supervising
scientist
report
209

eriss research summary
2013–2014

Supervising Scientist

Australian Government
Department of the Environment
Supervising Scientist

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Contents

Part 2: Ranger – Rehabilitation

KKN 2.2: Landform design

KKN 2.5: Ecosystem establishment

KKN 2.6: Monitoring

Preface

The Environmental Research Institute of the Supervising Scientist (***eriss***) is part of the Supervising Scientist Division (SSD) of the Australian Government's Department of the Environment (DoE). ***eriss*** provides specialist technical advice to the Supervising Scientist on the protection of the environment and people of the Alligator Rivers Region (ARR) from the impact of uranium mining. Its major function is to conduct research into developing leading practice methodologies for monitoring and assessing the impact of uranium (U) mining on water and air (transport pathways) and soil, and on the bushfoods that are consumed by the local Indigenous people. This research spans the operational, decommissioning and post-rehabilitation phases of mining.

eriss also applies its expertise to conducting research into the sustainable use and environmental protection of tropical rivers and their associated wetlands, and to undertaking a limited programme of contract research on the impacts of mining elsewhere in the north Australian tropics.

The balance and strategic prioritisation of work within the uranium component of ***eriss***' project portfolio are defined by Key Knowledge Needs (KKNs) developed through consultation between the Alligator Rivers Region Technical Committee (see ARRTC membership and function in Appendix 2), the Supervising Scientist, Energy Resources of Australia Ltd (ERA) and other stakeholders. The KKNs are subject to ongoing review by ARRTC to ensure their currency in the context of any significant changes that may have occurred in U mining related activities and issues in the ARR.

Not all of the KKN research areas (Appendix 3) are able to be covered by ***eriss***, since not all of the required disciplines are available within the Institute. To address these particular gaps, collaborative projects are conducted between ***eriss*** and researchers from other organisations, and consultants are commissioned by ***eriss*** and others to undertake specific pieces of work. For example, KKN projects related to detailed hydrogeology or tailings management on the Ranger lease are conducted and reported separately by consultants engaged by ERA. A more complete picture of the scope of research work that is conducted by all parties can be obtained by referring to the minutes that are produced for the meetings of ARRTC: www.environment.gov.au/ssd/communication /committees/arrtc/meeting.html.

This report documents the monitoring and research projects undertaken by ***eriss*** over the 2013–14 financial year (1.7.13 to 30.6.14). Most of the work presented in the report was also presented to ARRTC at its 33rd Meeting in November 2014. Where possible, papers have been updated following significant feedback from ARRTC, although this may not be the case for all papers. The report is structured according to the five major topic areas in the KKN framework, noting that this year there are no papers for the last two topics.

1. Ranger – current operations
2. Ranger – rehabilitation
3. Jabiluka
4. Nabarlek
5. General Alligators Rivers Region

Not all of the projects in ***eriss'*** KKN project suite have been reported on since; for reasons of staff resourcing or other demands in the context of absolute project priority, no substantive work may have been done on a particular project during the past year. A list of those projects that have not been reported on, and a summary of the reasons for the reduced activity, is provided in Appendix 2.

Much of the monitoring and research work conducted by ***eriss*** is focused on the wet season and its immediate aftermath, since it is during this period that the environment is potentially at most risk from past and current uranium mining activities. The 2013–14 wet season was well above average with a total of rainfall at Jabiru Airport of 1963 mm (annual average 1536 mm).

During 2013–14, progress of the research programme was again limited by budget constraints making additional/new recruitment difficult. However, the opportunity during the third quarter to recruit several short-term (3 month) non-ongoing contracts in the Revegetation and Landscape Ecology (RLE) programme and Hydrologic, Geomorphic and Chemical Processes (HGCP) programme, and retain existing contract staff in the Ecotoxicology programme, made a significant and noticeable difference to research progress (and staff morale) towards the end of the year. This highlighted the issue of 'critical mass' within the research programme, and provided us with a benchmark of the point of critical mass across much of the programme. By the year's end, project milestone completion totaled 55/77 (71%) compared to 64/101 (63%) in 2012–13, reflecting an improvement in completion rate as well as a more manageable workload.

Selected key areas of research progress during the year are very briefly described below, with further specific details of these and other projects provided throughout this report. Of the 30 or so active research projects during 2013–14, the majority (>95%) were addressing issues associated with the current operational phase and/or proposed rehabilitation and post-rehabilitation phases of Ranger mine.

Key research projects for 2013–14:

- *Toxicity of ammonia to freshwater species* – R&D to design a test system in which test water pH could be maintained within strict limits (due to strong influence of pH on ammonia speciation and toxicity) was successful, and solid progress on the toxicity testing of ammonia under typical Magela Creek water quality conditions was finalised for three of the six species (see p. 26);

- *Erosion and chemistry studies on the trial landform* – 2013–14 saw the collection of a fifth year of data from the trial landform erosion plots, with the bedload data in particular confirming the rapid decline in materials leaving the site in the years post-construction. Significant in-roads were made on processing of back-logged samples, and a workshop was held in August 2014 to review all available results (see p. 131);

- *Developing monitoring methods using an Unmanned Aerial System (UAS)* – The rehabilitated Jabiluka site is being used as a test site to assess and develop methods for monitoring revegetation and erosion using an UAS. Such methods may represent cost-effective ways of monitoring rehabilitation success at Ranger (see p. 222);

- *Developing water quality closure criteria for solutes in billabongs* – The assessment of Ranger mine-affected billabongs in comparison to reference billabongs to identify water quality thresholds at or below which aquatic communities can be maintained was updated with macroinvertebrate data from the 2012–13 billabong survey. Moreover, two potential confounding factors for the analyses were assessed and discounted as significant causative factors in macroinvertebrate community responses. This work is currently informing the development of water quality closure criteria (see p. 166).

- *Further population and application of the BRUCE tool to inform predictions of radiological dose to humans and non-human biota* – The BRUCE tool database now stores radionuclide and metal concentration data for almost 10,000 biota (plants and animals) and environmental media (water, soil and sediment) samples. In 2013–14, it was used for numerous projects, including the prediction of (i) post-rehabilitation radiation exposure due to the consumption of terrestrial Aboriginal bush foods (see p. 199) and (ii) environmental media concentration limits for terrestrial wildlife (see p. 213).

Jabiluka is in long-term care and maintenance and the current work of the Supervising Scientist is focused on maintaining a routine continuous monitoring programme for flow, electrical conductivity and turbidity downstream of the formerly disturbed area. These results are reported in Part 3. The Jabiluka rehabilitated area is being used a study site for the project developing UAS-based monitoring methods for the Ranger rehabilitation, while staff of the HGCP programme undertook an erosion assessment at the site and downstream drainage tributaries following the first wet season since rehabilitation of the formerly disturbed area.

The key non-uranium mining related external activity for 2012–13 was the involvement of several ***eriss*** staff in the current revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Details of the involvement in the Water Quality Guidelines revision, and other science and technical activities not related to the core research and monitoring programme, are provided in the Supervising Scientist 2013–14 Annual Report.

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

For additional information, readers are referred to the annual publications list (Appendix 1) that details all of the material published, and conference and workshop papers presented, by ***eriss*** staff in 2013–14.

Rick van Dam, *Director,*
Environmental Research Institute of the Supervising Scientist

