Atmospheric radiological monitoring in the vicinity of Ranger and Jabiluka

A Esparon & A Bollhöfer

Introduction

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

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

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

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

Results

Radon pathway

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

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

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

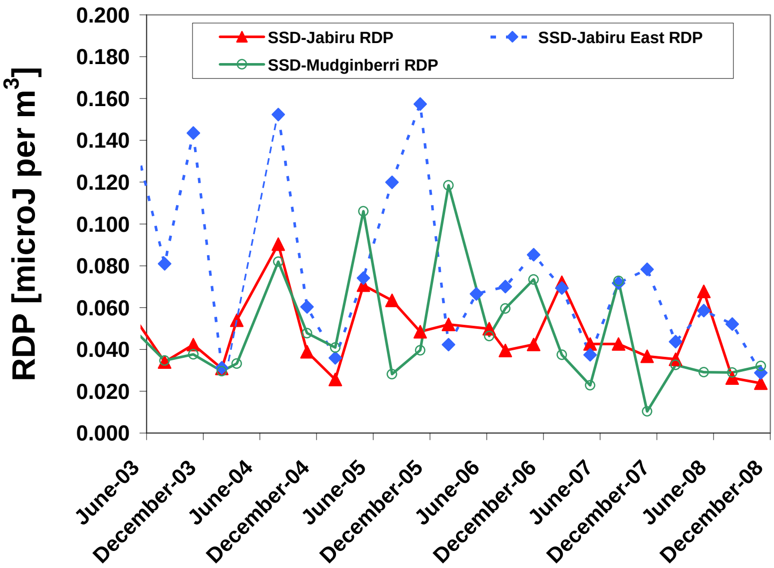

Figure 2 Radon decay product concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

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

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

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

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

[a] predicted from wind field model

Dust pathway

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

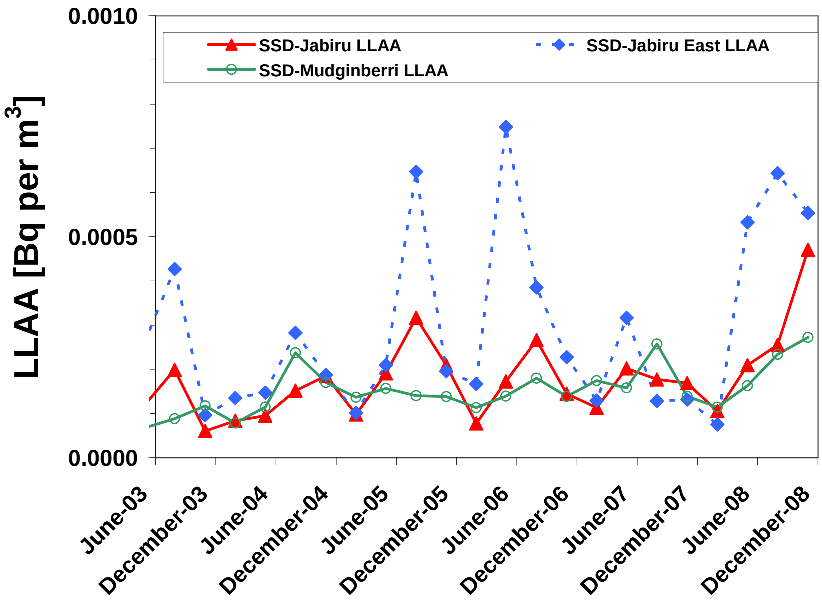

Figure 3 Long lived alpha activity concentration measured by SSD at Jabiru, Jabiru East and Mudginberri

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

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

Steps for completion

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

Summary

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

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Monitoring of radionuclides in groundwater at Ranger

