}Ra concentration in surface waters downstream of Ranger has been defined for human radiological protection purposes (Klessa 2001) and is based on the potential dose received from the ingestion of ^{226}Ra in the freshwater mussel *Velesunio angasi* (Martin et al 1998). This limit of 10 mBq/L is applied to the wet season median difference (Sauerland et al 2005) (figure 1). It is calculated by taking the median values at both the downstream and upstream sites for each wet season, and then substracting the upstream median from the median at the downstream site.

Methods

Water samples are collected weekly in Magela creek and monthly in Ngarradj on both upstream and downstream sites of the project areas according to the surface water monitoring protocol (Sauerland & Iles 2005). All Ngarradj samples and Magela creek samples from alternate weeks are analysed for total ^{226}Ra (ie combined filtered and particulate fraction) following a method described in Medley et al (2005). The remaining fortnightly samples for Magela Creek are combined into two wet season composite samples, one for the upstream site and one for the downstream site samples.

Results

Magela Creek

^{226}Ra results for the 2004–2005 wet season are compared in Figure 1 to wet season data from 2001–2004. The data shows very low levels of ^{226}Ra in Magela Creek. A paired two-tailed t-test (95% confidence interval, N = 45, T-Value = 1.04, P-Value = 0.304, Power = 0.15) indicates that the ^{226}Ra concentrations are not significantly different at the Magela creek downstream and upstream sites between 2001 and 2005. Moreover, Figure 1 shows that total ^{226}Ra activity concentrations at the upstream site can be higher than at the downstream site. The ^{226}Ra concentration of 8.8 mBq/L in a sample collected on the 15th of February 2005 at the upstream site is most likely due to a larger contribution of fine sediments than are present naturally in the Magela creek catchment (Murray et al 1993).

Figure 2 illustrates the distribution of total ^{226}Ra activity concentrations in Magela Creek and Ngarradj. Values are mostly situated between 1 and 5 mBq/L but outliers can be as high as

11 mBq/L. The analysis of chemical blanks, duplicates and spiked blanks (not shown here) indicates that outliers were unlikely to be due to contamination during the high-resolution alphaspectrometric procedures. For example, the results of duplicate analysis were always within the range of their intrinsic counting error that is due to counting statistics (indicated by error bars in figure 1 and figure 3).

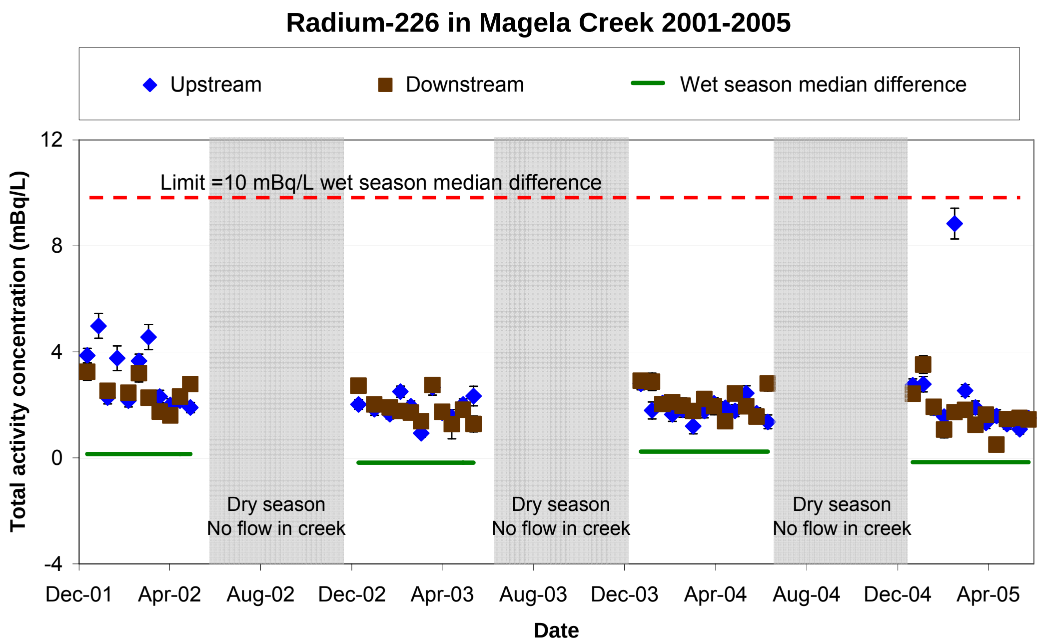

Figure 1 Time series of total radium-226 activity concentrations in Magela Creek (2001 to 2005); error shown is based on counting statistics only