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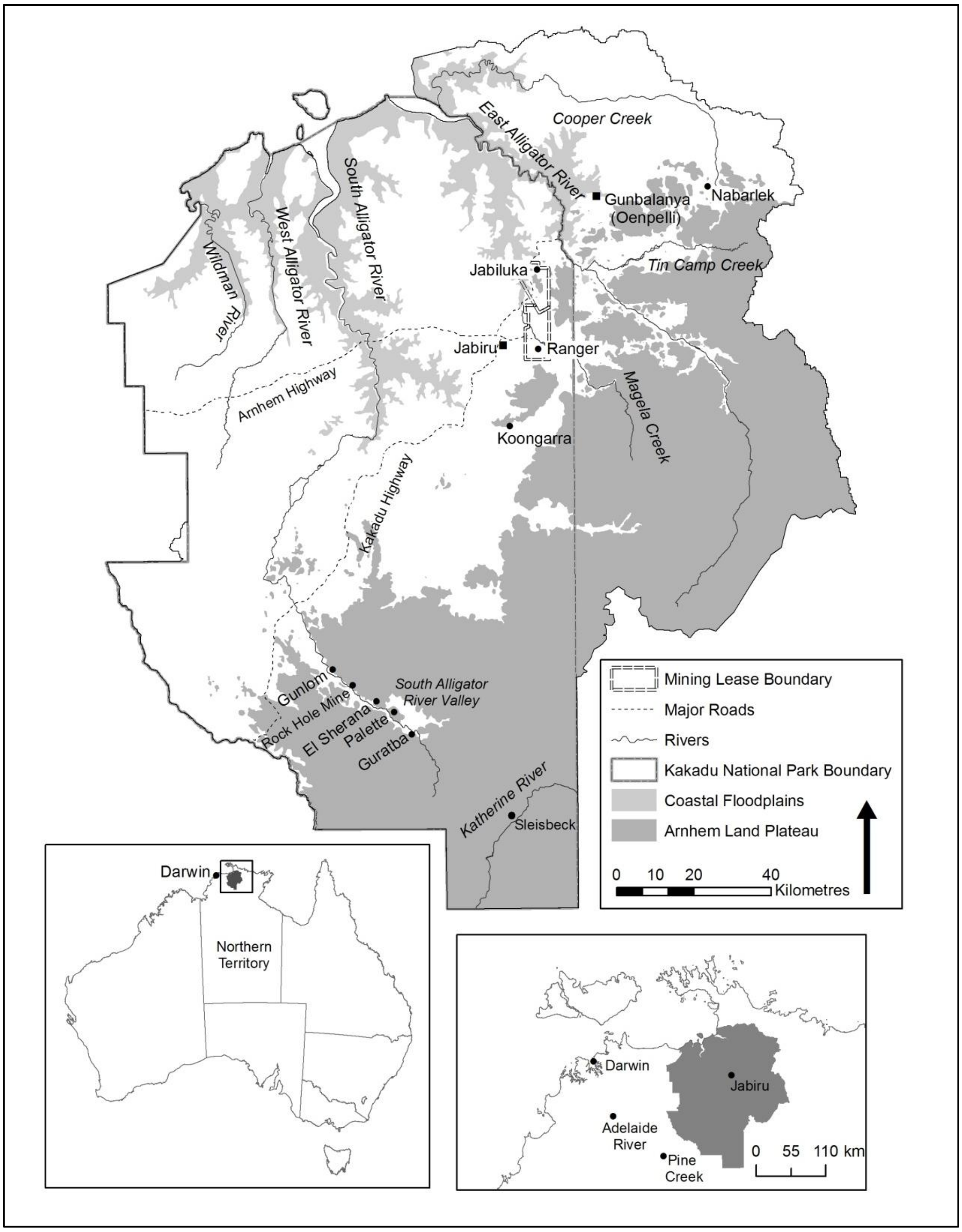
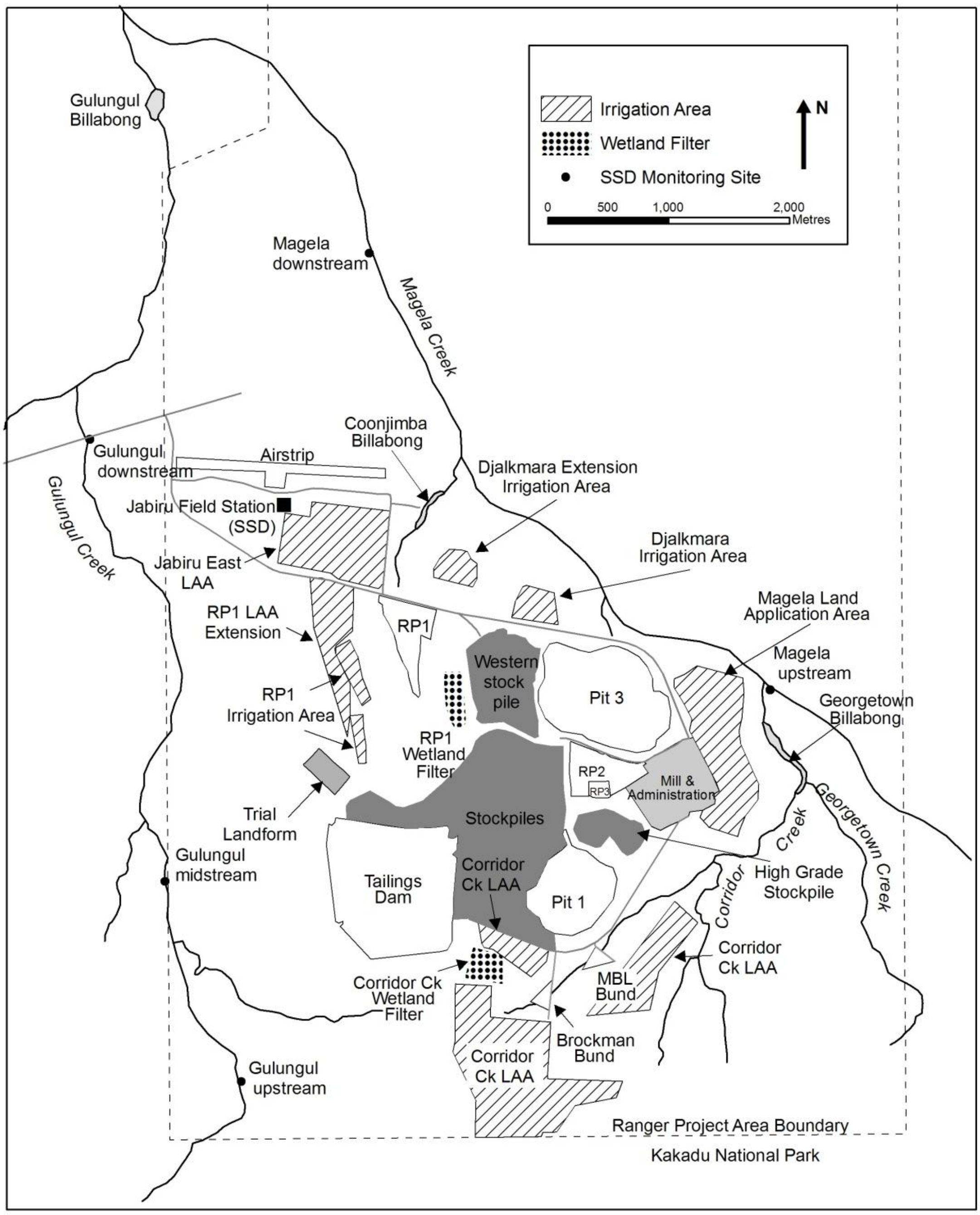
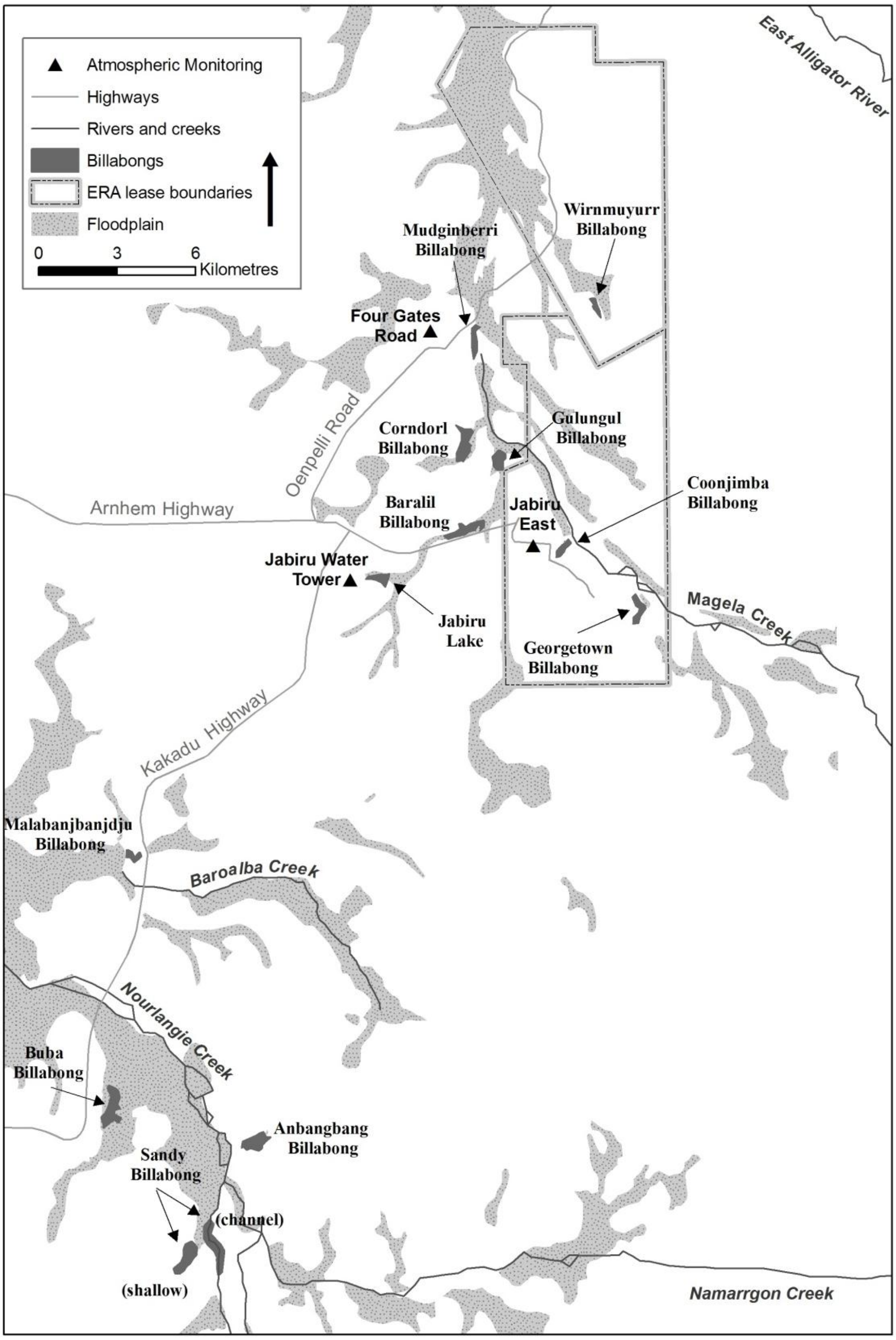
Maps