B Ryan

Introduction

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

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

Results

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

Steps for completion

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

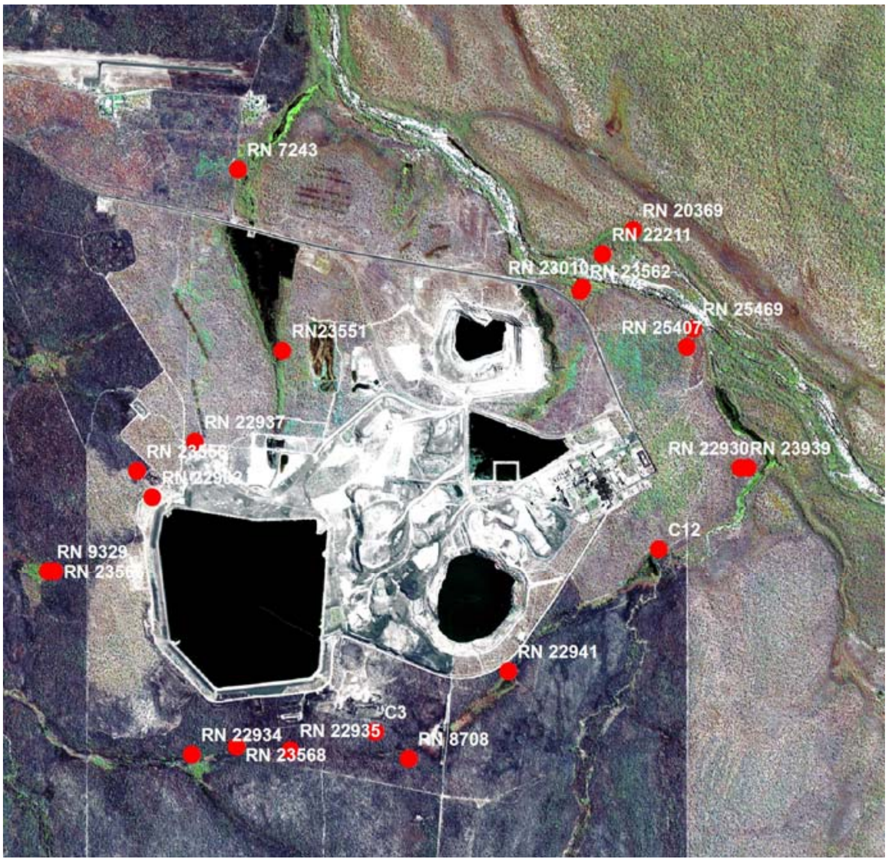

Figure 1 DRDPIFR monitoring bore sampling sites

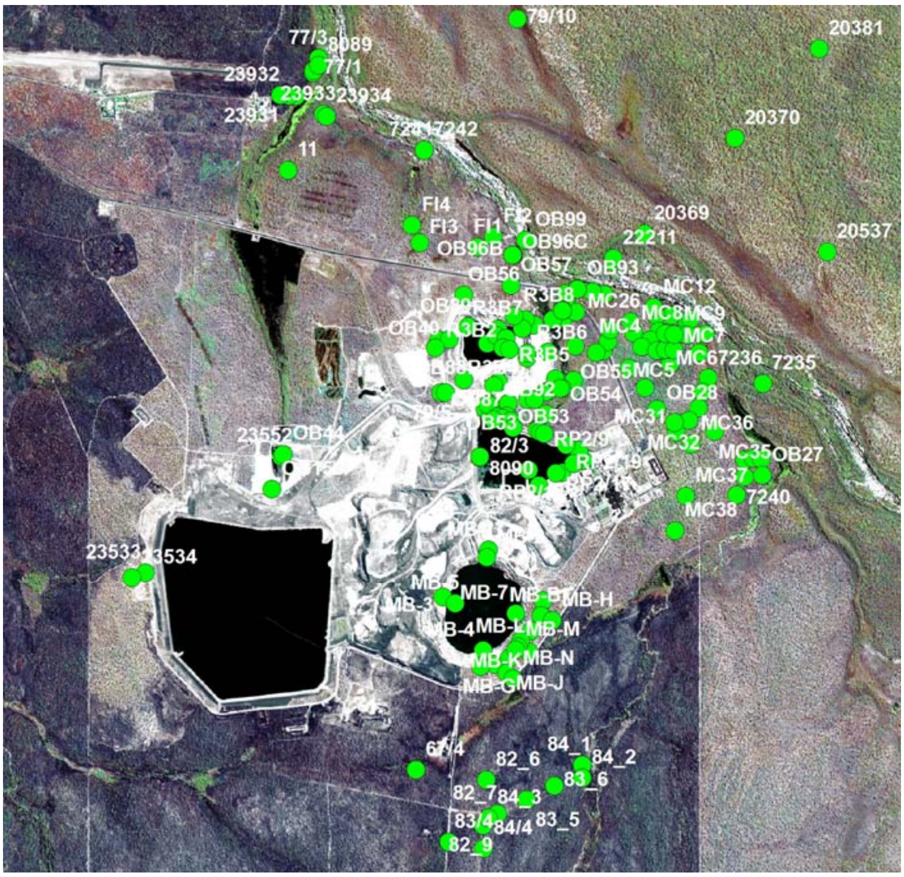

Figure 2 Ranger uranium mine monitoring bore sites associated with Pit 3

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

Acknowledgments

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

Surface water radiological monitoring in the vicinity of Ranger and Jabiluka

P Medley, A Bollhöfer & J Brazier

Introduction

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

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

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

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

Progress to date

Magela Creek

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

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

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

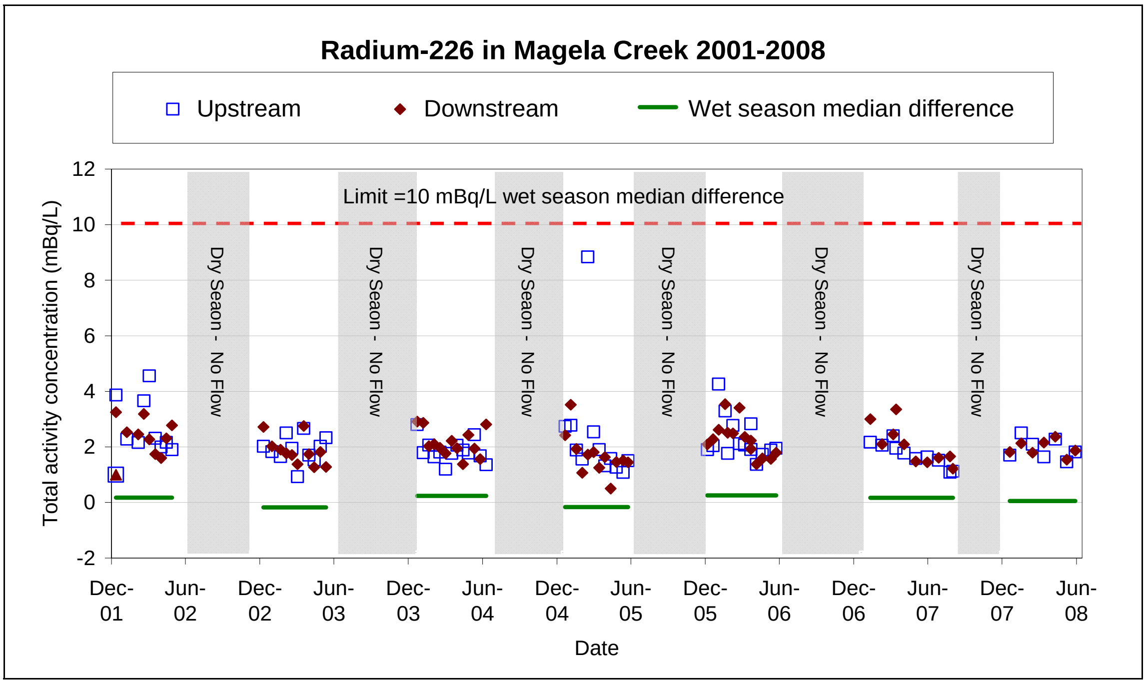

Figure 1 Radium-226 in Magela Creek for the 2001–08 wet seasons

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

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

Ngarradj Creek

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

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

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

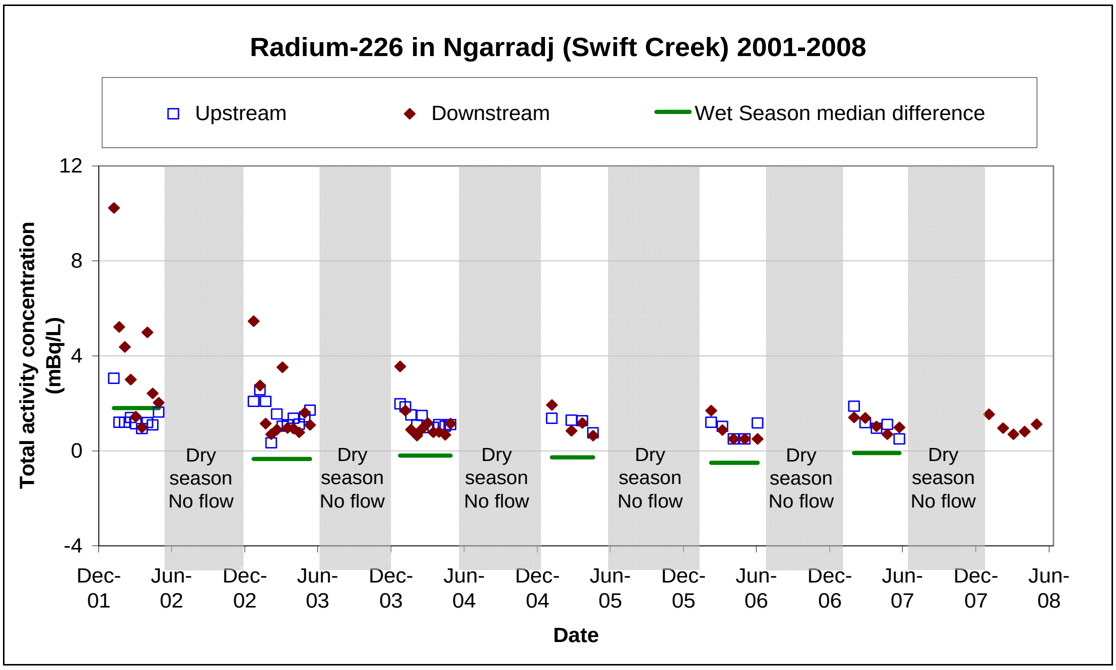

Figure 2 ^{226}Radium in Ngarradj for the 2001–08 wet seasons

Steps for completion

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

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

P Medley & K Turner

Introduction

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

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

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

Method

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

Magela Creek

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

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

Gulungul

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

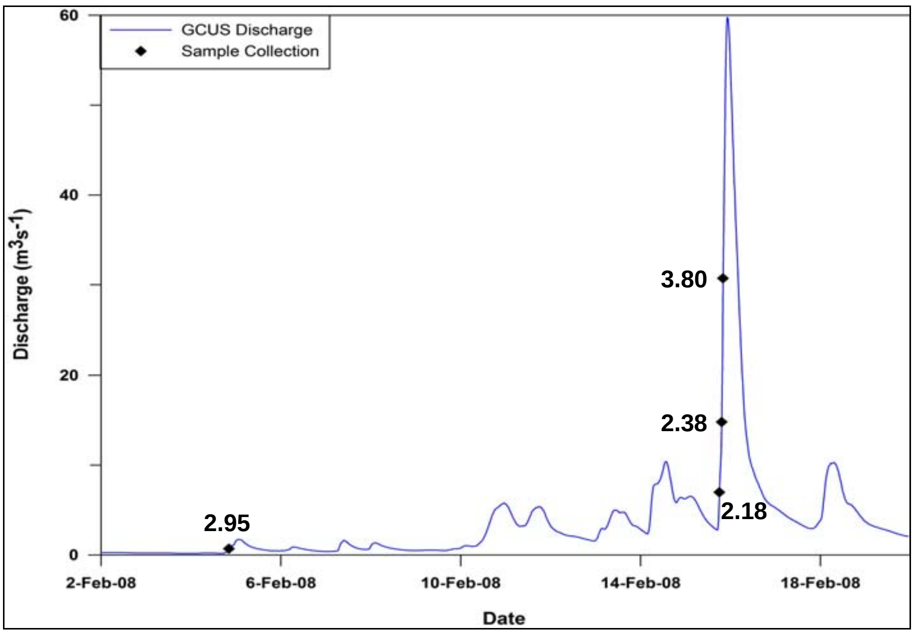

Figure 1 Flow hydrograph of Gulungul Creek, showing discharge at the upstream site, sample collection events and ^{238}U activity concentration [mBq·l^{-1}] measured in the samples

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

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

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

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

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

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

Steps for completion

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

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

Introduction

C Humphrey, A Bollhöfer & D Jones

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

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

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

Early detection of short or longer-term changes

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

Assessment of changes in biodiversity

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

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

Chemical and physical monitoring

J Brazier

Routine weekly sampling program in Magela Creek

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

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

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

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

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

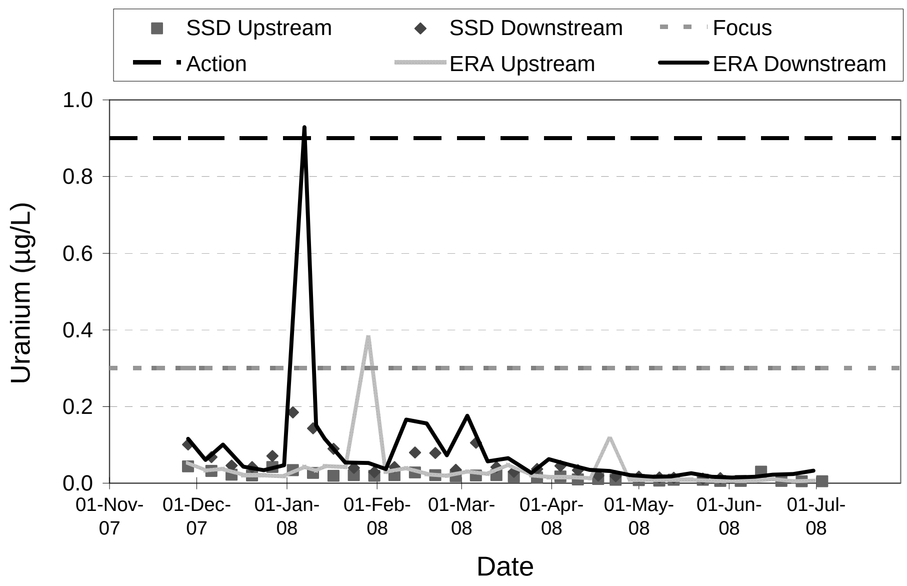

Figure 1 Uranium concentrations measured in Magela Creek by SSD and ERA between November 2007 and July 2008

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

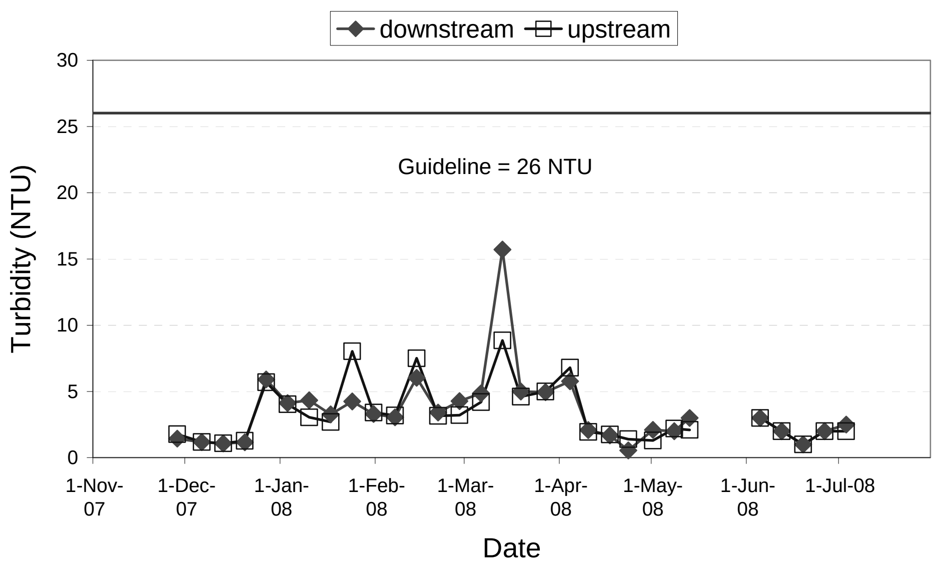

Figure 2 Turbidity concentrations in Magela Creek (SSD data) between November 2007 and July 2008

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

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

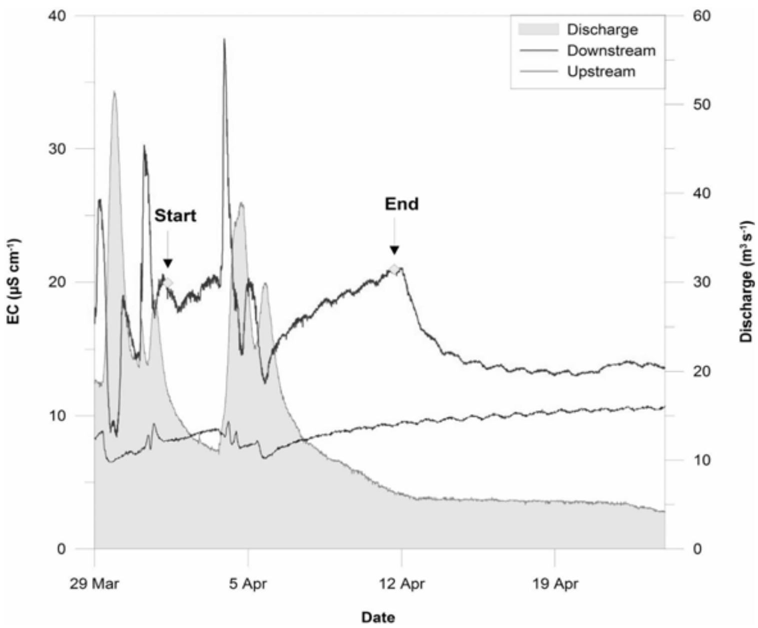

Figure 3 Continuous monitoring of EC and stream discharge in Magela Creek April 2008. Period of RP1 siphoning indicated by 'Start' and 'End' shown on EC trace.