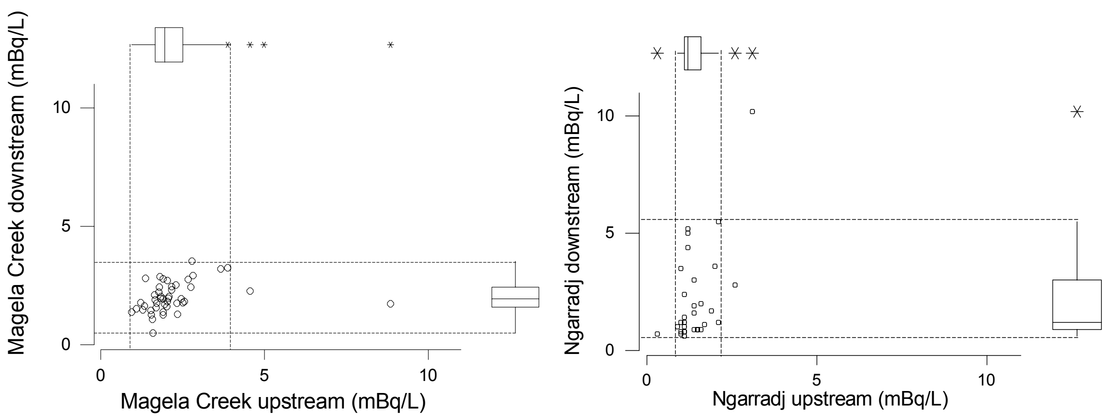

Figure 2 Scatter plot for total ^{226}Ra activity concentrations in Magela Creek and Ngarradj with boxplots in each direction in the margins of the x- and y-axes (rectangular boxes represent roughly the middle 50% of the data; asterisks represent outliers; dashed lines indicate the general extent of data.

Table 1 shows the median and standard errors for individual wet seasons and 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 (eg in 2004–05 the '*wet season composite sample*' from Magela Creek was 2.3 ± 0.1 mBq/L for duplicates from both the upstream and downstream site).

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

In Magela Creek		All years	2001–02	2002–03	2003–04	2004–05
Median and standard error	upstream	2.0 (± 0.2)	2.3 (± 0.3)	2.0 (± 0.1)	1.8 (± 0.1)	1.7 (± 0.6)
	downstream	2.0 (± 0.1)	2.5 (± 0.2)	1.8 (± 0.2)	2.0 (± 0.1)	1.6 (± 0.2)
Wet season median difference		N/A	0.2	- 0.2	0.2	- 0.1
In Ngarradj		All years	2001–02	2002–03	2003–04	2004–05
Median and standard error	upstream	1.2 (± 0.1)	1.2 (± 0.2)	1.4 (± 0.2)	1.1 (± 0.1)	1.3 (± 0.1)
	downstream	1.2 (± 0.3)	3.0 (± 0.9)	1.1 (± 0.5)	0.9 (± 0.3)	1.0 (± 0.3)
Wet season median difference		N/A	1.8	- 0.3	- 0.2	- 0.3

Based on the available data, the wet season median difference for all wet seasons from 2001 to 2005 is approximately zero (see table 1). The available data for the four sampling seasons indicate that ^{226}Ra levels in Magela Creek are due to the natural occurence of radium in the environment and that radium originating from the Ranger mine has not caused any impact on human health.

Ngarradj

^{226}Ra results in Ngarradj are available for the 2001–02, 2002–03, 2003–04 and 2004–05 wet seasons. The data shows that ^{226}Ra levels are very low in Ngarradj (see table 1). A paired two-tailed t-test (95% confidence interval, N=34, T-Value = -2.36, P-Value = 0.024, Power = 0.83) implies that ^{226}Ra concentrations are significantly different between the Ngarradj downstream and upstream sites between 2001 and 2005. An increase in ^{226}Ra activity concentrations due to the Jabiluka project can therefore not be excluded. However, it is more likely that the difference in downstream and upstream activity concentrations measured in 2001–02 were due to a late dry season fire that affected only the catchment downstream of the upstream sampling site (Evans et al 2004).

Figure 3 shows that ^{226}Ra activity concentrations at the Ngarradj downstream site were similar to those at the upstream site since December 2003, coinciding with the establishment of the long-term care and maintenance phase at Jabiluka in the 2003 dry season.