Map 1 Alligator Rivers Region



Map 2 Ranger minesite



Map 3 Sampling locations used in SSD's research and monitoring programmes

Part 1

Ranger – Current operations

Key Knowledge Need 1.2: Ongoing operational issues

Investigation of water quality in the Gulungul Creek catchment to the west of Ranger's tailings storage facility

K Turner & S Marshall

Executive summary

During the 2013–14 wet season, there were a number of electrical conductivity (EC) spikes in Gulungul Creek that were higher than historically-recorded levels, the most significant of these being a 3-hour EC event that peaked at 184 µS/cm, measured at ERA's continuous monitoring site Gulungul Creek Lease Boundary (GCLB) on 19 January 2014 (Figure 1). Whilst of a greater magnitude than previously observed, this event fell below the Magela Creek toxicity derived threshold for possible environmental effect of 1,138 µS/cm for 3 hour exposure. These EC events indicated a notable change in Gulungul Creek water quality in the vicinity of the minesite, which had a previous maximum recorded EC of 56 µS/cm at GCLB. No such change was detected at ERA's downstream monitoring compliance point (GCH) where the EC measured in weekly grab samples during the 2013–14 remained within previously reported ranges (towards the upper end), which may indicate that the far-field effects on water quality are limited.

In order to determine the source of the elevated solutes in Gulungul Creek, investigations were carried out by both the Supervising Scientist and ERA throughout the 2013–14 wet season and into the 2014 dry season. It was determined that the solutes were entering Gulungul Creek along Gulungul Creek Tributary 2 (GCT2), a flow path that originates at the western wall of the Tailings Storage Facility (TSF) (Figure 1). As well as increasing the conductivity of Gulungul Creek surface waters, the high solute loading in the GCT2 tributary also resulted in the formation of salt crystals, or evaporites, along the full extent of the flow path during the dry season. The salts encrusted surface soils and accumulated around the base of trees and on other vegetation and leaf litter.

Historical literature detailing investigations into contaminated surface water and groundwater to the west of the TSF is currently being reviewed in detail by the Supervising Scientist to ensure that the most recent data are interpreted in the context of work previously carried out. The review is ongoing and will be used to underpin the formulation of future research projects undertaken by either the Supervising Scientist or ERA. The present report examines data collected as part of ERA's and the Supervising Scientist's 2013–14 investigation into the high EC levels recorded in Gulungul Creek and contains some preliminary evidence that suggests TSF seepage should not be ruled out as a contributor to solutes in the Gulungul Creek catchment.

Introduction

The TSF at Ranger uranium mine has been in operation since 1979. The design of the TSF is a ring-dyke or turkey's nest structure, covering an area of approximately 1 km^2 (Townley 1996). It straddles the Coonjimba Creek, Gulungul Creek and Corridor Creek surface water catchment divides. When built, the wall foundations were prepared through the clearance of vegetation and the excavation of alluvial and lateritic sediments (classified as Aquifer 1a and 1b, respectively) to a depth of approximately 1.5–2 m below ground level (Volk et al. 1980, in URS 2010). The TSF foundations were only grouted along the northern wall in zones where it was thought that faults were present in the bedrock (Coffey Mining, 2010 in Puhalovich et al. 2012). The TSF walls contain a compacted clay core located at the upstream face and these cores were keyed into the weathered gneiss and schist bedrock (Burgess 2008, in URS 2010). The floor of the TSF was not excavated to bedrock, nor was it lined, but it was cleared of vegetation prior to the deposition of tailings (Burgess 2008, in URS 2010). The lateritic clays remaining at the floor of the TSF have been classified as having a low horizontal hydraulic conductivity of approximately 10^{-9}–10^{-10} m/s (Peter et al. 1994 in URS 2010). However, the vertical hydraulic conductivity may be approximately 100 times greater due to sand drain structures present (Brown and Lowson 1990 and Peter Burgess *pers. comm.* 2010, in URS 2010).

The various capacity expansions of the TSF are summarised in Table 1. For Stages II to IV, the lateritic sediments were excavated following the same procedure as for Stage I. During the Stage III and IV lifts, low grade ore was used in the TSF walls, and water from RP1 and RP4 was used for dust suppression and compaction (Townley 1996). Stages VA, VB and VI included the use of low grade ore and waste rock as the foundation for the lifts.

Table 1 Summary of the TSF lifts (from URS 2010).

Stage	Relative Level (m AHD)	Date Completed
I	36	1980
II	39	1983
III	41	1985
IV	44.5	1990
VA	47.5	2006
VB	51	2007
VI	54	2009

Numerous investigations related to the potential for, and the occurrence of, seepage from the TSF have been carried out over time. In a review of all TSF-related groundwater studies carried out prior to 1996, Townley (1996) indicated that the TSF had not been engineered to prevent seepage and that seepage was occurring within the design expectations. Groundwater contamination in proximity to the TSF has been reported since the 1990s. Martin & Akber (1994) identified bores to the north of the TSF along the Coonjimba Creek line that showed an increase in uranium concentrations. Salama & Foley (1997) showed a steady temporal increase in groundwater sulfate concentrations in the vicinity of the TSF, which was later supported by a study carried out by Klessa (2001). Buselli & Lu (1999, 2000) and Buselli et al. (2001) conducted a series of studies using geophysical techniques to detect groundwater contamination around the TSF and to configure distribution of aquifers across the site. Lu & Puhalovich (2002) also used geophysics in combination with groundwater data to characterise the nature and behavior of the groundwater mound beneath the TSF. Puhalovich & Cook (2009) used recent geophysical surveys to assess seepage pathways around the tailings dam. This in only a very brief summary of some of the work conducted.

According to URS (2010), who conducted the most recent extensive review of previous hydrogeological investigations in the area of the TSF, seepage from the tailings through the base of the TSF is considered to be the primary source of existing, and potential, groundwater contamination to the west and north of the TSF. At the time of the review, time-series sulfate concentrations measured in shallow and deep groundwater were used to depict the TSF seepage plume in a conceptual hydrogeological model with a cross section running through the centre of the TSF from south to north. This model indicated that while the seepage plume was located directly underneath the TSF footprint, seepage-affected groundwater (with increased sulfate concentrations) extended vertically to a depth of approximately 40 meters below ground level and extended laterally to the north under Retention Pond 1 (RP1) (Figure 1). The report did not show a similar conceptual model for the cross section from east to west.

During the 2013–14 wet season, there were a number of EC spikes in Gulungul Creek that were higher than historically-recorded levels, indicating a notable change in Gulungul Creek water quality compared to previous wet seasons. In order to determine the source of the elevated solutes in Gulungul Creek, investigations were carried out by both the Supervising Scientist and ERA throughout the 2013–14 wet season and into the 2014 dry season.

2013–14 Investigation

As part of the Supervising Scientist's investigation, surface water samples were collected from various sites in the Gulungul Creek catchment, both on and off the Ranger minesite (Figure 1). Soil samples were also collected however these data are still undergoing analyses and are not reported here.

During the 2013–14 wet season, a slight deterioration was observed in water quality at the Supervising Scientist's downstream monitoring site at Gulungul Creek (GCDS) (Figure 1). Continuous monitoring of EC at GCDS showed that the frequency and magnitude of EC spikes above 42 µS/cm (the 72 hour chronic exposure limit for Magela Creek) had increased compared to previous years, with a 2014–13 season maximum of 65 µS/cm, the highest EC recorded at this site since monitoring commenced. The maximum EC value measured further upstream at ERA's continuous monitoring site, GCLB, was 184 µS/cm. The maximum EC measured at ERA's downstream compliance site was 32 µS/cm, which unlike GCDS and GCLB was measured in weekly grab samples. The 2013–14 continuous EC data measured at GCDS, GCLB and the Supervising Scientist's upstream monitoring site, GCUS, are shown in Figure 2.