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

Chemical and physical monitoring of Gulungul Creek

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

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

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

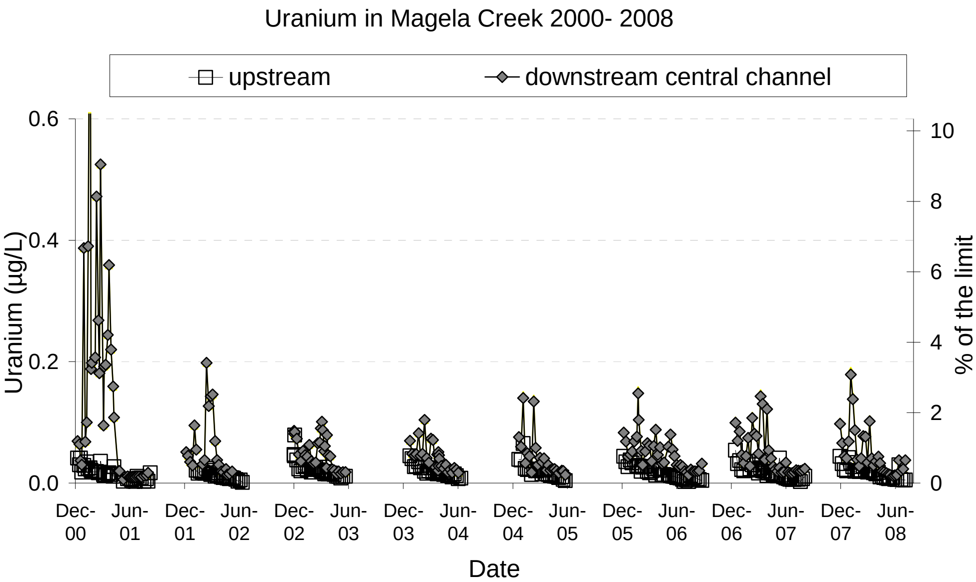

Figure 4 Uranium concentrations in Magela Creek since the 2000–01 wet season (SSD data)

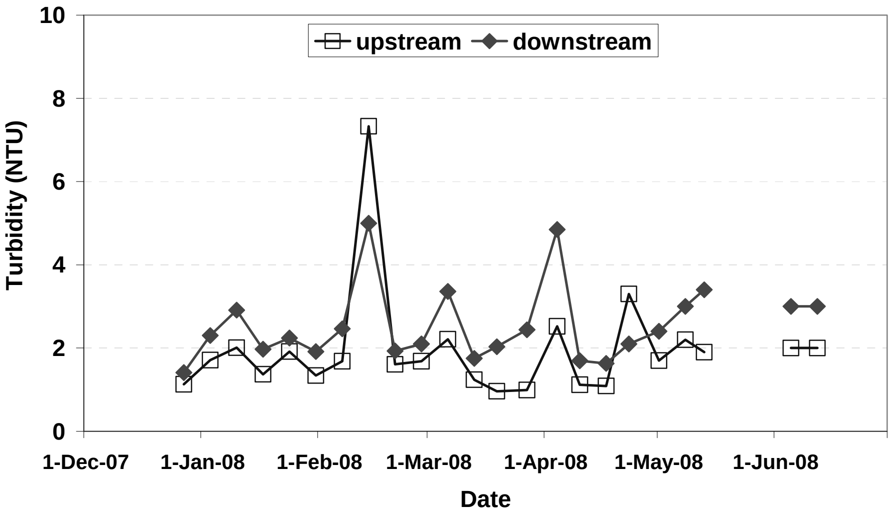

Figure 5 Turbidity measurements in Gulungul Creek for the 2007–08 wet season

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

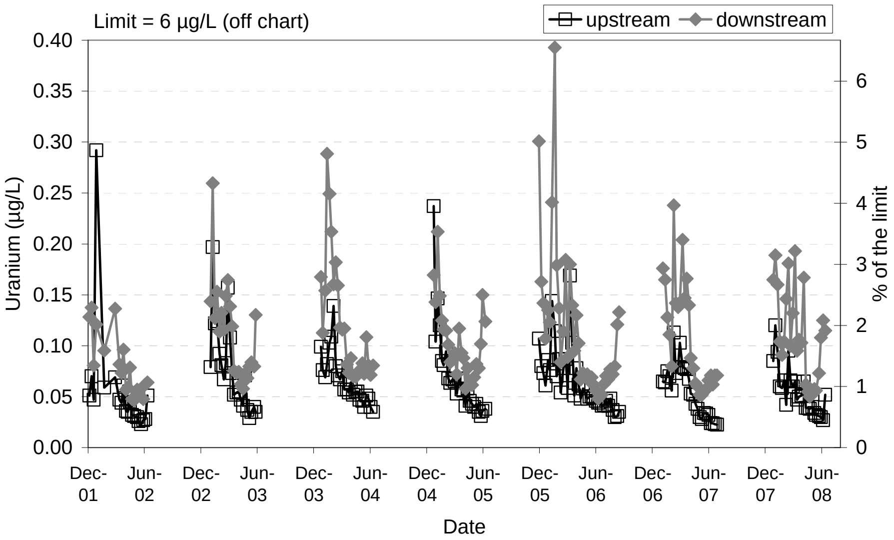

Figure 6 Uranium concentrations in Gulungul Creek between 2000 and 2008 (SSD data)

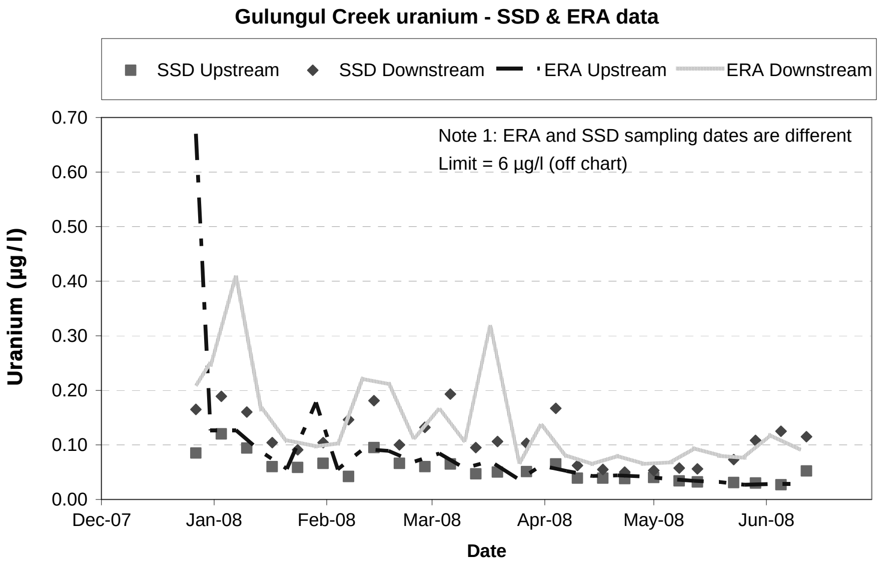

Figure 7 Uranium concentrations measured in Gulungul Creek by SSD and ERA during the 2007–08 wet season

References

Iles M 2004. Water quality objectives for Magela creek – revised November 2004. Internal Report 489, December, Supervising Scientist, Darwin. Unpublished paper.

Toxicity monitoring in Magela Creek

C Humphrey, C Davies & D Buckle

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

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

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

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

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

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

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

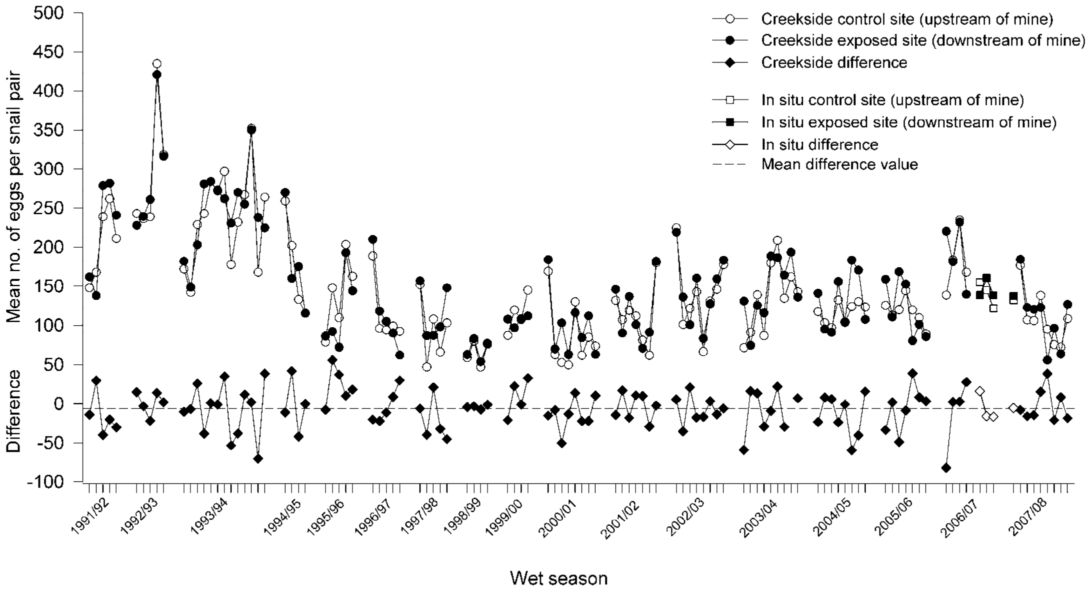

Figure 1 Creekside monitoring results for freshwater snail egg production for wet seasons between 1992 and 2008. Note the last three tests in 2006–07 and the first test for 2007–08 used the in situ monitoring method only.