The wet season median difference (figure 3) in the three sampling seasons is very low, indicating that human health was not adversely affected by the presence of ^{226}Ra in Ngarradj from the Jabiluka project.

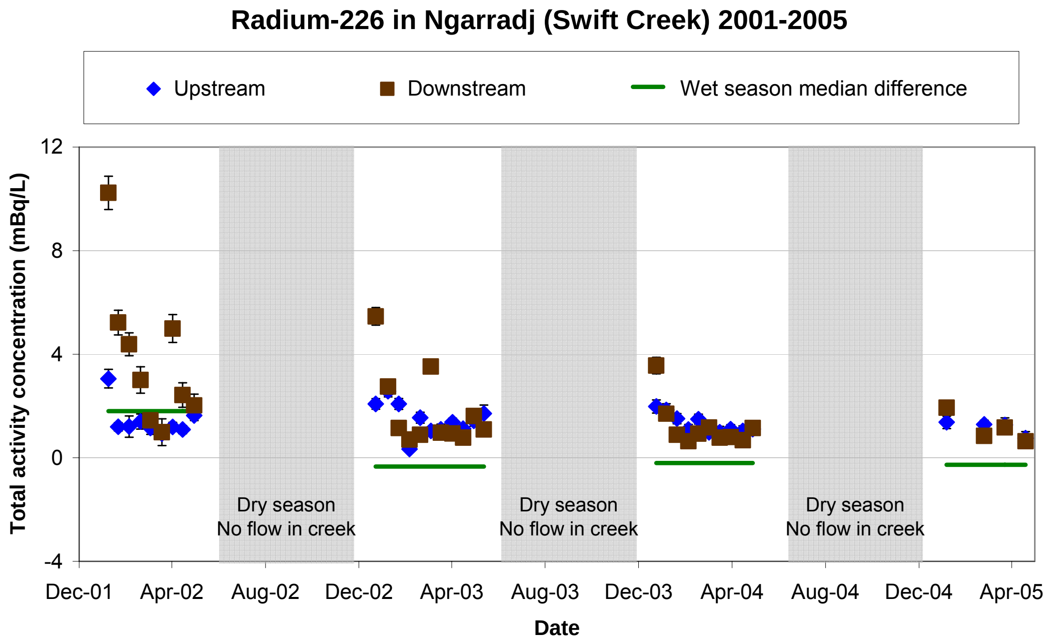

Figure 3 Time series of total radium-226 activity concentrations in Ngarradj (2001 to 2005); error shown is based on counting statistics only

References

Evans KG, Moliere DR, Saynor MJ, Erskine WD & Bellio MG 2004. *Baseline suspended-sediment, solute, EC and turbidity characteristics for the Ngarradj catchment, Northern Territory, and the impact of mine construction*. Supervising Scientist Report 179, Supervising Scientist, Darwin NT.

Klessa D 2001. Water quality in Magela Creek upstream and downstream of Ranger: A summary of performance for 2000–2001 and derived triggers and limits for 2001–2002. Internal Report 380, Supervising Scientist, Darwin. Unpublished paper.

Martin P, Hancock GJ, Johnston A & Murray AS 1998. Natural-series radionuclides in traditional north Australian Aboriginal foods. *Journal of Environmental Radioactivity* 40, 37–58.

Medley P, Bollhöfer A, Iles M, Ryan B & Martin P 2005. Barium sulphate method for radium-226 analysis by alpha spectrometry. Internal Report 501, June, Supervising Scientist, Darwin. Unpublished paper.

Murray AS, Johnston A, Martin P, Hancock G, Marten R & Pfitzner J 1993. Transport of naturally occurring radionuclides by a seasonal tropical river, northern Australia. *Journal of Hydrology* 150, 19–39.

Sauerland C, Martin P & Humphrey C 2005. Radium 226 in Magela creek, northern Australia: Application of protection limits from radiation for humans and biota. *Radioprotection* Suppl. 1, 40, S451–S456.

Sauerland C & Iles M 2005. Environmental monitoring protocols used to assess potential impacts upon streams associated with the Ranger mine site, Surface Water Chemistry Monitoring. Contribution to: Draft Supervising Scientist Report, Supervising Scientist, Darwin. Unpublished paper.

Surface water transport of uranium in the Gulungul catchment

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Introduction

The aim of this project is to investigate the cause of elevated concentrations of uranium and some ions that were detected at the Gulungul Creek downstream monitoring site over the month of January 2004 and, to a lesser extent, in previous wet seasons. These concentrations, higher than those at the upstream site, coincided with higher EC values and lower pH values. The study is part of an Honours project in collaboration with Charles Darwin University, and will integrate radioanalytical, physico-chemical and hydrological transport aspects relating to contaminant transport in the Gulungul catchment.