Figure 1 Surface water sites in the Gulungul Creek catchment area.

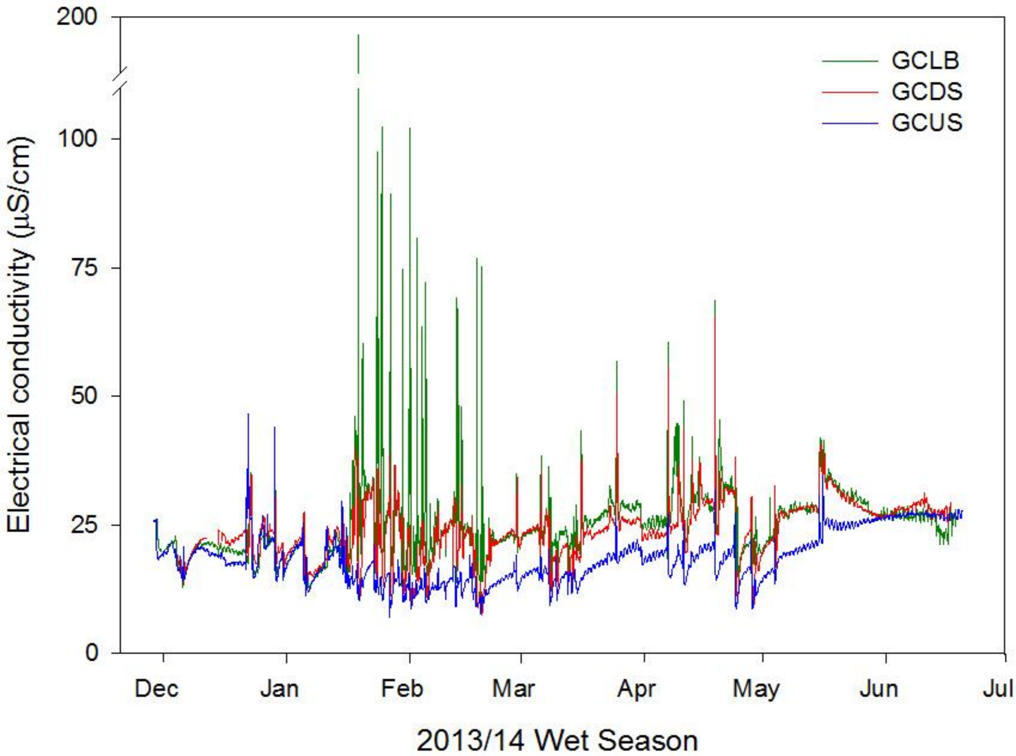


Figure 2 Electrical conductivity measured at 20 minute intervals at the Supervising Scientist's upstream (GCUS) and downstream (GCDS) monitoring sites and at ERA's lease boundary (GCLB) monitoring site. See Figure 1 for site locations.

ERA carried out a number of field investigations throughout the wet season to determine the source of the solutes causing the deterioration in Gulungul Creek water quality. On 13 February 2014, ERA staff identified high EC surface runoff flowing from the base of the western TSF wall into the GCT2 tributary, with values of up to 2,900 µS/cm. On 3 March 2014, Supervising Scientist staff conducted a field investigation to confirm the source of the high EC water. Low EC was recorded in pooled water throughout the Gulungul Creek catchment, with a range of 15–25 µS/cm. In contrast, the EC measured in the GCT2 tributary at the Radon Springs access track crossing (GCT2RST) (Figure 1) was in the range of 1,250–1,300 µS/cm. At the same location the pH was 5.4, and an orange flocculant was observed along the tributary flow path (Figure 3). The GCT2 tributary was tracked to its confluence with Gulungul Creek and the EC measured in the tributary approximately one meter upstream of its confluence with Gulungul Creek (GCT2GCC) was around 500 µS/cm (Figure 1).

Figure 3 Orange flocculant at GCT2RST on 3 March 2014.

On 3 March 2014, the Supervising Scientist deployed continuous EC data loggers at both GCT2RST and at GCT2GCC and the data are shown in Figure 4 along with cumulative rainfall data measured at a nearby hydrological monitoring station.

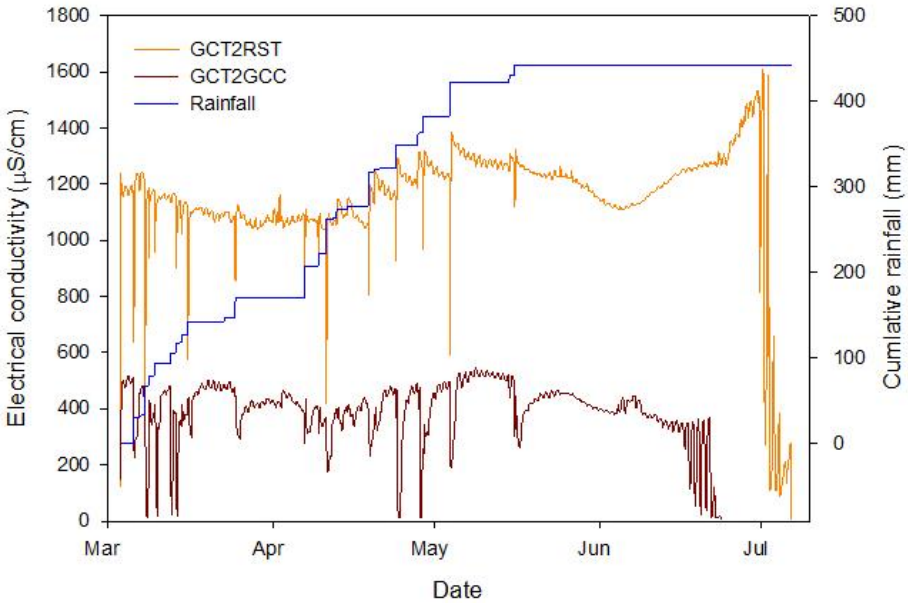


Figure 4 EC data measured in the GCT2 tributary at the Radon Springs access track crossing (GCT2RST) and at the confluence of the GCT2 tributary with Gulungul Creek (GCT2GCC) between March and August 2014. Also shown is the cumulative rainfall measured at a nearby hydrological station on Gulungul Creek.

The baseflow at each site along the GCT2 tributary is dominated by high EC water that is diluted briefly during rainfall events, as shown by the intermittent troughs coinciding with rainfall. Based on the EC measured at GCT2GCC, the mean EC of the input to the eastern side of the Gulungul Creek channel between March and July was about 400 µS/cm, which is 20 times higher than the 20 µS/cm baseflow in Gulungul Creek (Figure 5). The direct effect of the input from the GCT2 tributary on Gulungul Creek EC at GCDS is shown in Figure , where GCDS EC events coincide with the flushing of the tributary waters into the creek.

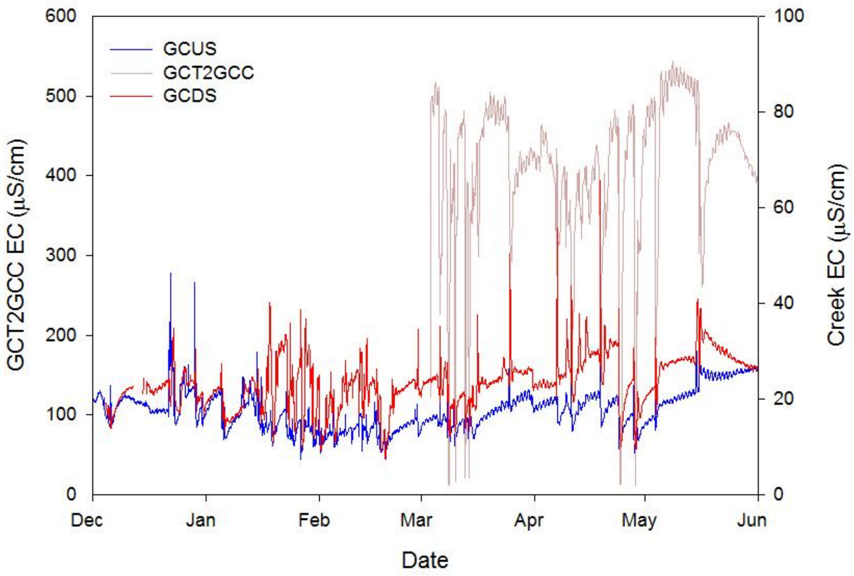


Figure 5 Peaks in EC at GCDS caused by input from the GCT2 tributary during the 2013/14 wet season.