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

References

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

Bioaccumulation in fish and freshwater mussels from Mudginberri Billabong

J Brazier, A Bollhöfer, B Ryan & C Humphrey

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

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

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

Bioaccumulation of uranium and radium in freshwater mussels

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

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

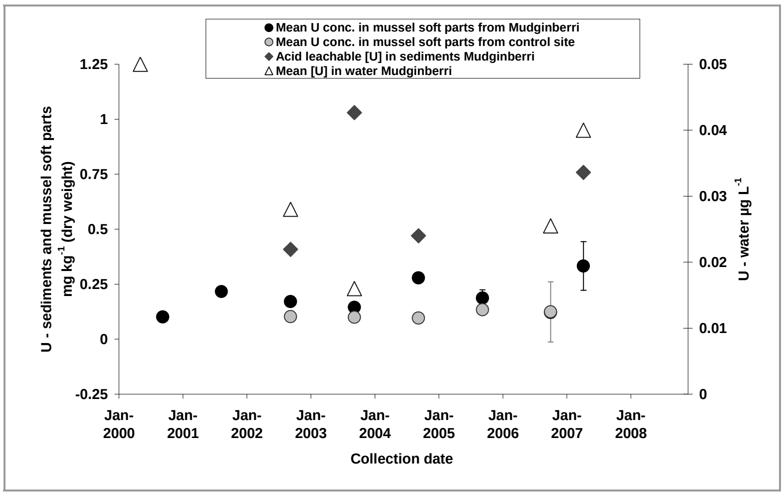

Figure 1 Mean concentrations of U measured in mussel soft-parts, sediment and water samples collected from Mudginberri Billabong and control billabongs since 2000

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

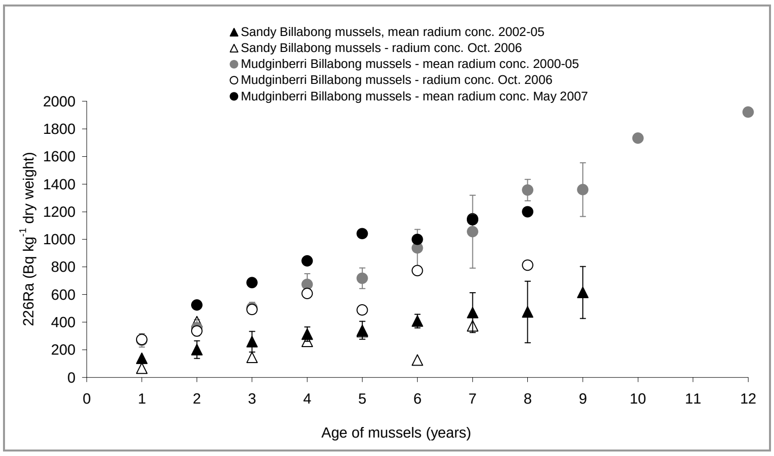

Figure 2 ^{226}Ra activity concentrations in the dried flesh of freshwater mussels collected from Mudginberri Billabong 2000–2007 and Sandy Billabong 2002–2006. The error bars are ± 1 standard deviation.

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

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

Bioaccumulation of uranium in fish

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

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

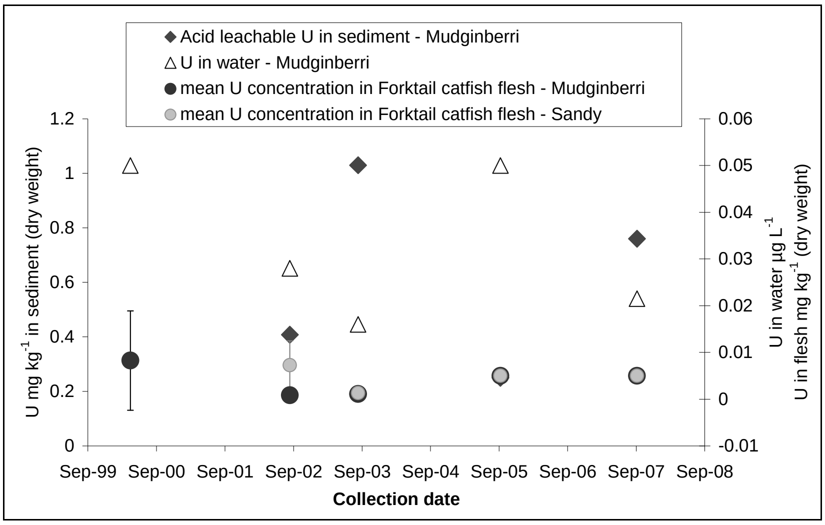

Figure 3 Mean concentrations of U measured in the flesh of forktail catfish, sediment and water samples collected from Mudginberri and Sandy Billabongs, since 2000. Error bars represent standard error. The lack of error bars in some years for forktail flesh samples indicates that the uranium concentration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

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ICRP 1996. *Age-dependent doses to members of the public from the intake of radionuclides: part 5. Compilation of ingestion and Inhalation dose coefficients*. ICRP Publication 72.

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Monitoring using macroinvertebrate community structure

C Humphrey, L Chandler & J Hanley

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

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

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

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

Thus for the first time, the paired-site dissimilarity indices are plotted by way of replicate site data (Figure 1) while multi-factor ANOVA has been extended to include analysis of the replicate (paired-site) dissimilarity values. For ANOVA, only data gathered since 1998 have been used. Data gathered prior to this time were based upon different and less rigorous sampling and sample processing methods, and/or absence of sampling in three of the four streams.

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

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

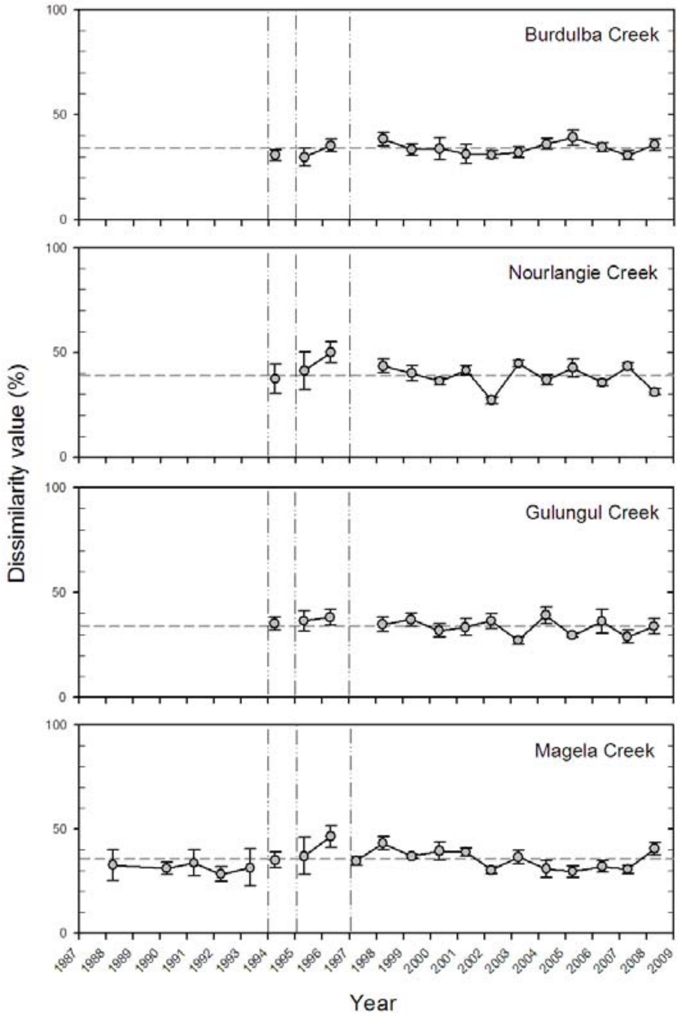

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2008. The dashed vertical lines delineate periods for which a different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

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

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

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

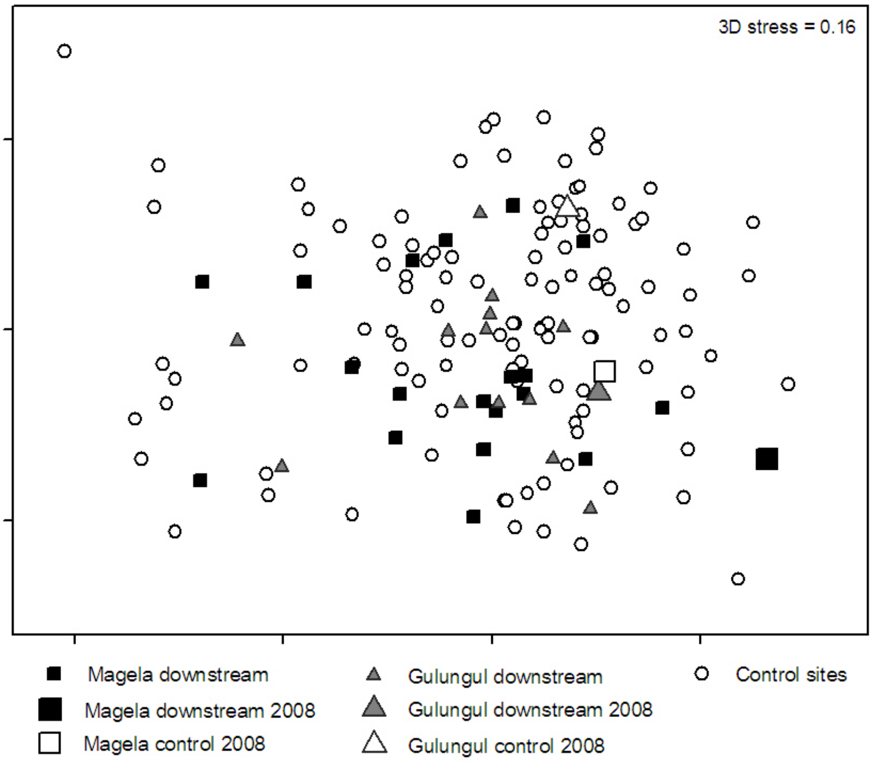

Figure 2 Axis 1 & 2 of a three dimensional ordination plot of macroinvertebrate communities sampled from sites in several streams in the vicinity of Ranger mine for the period 1988 to 2008. Data from Magela and Gulungul Creeks for 2008 are indicated by the larger symbols.

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

Reference

Supervising Scientist 2004. *Annual Report 2003–2004*. Supervising Scientist, Darwin.