Methods

During a field trip in the 2004 dry season, it was observed that the overland flow from the Tailings Dam South Road Culvert (TDSRC) results in a visible channel with wetland vegetation that flows into Gulungul Creek just upstream of the midstream monitoring site, GCMS. Consequently, three field trips were undertaken in the 2005 wet season (8 February, 18 March & 10 May) during which samples were taken in the Gulungul Creek catchment: upstream (GCUS), downstream (GCDS) and at several locations in the vicinity of the midstream site (GCMS), ie 'GCMS–10 m', GCMS–50 m, 'GCMS+50 m' and 'GCMS+150 m'. Samples were also taken from V-notches at TDSRC, the overland flow from TDSRC (TDSRC flow), a spring tributary flow (Spring) and from a swampy area of another suspected spring (Lower Spring).

Metal analyses of the samples, including uranium, were conducted via ICPMS at Charles Darwin University. Uranium concentration and activity ratios ($^{234}U/^{238}U$) were measured via alpha spectrometry at ***eriss*** (Martin & Hancock 2004) in order to identify uranium activity ratios of contaminating endmember(s).

Activity ratios measured downstream will be compared with ratios modelled via a mixing of various sources. Samples collected during the field trips in 2005 will be analysed as well as selected samples that were collected as part of the routine surface water quality monitoring between 2003 and 2005.

Results

Figure 1 illustrates the variations in uranium and sulfate concentrations in the samples collected from Gulungul creek and TDSRC flows over the last three wet seasons.

Review of selected concentrations for samples collected on 18 March 2005 shows little difference in concentrations among the samples collected in the vicinity of the midstream

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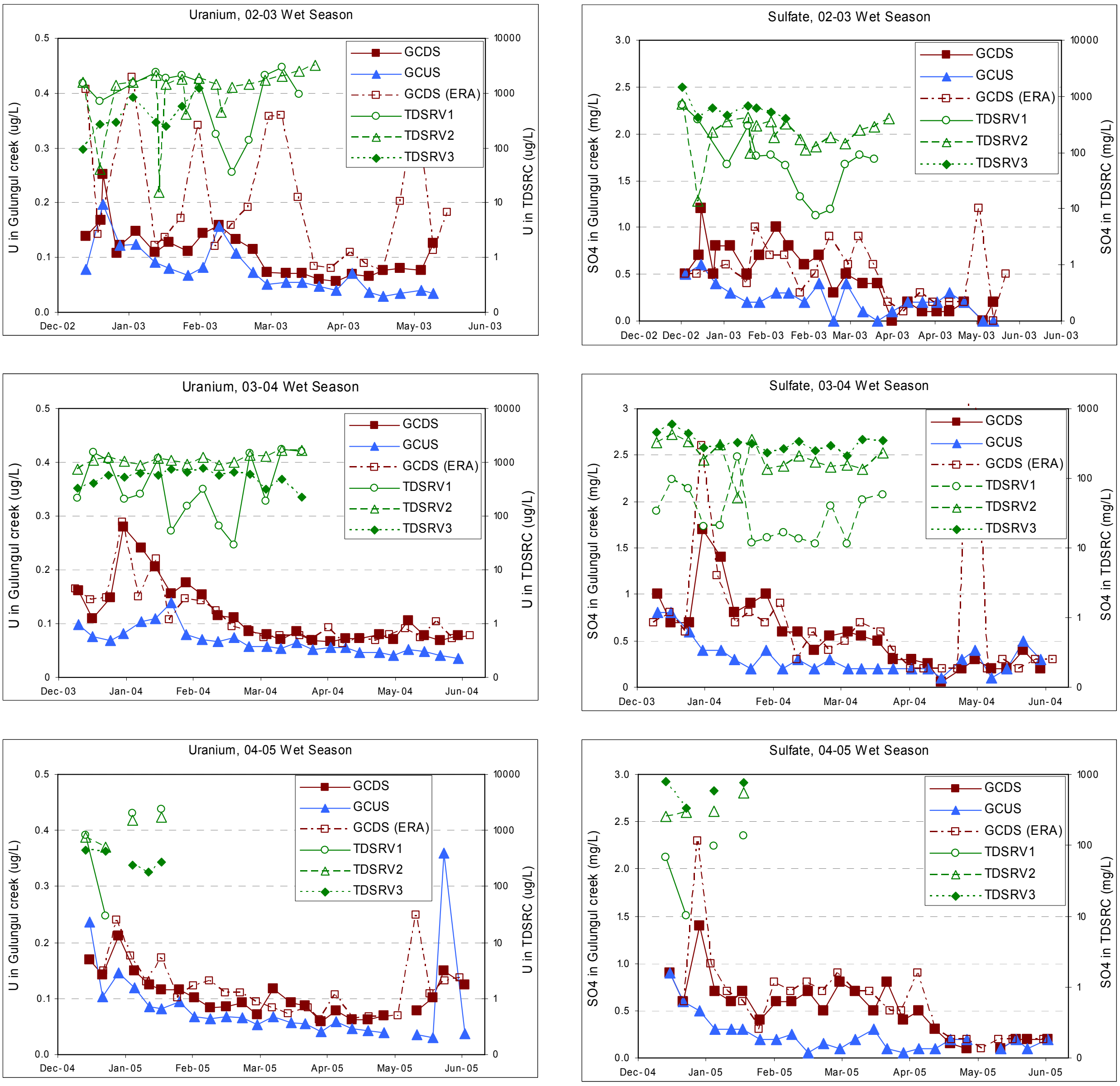
location and at GCDS, indicating that the flow from the contaminant source may enter the creek over a larger area most likely upstream of the sampling points adjacent to GCMS (table 1). Key indicators for Spring and TDSRC, also noticeably elevated at GCDS, are copper, sulfate and uranium. The difference in the indicators' concentrations between GCDS and GCUS confirm that both, Spring and TDSRC, are potential contamination sources, but the Spring's location downstream of GCMS makes it an unlikely source for the increase observed at GCDS on this field trip.