To further delineate the source of the high EC water, ERA installed a number of continuous EC data loggers at various locations in Gulungul Creek and along the GCT3 and GCT2 tributaries on 4 March 2014 (Figure 6).

Figure 6 Locations of ERA investigative EC data loggers deployed in March 2014.

The average EC measured in the GCT3 tributary at Point 6 was around 25 µS/cm and it represents a control against which to compare the EC measured elsewhere in the catchment. In contrast to the GCT3 control, the average EC measured in the GCT2 tributary at Point 7 was 600 µS/cm, which lies between the GCT2RST mean of around 1,200 µS/cm (closest to the minesite) and the GCT2GCC mean of around 400 µS/cm (closest to Gulungul Creek). This spatial gradient of EC may be caused by dilution of the source water by ambient fresh water in the catchment or attenuation of non-conservative solutes by soils and organic material along the GCT2 tributary.

The Point 2 and Point 3 EC data are shown in Figure 7 below, along with corresponding EC measured at GCUS, GCLB and GCT2GCC. The plot also shows the water level measured at a nearby hydrological monitoring station. The period shown reflects the period of record for Point 2 and the gaps in the Point 2 trace indicate periods where the water level dropped below the logger.

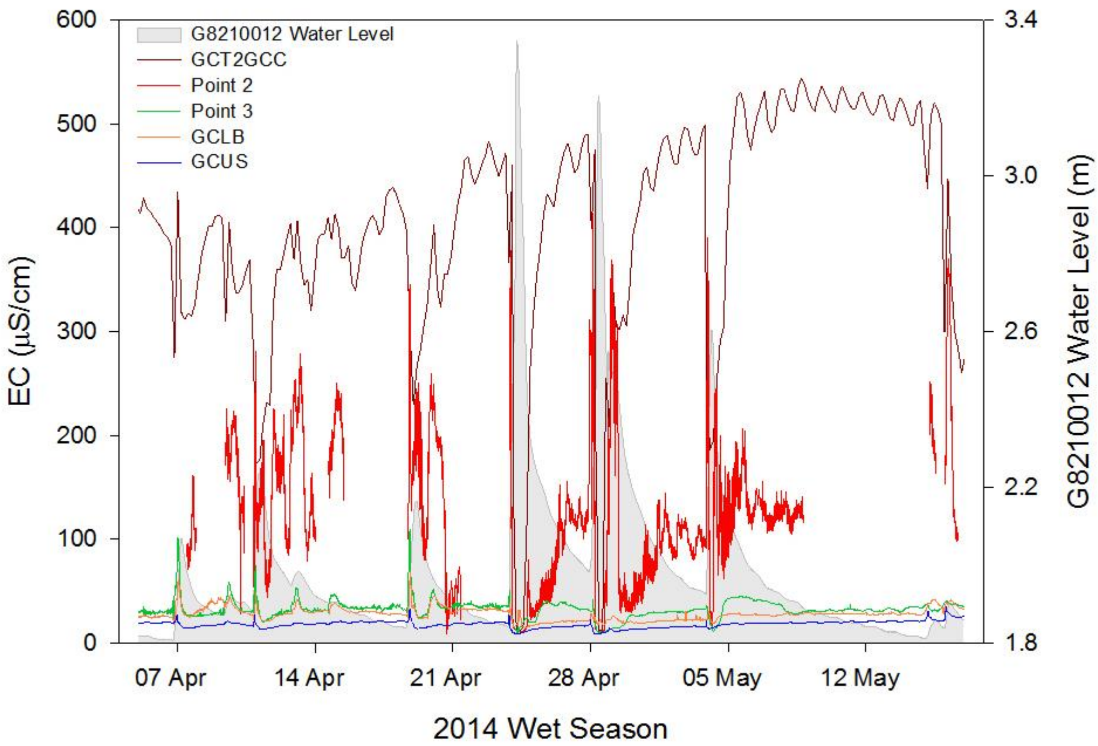


Figure 7 Electrical conductivity data measured at various points along Gulungul Creek, including GCUS and ranging downstream to GCLB. The water level measured at a nearby hydrological monitoring station is also shown.

Point 2 and Point 3, the two sites in Gulungul Creek closest to the confluence with the GCT2 tributary, both experienced EC above 100 µS/cm. The rising portion of the hydrograph for the larger flow events in Gulungul Creek coincide with an increase in EC in the creek, suggesting that high EC water is flushed from GCT2 into the creek during rising water levels. The maximum EC measured immediately downstream of the confluence of Gulungul Creek and GCT2, at Point 2, was 413 µS/cm. As the water level peaks in Gulungul Creek the EC measured at Point 2 returns to background levels (similar to GCUS), indicating that the entry point of the high EC water from GCT2 shifts or is hydraulically dammed, possibly due to over-bank creek flow, and the GCT2 signal is no longer observed at Point 2. It is likely that during high flow events the creek water inundates the GCT2 tributary itself as the EC measured at GCT2GCC under these conditions also reflected background levels. As the creek water level recedes it appears that the high EC water from GCT2 recommences flow into the creek, as its signal is once more detectable at Point 2.

The variability in the EC at Point 2 and at GCT2GCC reflects the dynamic mixing of creek water with the high EC water flowing from the GCT2 tributary. The data indicate a preferential flow path for the high EC water along the eastern side of the creek, resulting in the formation of a cross channel gradient in solute concentrations. This is supported by observations made by ERA staff on 19 February 2014 in response to a high EC event in the creek, which showed that after the EC in the creek had returned to background levels (around 10 µS/cm) some residual high EC water remained in a naturally low lying inundated area immediately upstream of GCLB. The overbank flow during the EC event would have pushed high EC water into this area (Figure 8).

Figure 8 EC measured by ERA staff in the Gulungul Creek channel and in standing water on the low lying right bank in the immediate vicinity of GCLB on 19 February 2014.

The GCT2RST site was inspected on 2 July 2014 and 19 August 2014 (Figure 9 and Figure 10, respectively). The last recorded rainfall in the catchment occurred in mid-May and in the drier conditions pH of the water at GCT2RST was 4.5. The orange flocculant was more concentrated and white salt crystals had formed along the GCT2 tributary flow path. The crystals formed in patches of varying density, covering the area depicted in yellow in Figure 11.

Figure 9 Orange flocculant at GCT2RST (upper frames) and salt crystals along the GCT2 tributary (lower frames) on 2 July 2014.

Figure 10 Salt crystals and orange flocculant along the GCT2 tributary on 19 August 2014.

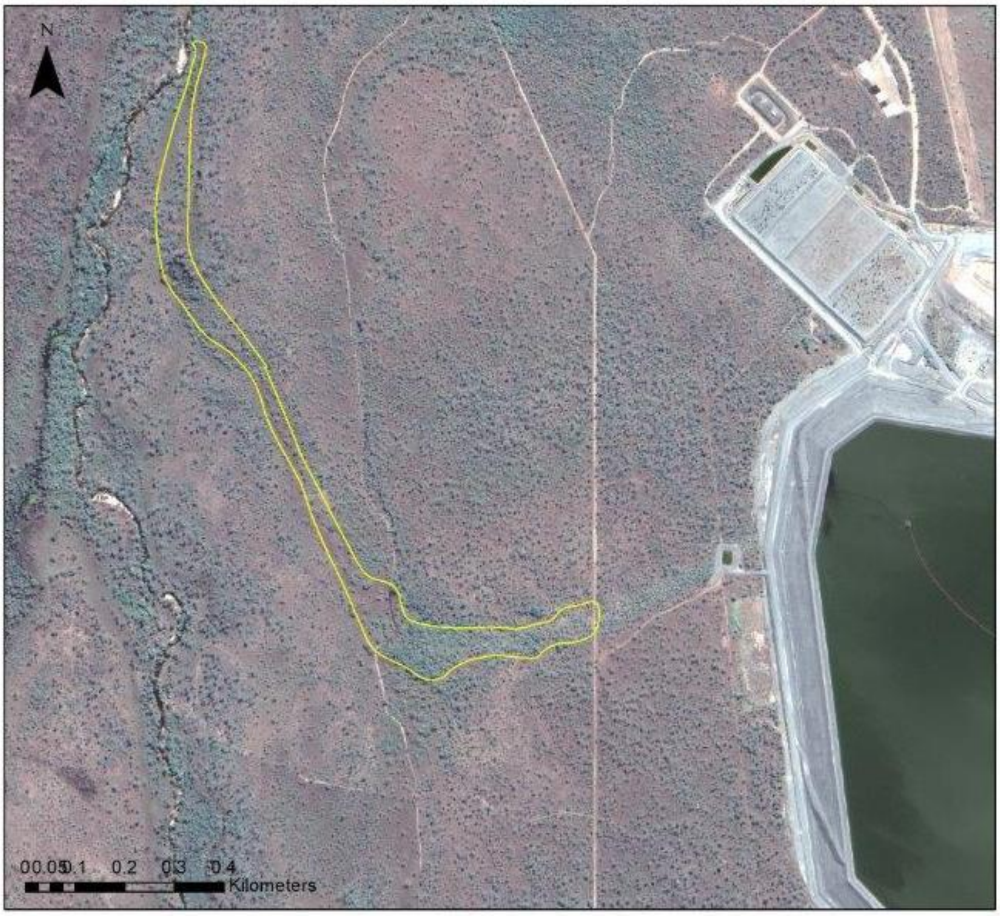


Figure 11 Spatial extent of salt formation along the GCT2 tributary, observed on 19 August 2014.