Monitoring using fish community structure

D Buckle, C Humphrey & C Davies

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

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

Channel billabongs

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

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

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

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

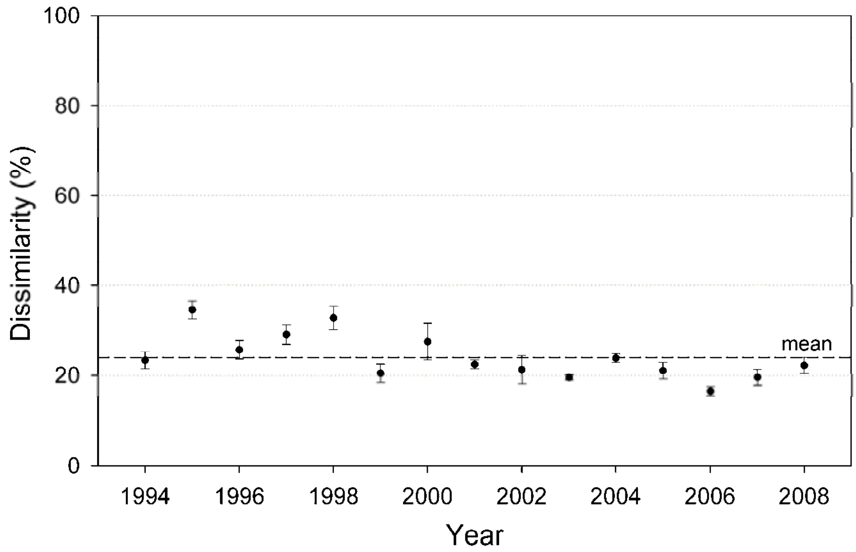

Figure 1 Paired control-exposed dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in Mudginberri ('exposed') and Sandy ('control') Billabongs in the vicinity of the Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

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

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

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

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

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

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

Shallow lowland billabongs

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

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

Introduction

C Humphrey, A Bollhöfer & D Jones

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

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

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

2 development of continuous monitoring;

3 development of in situ toxicity monitoring; and

4 longitudinal study of bioaccumulation in mussels in Magela Creek.

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

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

J Brazier, C Humphrey & D Buckle

Proposed relocation of sampling sites in Magela Creek

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

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

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

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

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

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

A. Upstream monitoring sites on Magela Creek near Ranger B. Downstream monitoring sites on Magela Creek

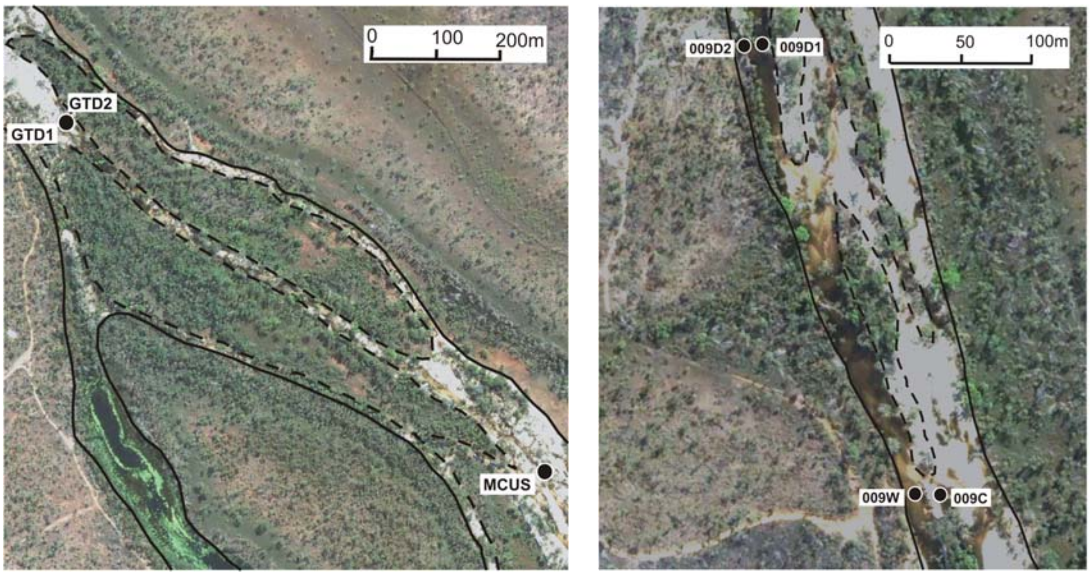

Figure 1 Upstream and downstream monitoring sites used in the SSD's water chemistry (grab sampling and continuous) and toxicity monitoring programs. Channel boundaries are indicated by the continuous or broken (water-level-dependent) lines.

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

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

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

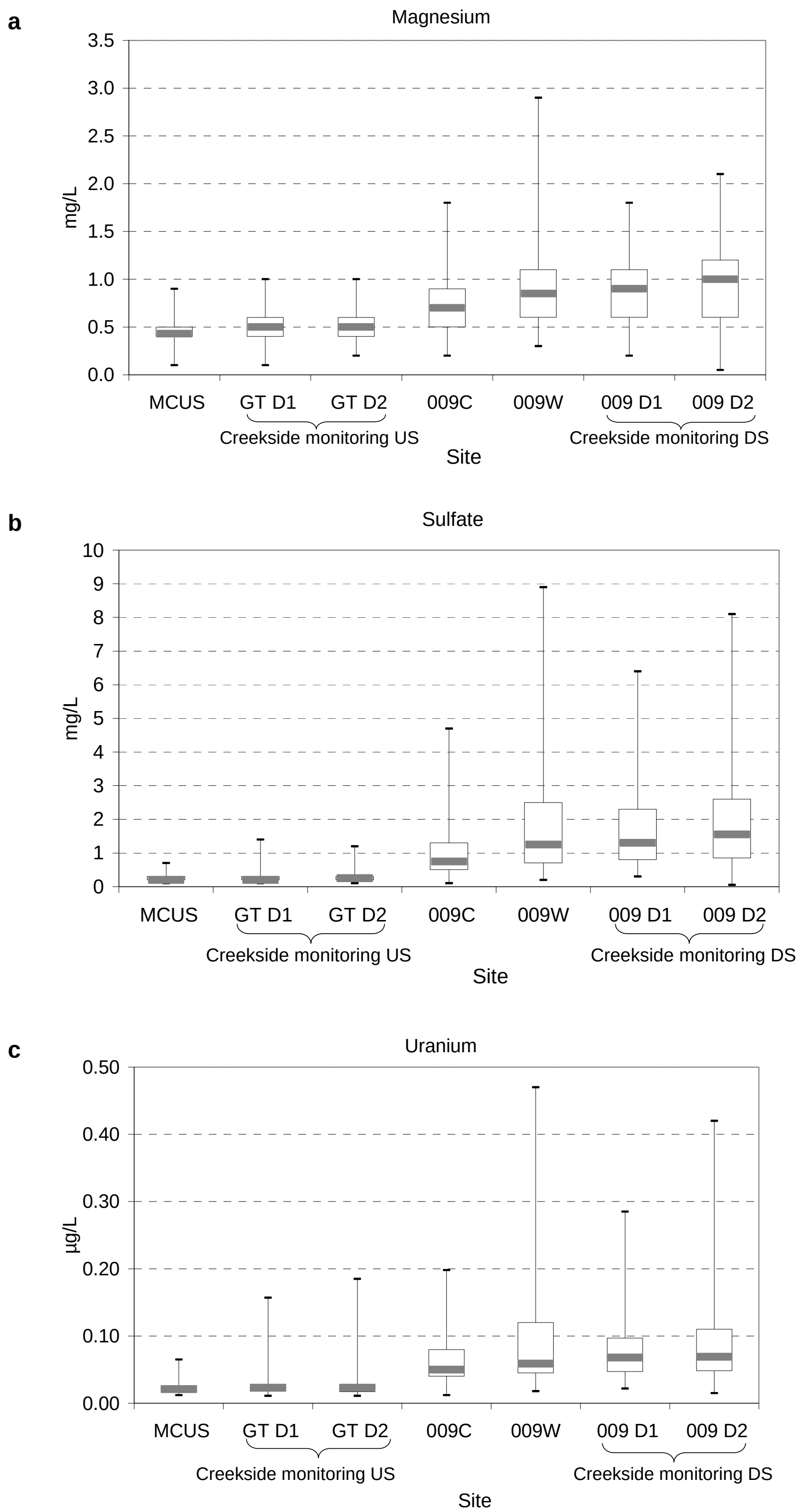

Figure 2 Box plots of concentrations measured between 2001 and 2008 for the upstream routine statutory monitoring site (MCUS), upstream creekside monitoring site (GT D1 and GT D2), downstream statutory compliance monitoring site (009C), downstream site adjacent to 009C but closer to the west bank (009W),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

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

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

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

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

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

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

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

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

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

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

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

Future of the weekly grab sampling program – Gulungul Creek

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

Continuous monitoring of water quality in Magela Creek

K Turner

Background

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

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

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

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

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

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

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

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

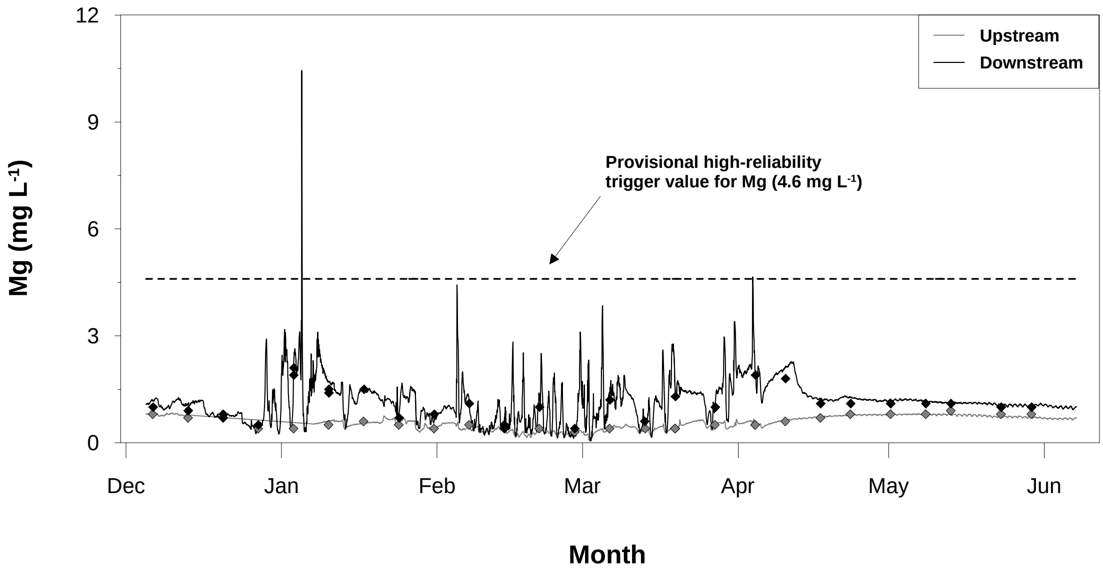

Figure 1 Predicted continuous Mg concentration (lines) with overlain measured grab sample values (diamond points) for upstream and downstream of the Ranger mine for the 2007–08 wet season