F igure 1 Time series of U and SO_4 concentrations at Gulungul Creek upstream (GCUS) and downstream (GCDS), and TDSRC V-notches between 2002 and 2005. GCDS data taken by ERA are shown as comparison.

During the field trip on 10 May 2005 however, several other dry tributaries entering the creek could be identified upstream of GCMS, which may originate from TDSRC flow, implying that previous sampling may not have extended far enough upstream. Once flow data for the TDSRC V-notches is received from ERA, a comparison of loads of the key indicators over all wet seasons since 2000 may indicate if TDSRC flow indeed contributed significantly to the observed increases between upstream and downstream sites.

Table 1 EC, pH, selected metals and major ions for samples collected 18/03/2005

Site	EC (µS/cm)	pH	As (µg/L)	Cu (µg/L)	Mg (µg/L)	Mn (µg/L)	SO_4 (mg/L)	U (µg/L)
Spring	130	6.1	0.2	0.16	12.4	11.6	53.1	5.92
TDRSC-V2	380	7.4	0.3	5.67	49.8	2490	168	1150
TDSRC-V3	693	7.5	0.25	1.32	92.1	1100	390	743
TDSRC flow at road	56	6.6	0.15	0.29	5.1	5.36	17.8	0.858
TDSRC flow at GC [1]	28	6.1	0.1	0.23	2.2	2.6	7.4	0.383
GCUS	10	5.7	<0.05	0.05	0.5	3.14	0.3	0.057
GCMS -10m [2]	12	5.8	<0.05	0.08	0.8	2.96	1.3	0.105
GCMS +50m	12	5.9	0.05	0.12	0.7	3.02	0.9	0.093
GCDS	12	6.0	0.05	0.10	0.7	2.55	1.0	0.105

1 collected about 100 m before entering the creek;

2 10 m upstream from GCMS and 60 m upstream from 'GCMS+50m';

Note: Cd, Hg, Pb Th and Zn were close or below detection limits in all samples

Uranium concentration and uranium activity ratios ($^{234}U/^{238}U$) have been measured via alpha spectrometry, in order to identify activity ratios of possible contaminating endmember(s). The uranium activity ratio analysis of samples from the routine monitoring program focuses on samples collected during January to March 2004, February 2003 and February 2005.

Some difficulty in the iron oxide precipitation step as described in Martin and Hancock (2004) was encountered resulting in low uranium recoveries. A slightly modified procedure provided the recovery and peak resolution required for uranium isotope ratio analysis. The effect of the sample matrix of Gulungul Creek waters is being investigated further by analysing spiked samples.

Reference

Martin P & Hancock GJ 2004. *Routine analysis of naturally occurring radionuclides in environmental samples by alpha-particle spectrometry*. Supervising Scientist Report 180, Supervising Scientist, Darwin NT.