The deterioration in water quality along the GCT2 tributary has also been reflected in ERA's surface water monitoring sites located in its flow path, GCT2 and TWWS (Figure 1). The maximum EC levels measured over the last four wet seasons at GCT2, TWWS and at key monitoring sties in Gulungul Creek are compared in Table 2.

Table 2 The maximum EC recorded at ERA's monitoring points along the GCT2 tributary (GCT2 and TWWS) and at key monitoring points in Gulungul Creek (GCUS, GCDS and GCLB) over the past 4 wet seasons.

Site	2010-11	2011-12	2012-13	2013/14
TWWS[1]	N/A	7,400	9,150	12,950
GCT2	2,000	2,400	4,600	4,800
GCLB	52	56	53	184
GCDS	50	35	46	65
GCUS	33	39	113[2]	46

[1] The EC in TWWS represents the maximum antecedent dry season EC measured at the site prior to the first rainfall of the season.

[2] The high upstream EC was caused by solutes entering Gulungul Creek from the GCT1 tributary which connects with Gulungul Creek upstream of GCUS. Salts and high EC flows have been observed in the tributary in the past; however, this contamination is not discussed in this report.

The pH and EC values and magnesium (Mg), calcium (Ca), sulfate, potassium (K) and sodium (Na) concentrations from the impacted monitoring sites along the GCT2 tributary (TWWS, GCT2 and GCT2ST) were ordinated using multivariate analysis techniques and plotted in multi-dimensional space, along with corresponding water quality data from key sites that might present suitable surrogates for the potential source terms (Figure 12). Process water quality measured at TDWW was used as a surrogate for TSF seepage, water quality measured at TSFS2 was used as a surrogate for rock-wall run-off, water quality measured at TSFS1 was used as a surrogate for rock-wall infiltrate and water quality measured in the GCT1 tributary was used as a surrogate for background catchment run-off. It is acknowledged that this is not an exact approach as the data used do not accurately reflect the source terms considered, however it does provide a preliminary comparison between these key sites and a coarse assessment of the potential source analogues.

The spatial distribution of the data reflects the overall differences in the water quality based on the water quality parameters listed above. The plot shows (i) delineation between GCT1 (inferred background), TDWW (inferred TSF seepage) and GCT2, reflecting different water quality amongst these sites; and (ii) overlap amongst TSFS1 (inferred rock-wall infiltrate), TSFS2 (inferred rock-wall runoff) and TWWS, reflecting similar water quality amongst these sites. Principal component analysis (results not shown here) matched closely the MDS plot shown in Figure12 and indicated that pH is the key driver for the variability along the y-axis and that the major ions are the key drivers for the variability along the x-axis. In Figure , the GCT2 data are similar to the rock-wall data along the x-axis but are drawn downwards along the x-axis, towards the tailings data. This indicates that samples at GCT2 with lower pH resemble the (inferred) TSF seepage signature more closely than the (inferred) rock-wall signatures. The points that are circled in Figure 13 represent samples collected during the recessional flow period towards the end of the wet season. Their separation from the rest of the GCT2 data signify that there is a significant change in water quality at this time which is likely to be caused by the increasing influence of groundwater contribution to the site, and this change moves the samples in the direction of the tailings data. In order to explore potential inferred mixtures from the various source waters in multi-dimensional space, the mean of data from various multiple-site combinations were derived and the data re-ordinated with data from individual sites. These site combination data are plotted in Figure 12 (the filled diamond symbols). The inferred mixtures of rock-wall runoff and rock-wall infiltration waters with background catchment runoff (TSFS2 + GCT1 and TSFS1 + GCT1, respectively) plot above the GCT2 data. In contrast, the inferred mixture of tailings water with background catchment runoff (TDWW + GCT1) plots closely to the left of the GCT2 data, which could indicate a TSF seepage signal at GCT2. However, simple averaging of water quality data amongst sites does not take into account either the (unknown) ratio of proportional mixing.

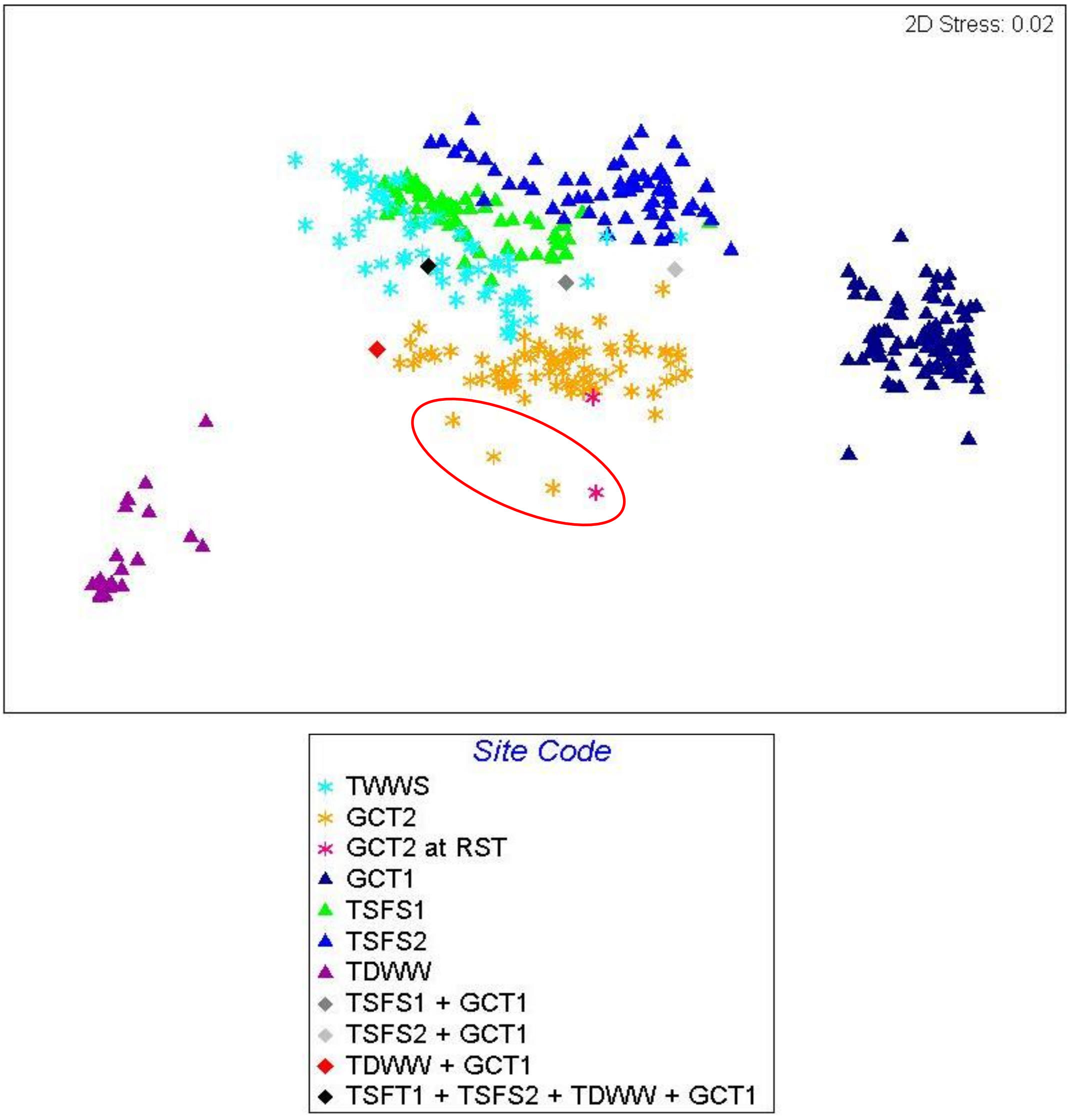


Figure 12 Multi-dimensional scaling (MDS) ordination based on Euclidean distance of water chemistry data for sampling locations along the GCT2 tributary and around the TSF.