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

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

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

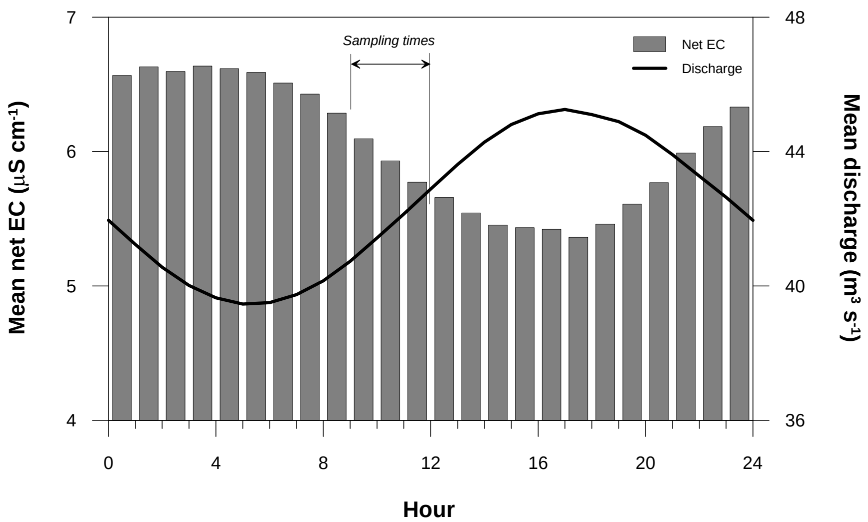

Figure 2 Mean hourly net EC (mean downstream EC minus mean upstream EC) in Magela Creek over the 2005–06, 2006–07 and 2007–08 wet seasons (grey bars). Also shown is the mean hourly Magela Creek discharge over the entire period of record (black line). The typical day time window (sampling times) during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

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

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

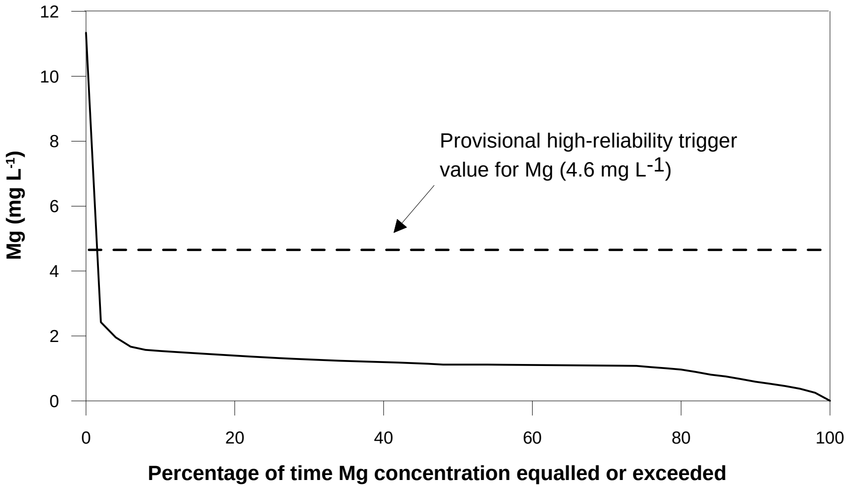

Figure 3 Frequency distribution curve for Mg concentrations predicted from the continuous (5-15 min frequency) EC record in Magela Creek downstream of the mine, showing the percentage of the time that the Mg concentrations equalled or exceeded values within the range of Mg concentrations between 2005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

Development of Magela Catchment area solute budget using continuous monitoring systems

K Turner & D Jones

Background

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

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

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

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

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

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

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

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

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

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

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

Methods

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

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

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

Electrical conductivity – Mg relationships

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

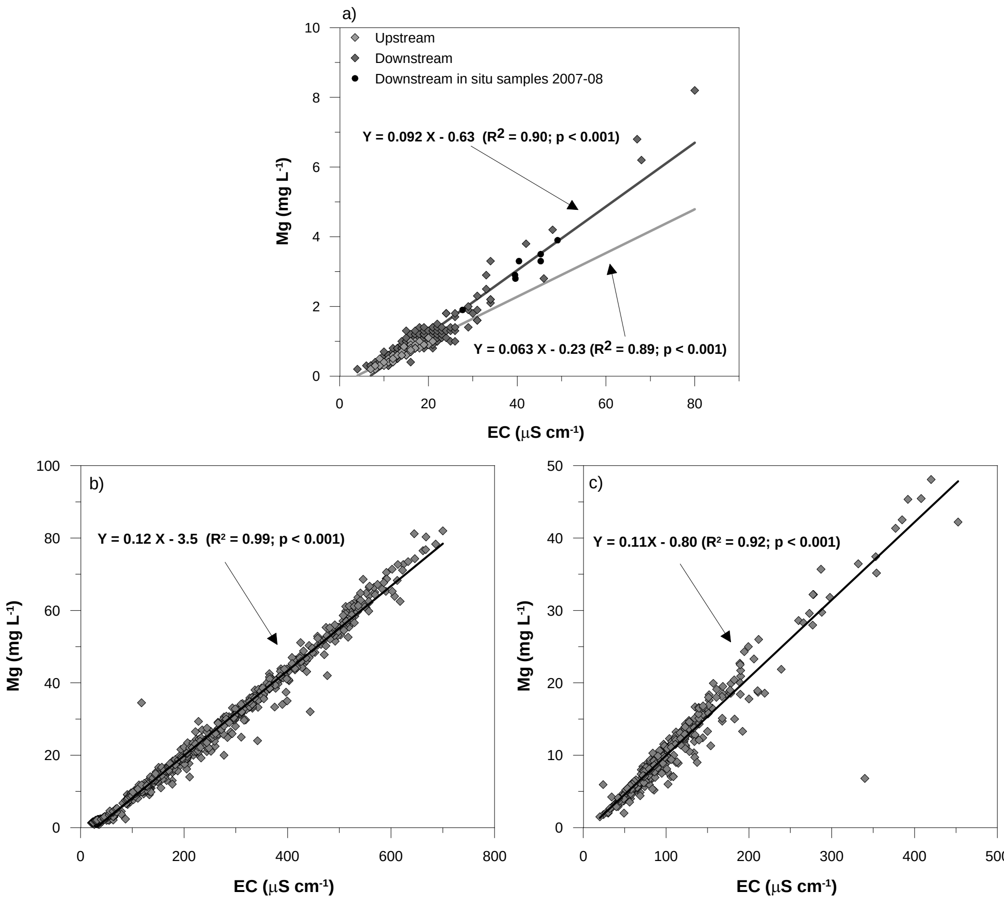

Figure 1 Relationships between EC and Mg concentration for the a) upstream and downstream sites on Magela Creek, b) RP1 in the Coonjimba Creek catchment and c) GC2 in the Corridor Creek catchment

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

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

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

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

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

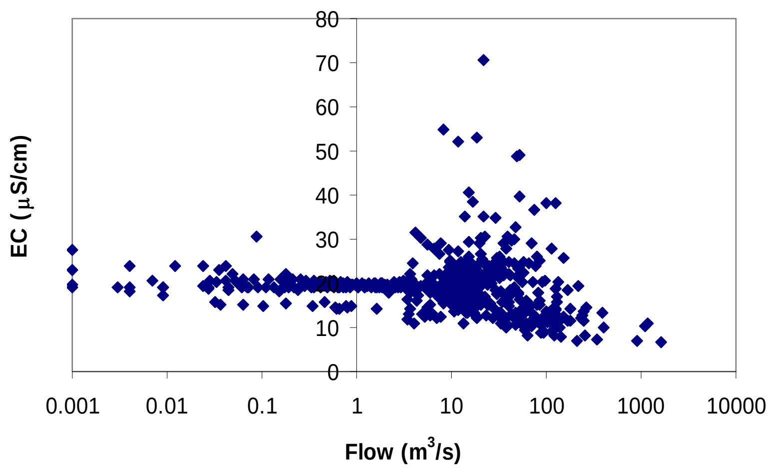

Figure 2 Distribution of daily mean EC values at the Magela Creek downstream site against flow at G8210009

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

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

Magnesium loads

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

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

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

Minesite

Point sources

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

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

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

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

Diffuse sources

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

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

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

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

Magela Creek

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

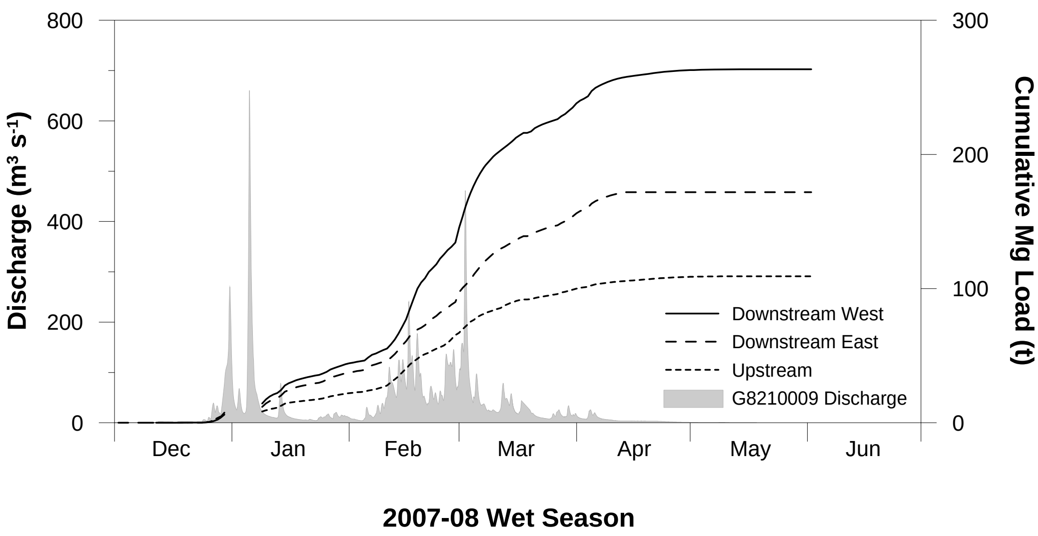

Figure 3 Cumulative Mg loads measured at the upstream and at both downstream sites during the 2007–08 wet season, along with discharge measured at G8210009

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

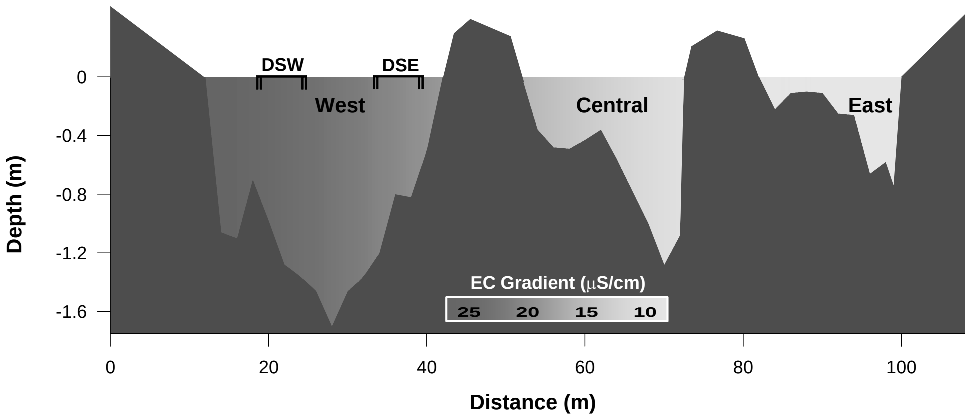

Figure 4 Cross section of Magela Creek at the downstream continuous monitoring site showing three distinct channels separated by sandbanks. The extent of shading illustrates the EC gradient (ranging from 25 µs/cm at the west bank to 10 µS/cm in the central channel) measured on 1 April 2008 when flow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