Discussion

The information collected during the ERA and Supervising Scientist investigations confirm that the GCT2 tributary is a conduit along which high EC water from the minesite is transported into Gulungul Creek. In the receiving Gulungul Creek catchment, these solutes are causing degradation of water quality at various monitoring points during the wet season and the formation of evaporites during the dry season. They are also causing contamination of shallow groundwater, as measured in a number of monitoring bores in the vicinity of the GCT2 tributary (Figure 13). Bore RN23566, which is screened in Aquifer 1 at 4.02 – 4.52 m, is located adjacent to the GCT2 tributary about 700 m to the west of the TSF and has shown an increase in solute concentrations since 2008 (Figure 14).

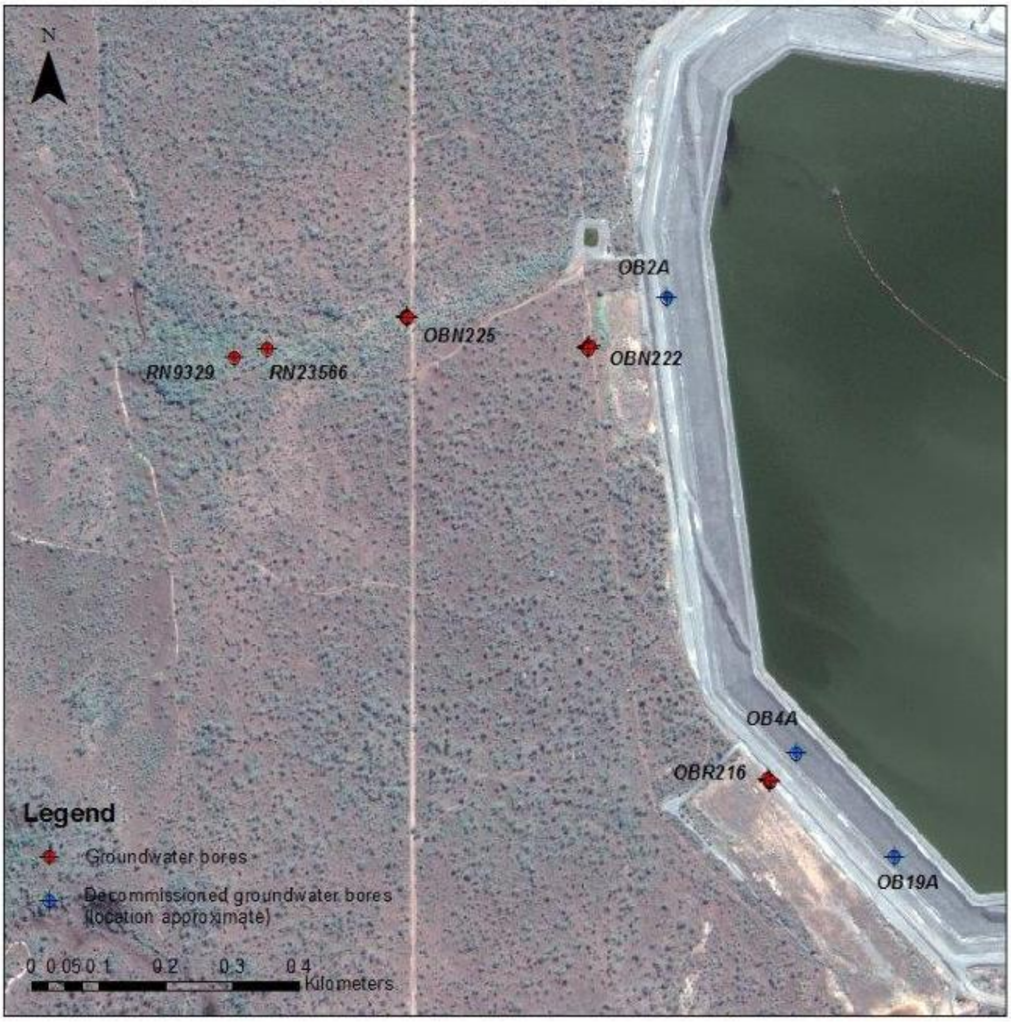


Figure 13 Location of key monitoring bores on the western TSF wall and in the vicinity of the GCT2 tributary. Red symbols show groundwater bores that are currently in use and the blue symbols show decommissioned bores that have be subsumed by the TSF walls.

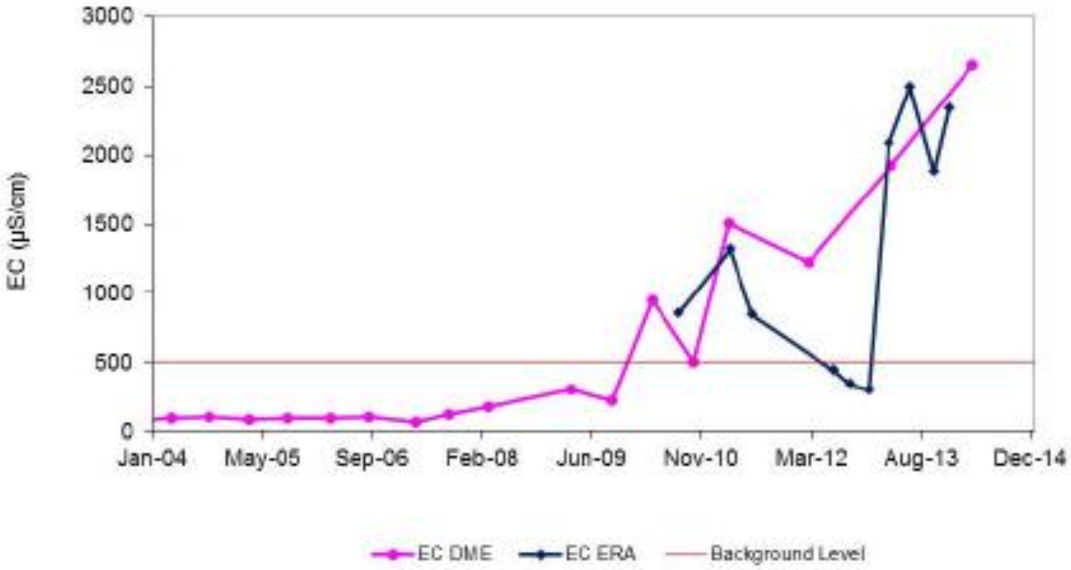


Figure 14 Electrical conductivity measured by DME and ERA in Bore RN23566 (DME 2014).

The increase in EC in RN23566 was initially detected by monitoring conducted by the Northern Territory Department of Mines and Energy, and prompted ERA to undertake an independent monitoring programme, commencing in July 2010 (O'Neill 2011). A nest of bores, OB225A, B and C, were installed upstream of the RN23566 according to recommendations made in the URS review (2010). The background chemistry conditions of Aquifer 1 are presented in Table 3 along with a summary of ERA's monitoring data for bores RN23566 and OBN225C.

Table 3 Summary of background chemistry conditions for Aquifer 1 (from URS 2010) and recent monitoring chemistry for RN23566 and OBN225C (Deacon, 2014).

Solute	**Background range**	**Range since Oct 2010 (OBN225C)**	**Range since Dec 2011 (RN23566)**
Screen depth (m)		**2 - 5**	**4.02 - 4.52**
EC	<500 µs/cm	2,000 - 6,000 µs/cm	250 - 2,500 µs/cm
Ca	<5 - 30 mg/L	50 - 275 mg/L	5 - 50 mg/L
Mg	5 - 50 mg/L	250 - 1,000 mg/L	25 - 375 mg/L
Mn	<5 - 350 µg/L	500 - 1,500 µg/L	50 - 600 µg/L
U	<10 µg/L	0 - 0.5 µg/L	0 - 1.5 µg/L
Sulfate	<5 mg/L	1,250 - 4,750 mg/L	100 - 1,600 mg/L

Both bores display EC values and Ca, Mg, sulfate and manganese (Mn) concentrations that are elevated compared to background. In contrast, the deeper groundwater observed in bores RN9329 (screened in Aquifer 2, at 17–19 m) and OB225A (screened in Aquifer 3, at 21–30 m) reflect the background values their respective aquifers, which may suggest that the solutes are confined to the shallow Aquifer 1. A number of investigations have been carried out in response to the elevated solute concentrations measured in shallow groundwater to the west of the TSF (O'Neill 2011, O'Neill & Tsang 2012). These investigations conclude that there was a top-down influence on shallow groundwater caused by the increased solute concentrations in the surface water flowing along the GCT2 tributary.