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

Load balance at Magela Creek downstream

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

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

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

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

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

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

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

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

Diffuse inputs

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

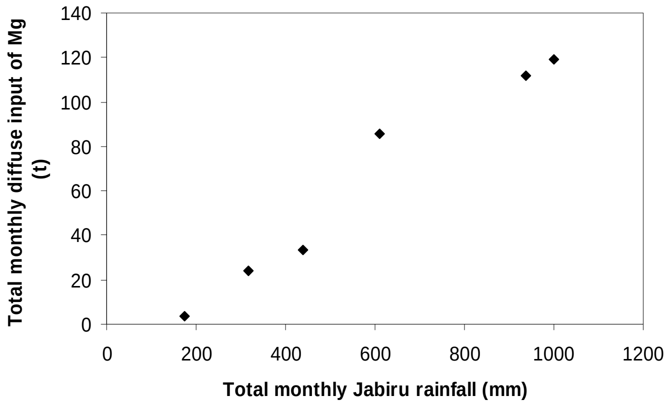

Figure 5 Total monthly rainfall measured at Jabiru airport versus total monthly input of Mg to Magela Creek via diffuse sources for the 2005–06 and 2006–07 wet seasons. Data used for this analysis is from Feb to May each year.

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

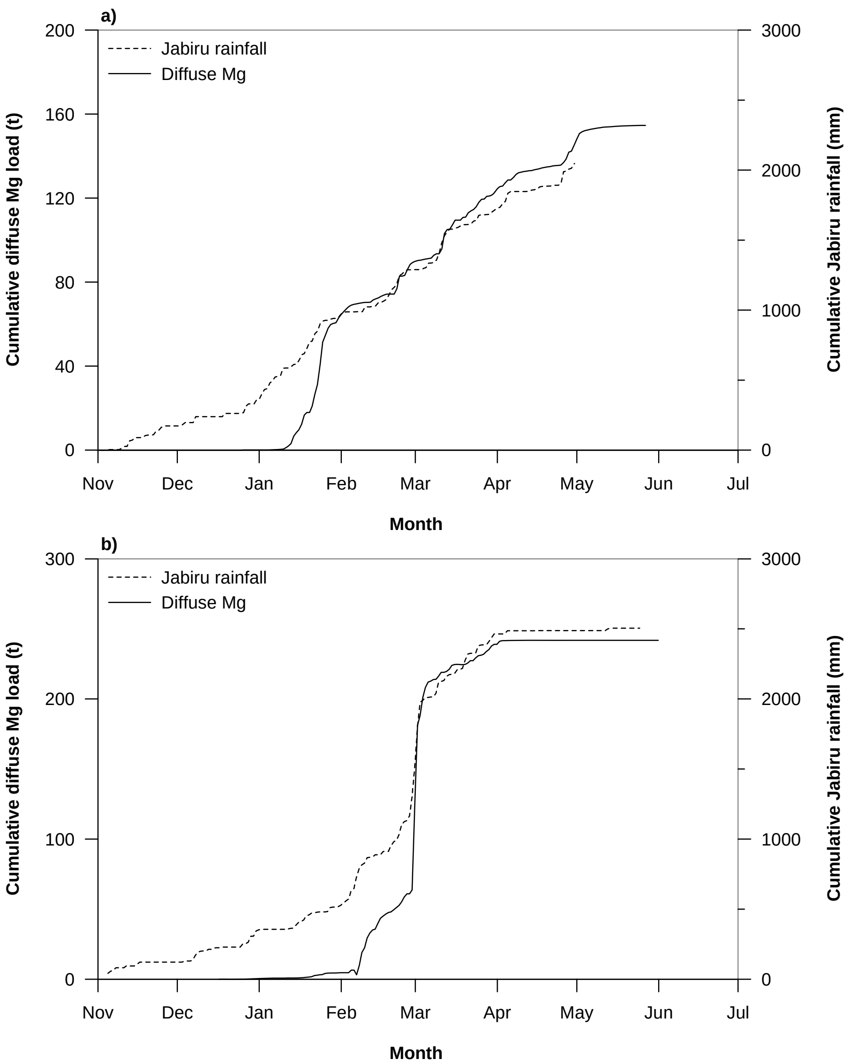

Figure 6 Cumulative diffuse Mg load (downstream load minus sum of point source inputs) entering Magela Creek over the a) 2005–06 and b) 2006–07 wet seasons, along with cumulative rainfall measured at Jabiru airport

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

Summary

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

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

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Development of in situ toxicity monitoring methods for Magela Creek

C Humphrey D Buckle & C Davies

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

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

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

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

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

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

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

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

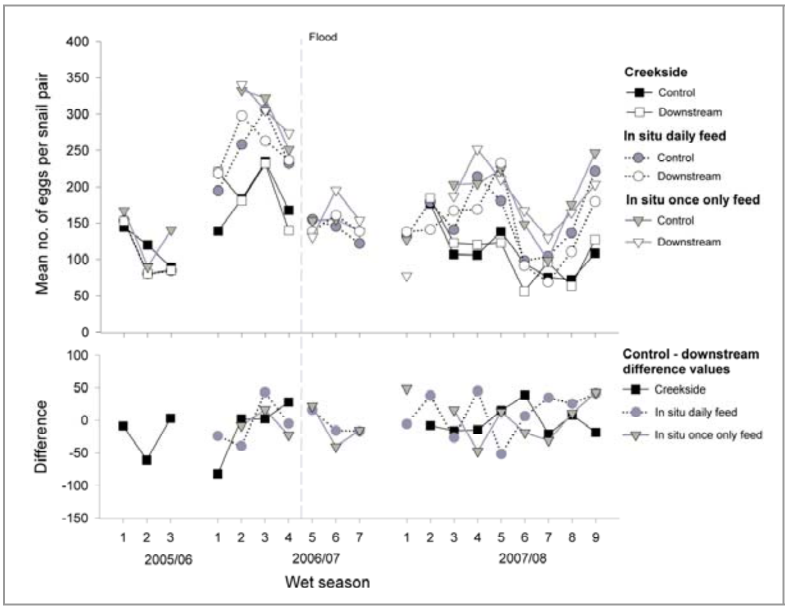

Figure 1 Comparison of snail egg production for creekside monitoring and in situ monitoring, 2005–06, 2006–07 and 2007–08 wet seasons. Note that in 2005–06, in situ testing was confined to the control site while the once-only feeding treatment was not included in the first test of 2006–07.

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

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

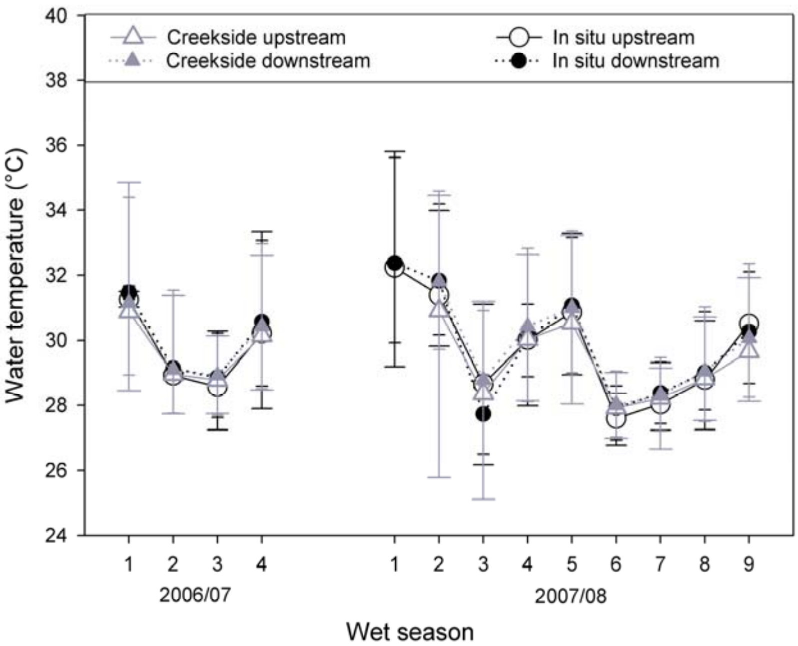

Figure 2 Comparison of test water temperatures recorded between creekside and in situ tests over the 2006–07 and 2007–08 wet seasons. Symbols and vertical bars depict the mean and range (maximum and minimum) in temperatures, respectively, at least hourly over the 96 hr test period.

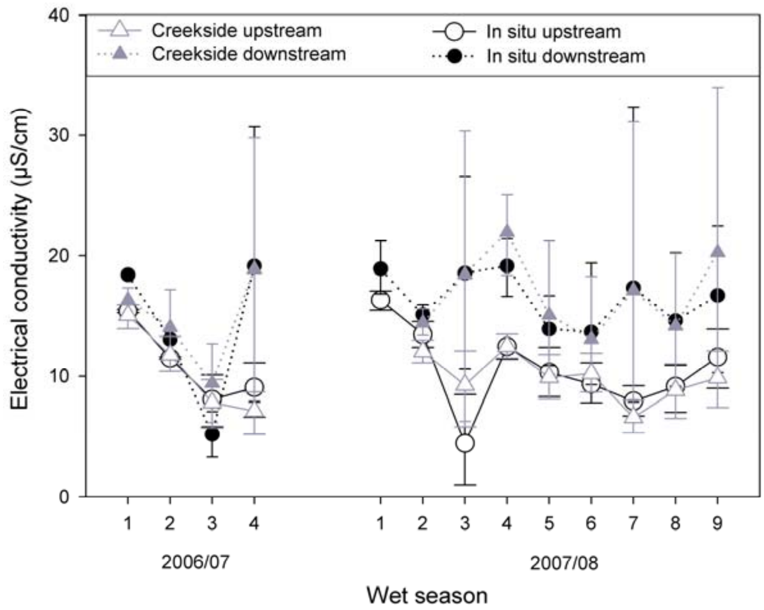

Figure 3 Comparison of test water electrical conductivity (EC) recorded between creekside and in situ tests over the 2006–07 and 2007–08 wet seasons. Symbols and vertical bars depict the mean and range (maximum and minimum) in EC, respectively, at least hourly over the 96 hr test period.

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

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

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

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

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

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

J Brazier, A Bollhöfer, C Humphrey & B Ryan

Background

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

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

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

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

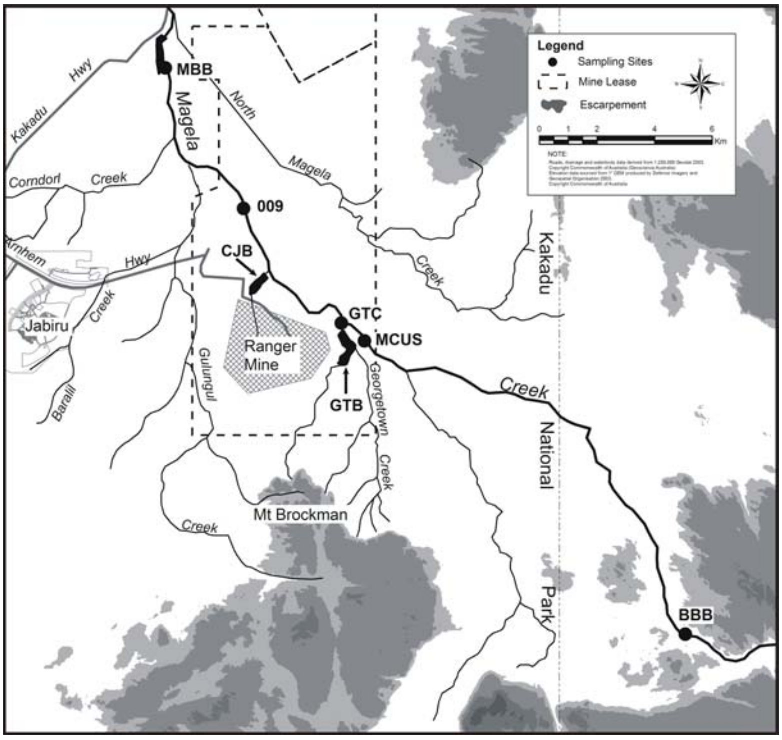

Figure 1 Location of sampling sites along Magela Creek. BBB, Bowerbird Billabong; MCUS, Magela Creek upstream; GTC, Georgetown confluence; G8210009, Magela Creek downstream; MBB, Mudginberri Billabong; GTB, Georgetown Billabong; CJB, Coonjimba Billabong

Potential effects of variation in mussel weights upon measured contaminant concentrations

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

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

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

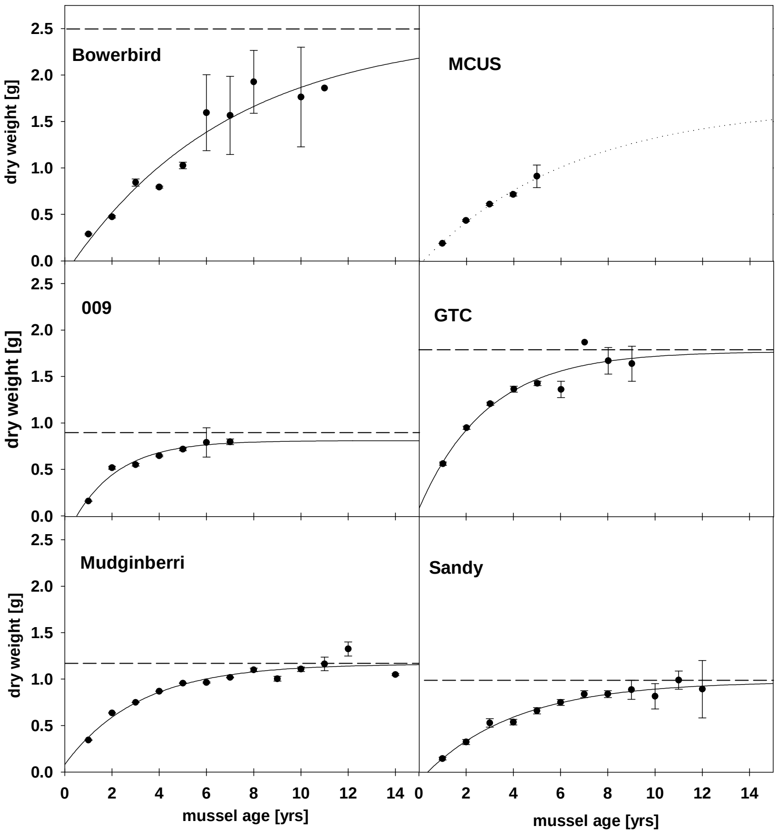

Figure 2 Mussel dry weight plotted versus age at the sampling locations along the Magela Creek channel and from the control site, Sandy Billabong. The solid line represents the results of a fit assuming that the mussel weight-age relationship follows a form of the von Bertalanffy Growth Equation. Dashed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

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

Radium in mussels and sediments

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

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

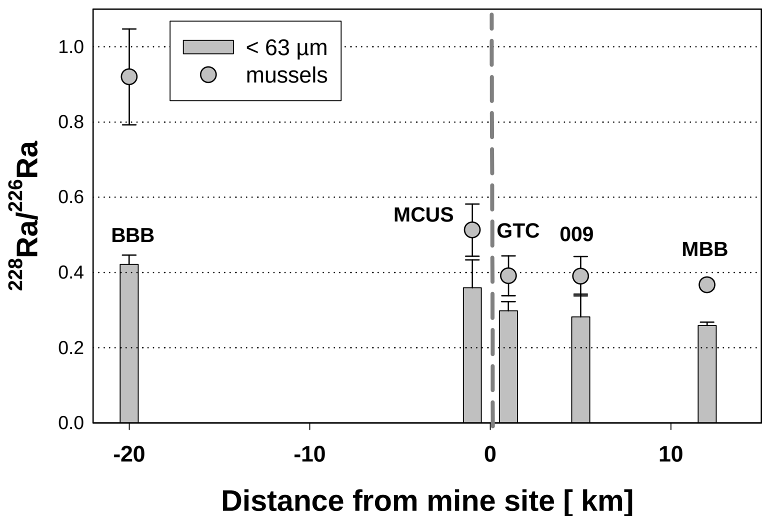

Figure 3 $^{228}Ra/^{226}Ra$ activity ratio measured in sieved (<63 µm) sediment and in mussels collected along Magela Creek. BBB, Bowerbird Billabong; MCUS, Magela Creek upstream; GTC, Georgetown confluence; G8210009, Magela Creek downstream; MBB, Mudginberri Billabong

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

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

Uranium in mussels and sediments

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

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

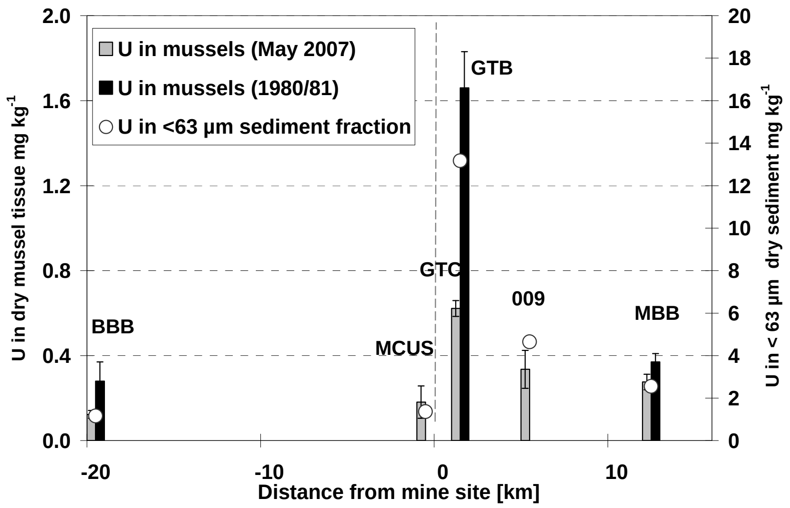

Figure 4 Comparison of May 2007 uranium concentrations in mussels along Magela Creek, upstream and downstream of the mine with historical (1980) data. Uranium concentration in the < 63 μm fraction that is potentially bioavailable (1 M HCl acid digest) is also shown.

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

Stable lead (Pb) isotope ratios in mussels and sediments

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

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

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

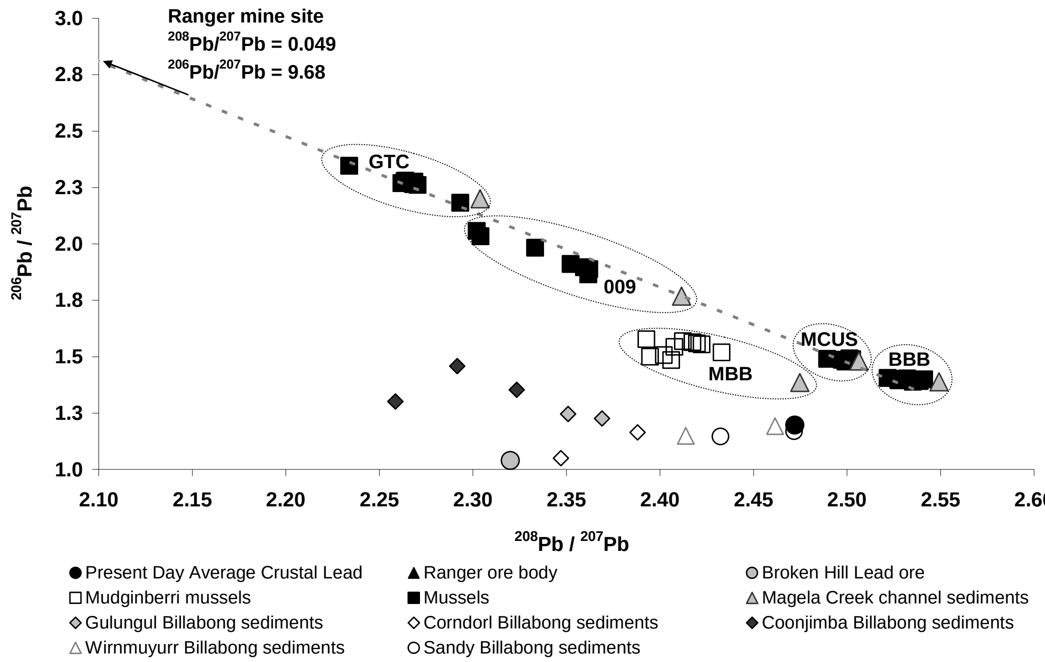

Figure 5 $^{206}Pb/^{207}Pb$ plotted versus $^{208}Pb/^{207}Pb$ isotope ratios measured in mussel tissue and < 63 μm sediment fraction (1 N HCl digest) from Magela Creek. The dashed trendline assumes mixing of radiogenic lead with the Ranger ore signature (not shown as outside of axes range) and lead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

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

Conclusion

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

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

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