Identifying the source(s) of contamination in either surface water or shallow groundwater surrounding the TSF, including whether or not it is tailings-derived, has been a focus of many investigations since the 1980s. The three primary proposed sources of contamination in the Gulungul Creek catchment are: (1) seepage from the base of the TSF; (2) rainfall infiltration of the TSF rock-wall (rock-wall infiltrate); and (3) surface run-off from the TSF rock-wall. However the ability to delineate between the three is confounded by the fact that they all share a similar chemical signature (Alarcon Lean et al. 2007). Due to its conservative behaviour, sulfated has been found to be a useful indicator of groundwater contamination at Ranger (Klessa 2001). Sulfur isotope ratios were historically used as a reliable indicator of TSF seepage (leGras et al. 1991, 1992); however they became less useful over time due to changing sources of sulfur on the mine site (Jacobsen 2009). Many of the other key mine-derived contaminants are not useful indicators because they are reactive and do not travel conservatively in groundwater as they become adsorbed and bound to sub-surface materials or their speciation and solubility changes due to fluctuating redox conditions and pH (Duerden et al. 1992, Martin & Akber 1996, Sinclair 2004, Iles 2008, Jacobsen 2008, Alarcon Leon et al. 2007). Lowson and Jeffrey (1988) determined attenuation coefficients for a range of ions, which assisted in understanding these adsorption processes from tailings water down gradient of the TSF.

The routine monitoring carried out by ERA focuses on key mine-derived contaminants, including U, Mn and major ions. Analysis of some of these commonly measured variables in the multi-dimensional space suggests that the water quality at GCT2 is neither similar to those sites associated with rock-wall contaminants, nor the tailings alone, rather it resembles a mixture of three sources, including background runoff (Figure 12). While work carried out by ERA has concluded that the source of the contamination in the GCT2 tributary is the rock-wall of the TSF, in the absence of a clear TSF seepage chemical fingerprint, or signature, it is difficult to rule out a TSF seepage contribution. This is supported in the URS (2010) review which concluded that at the time, there was not enough information to distinguish the source of solutes in the Gulungul Creek catchment. The suite of elements measured routinely by ERA, particularly sulfate, are crucial for detecting off-site changes in water quality, however there may be other elements that are more suitable for identifying a chemical fingerprint or signature for TSF seepage and work should be carried out in the future to investigate this further. It would be very useful for samples from key monitoring sites to be analysed using full elemental scans by inductively coupled plasma mass spectrometry (ICP-MS) or similar.

In determining the source of the solutes it is also important to take into account the hydrogeology in the area. It is accepted that seepage-derived contamination expressing at the surface is most likely to occur where there are higher hydraulic conductivity pathways between the region below the TSF and the surface. Ahmad & Green (1986) used stable deuterium (δ^2H) and oxygen ($\delta^{18}O$) isotopes to compare the isotopic signature of groundwater from Aquifer 2 with TSF water and the results suggested there was no connectivity between the two in the region of the TSF. However, Hollingsworth & Klessa (2004) indicated that this connectivity is most likely to occur in the vicinity of fault lines and downgradient of locations where sulfate breakthrough has already occurred.

In order to identify and map significant faults in the vicinity of the TSF, Lu (2002) used a geophysical resistivity survey of the area along with groundwater chemistry data and aerial photography (in particular, drawing on a study by Verma & Salama, 1986). The five major faults identified are shown in Figure 15. They also found that the GCT2 tributary appeared to be coincident with a fault line that intersects the TSF (not shown). These faults could potentially act as a conduit for flow of tailings-derived water towards the west and possibly towards the surface. Faults F4 and F5 intersect one another in the vicinity of GCT2RST. Further investigation needs to be carried out to determine whether the low pH water and orange flocculant observed at GCT2RST is related to the presence and intersection of these faults, as this might indicate whether the iron-rich, anoxic water at GCT2RST is expressing from a shallow or deep groundwater source.

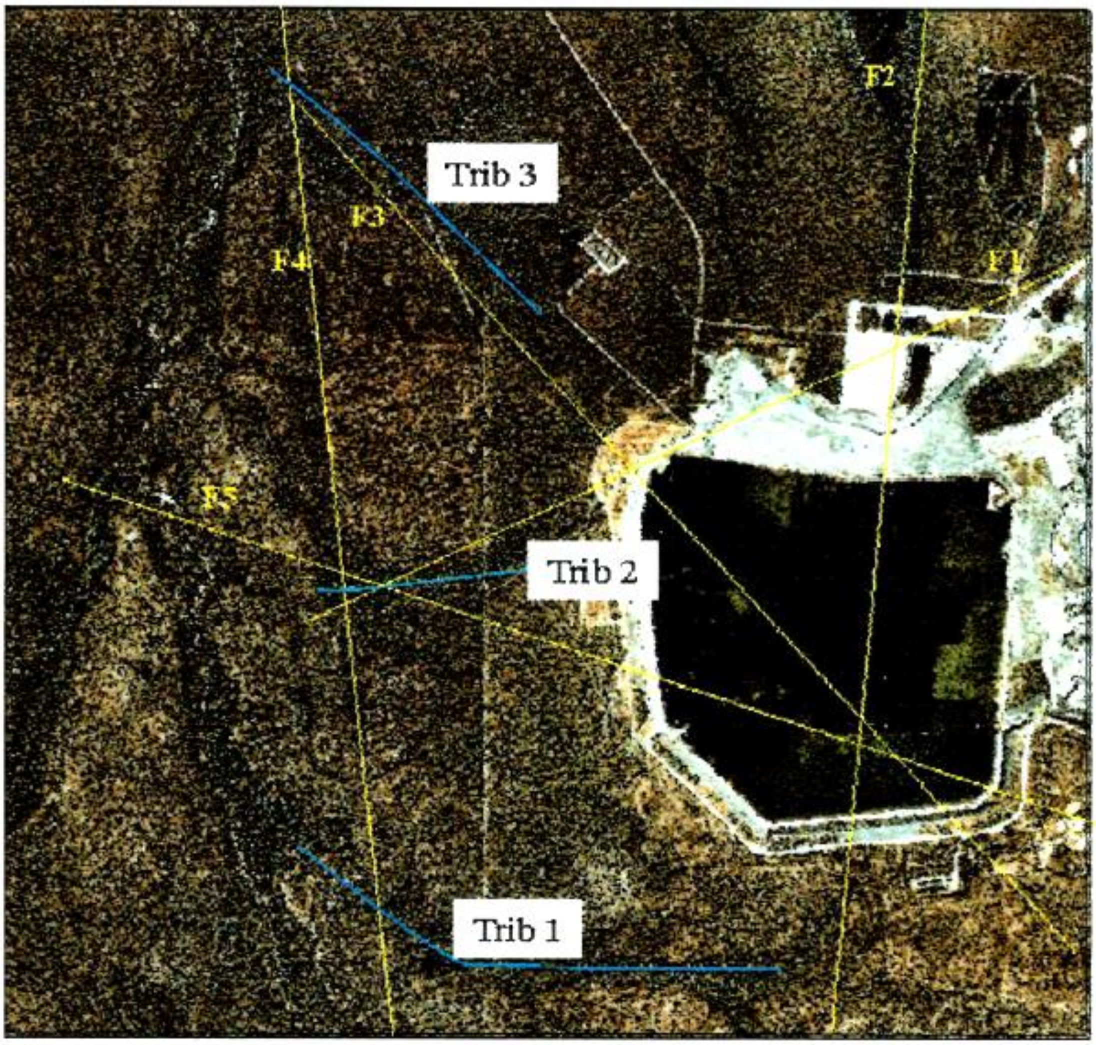


Figure 15 Faults that were identified as potential seepage paths from the TSF to Gulungul Creek by Lu (2002). Faults were deduced from previous reports and from aerial photographs.

Sulfate breakthrough is a term that describes a consistent increase in sulfate concentrations in groundwater monitoring bores that intercept the advancing front of the TSF seepage plume. This plume is likely to contain various other constituents derived from TSF seepage, however its chemistry has not been well described (URS 2010). URS (2010) indicated that bores OB2A, OB4A and OB19A had been contaminated due to sulfate breakthrough prior to being subsumed by the subsequent TSF wall expansions (Figure 16).

During the stage V TSF wall expansion, which was completed in 2007, seepage of water from the foundations of the TSF was observed in the vicinity of OB19A. This water was characterised as Mg and sulfate dominant and was attributed to rock-wall infiltrate and the possible presence of TSF seepage (Alarcon Leon & Pulhalovich, 2007). Similarly, during the stage VI TSF wall expansion, completed in 2008, seepage water was observed in the vicinity of OB4A and was determined to potentially contain a small percentage of TSF seepage (Jacobsen 2009). Alarcon Leon and Pulhalovich (2007) suggested that the pools observed near the base of the TSF were caused by the expression of deeper geological features during the wall expansions, as the excavations increased the groundwater pressure beneath the TSF and decreased the resistance to upwards flow of groundwater to shallow soils via localized faulted rocks.

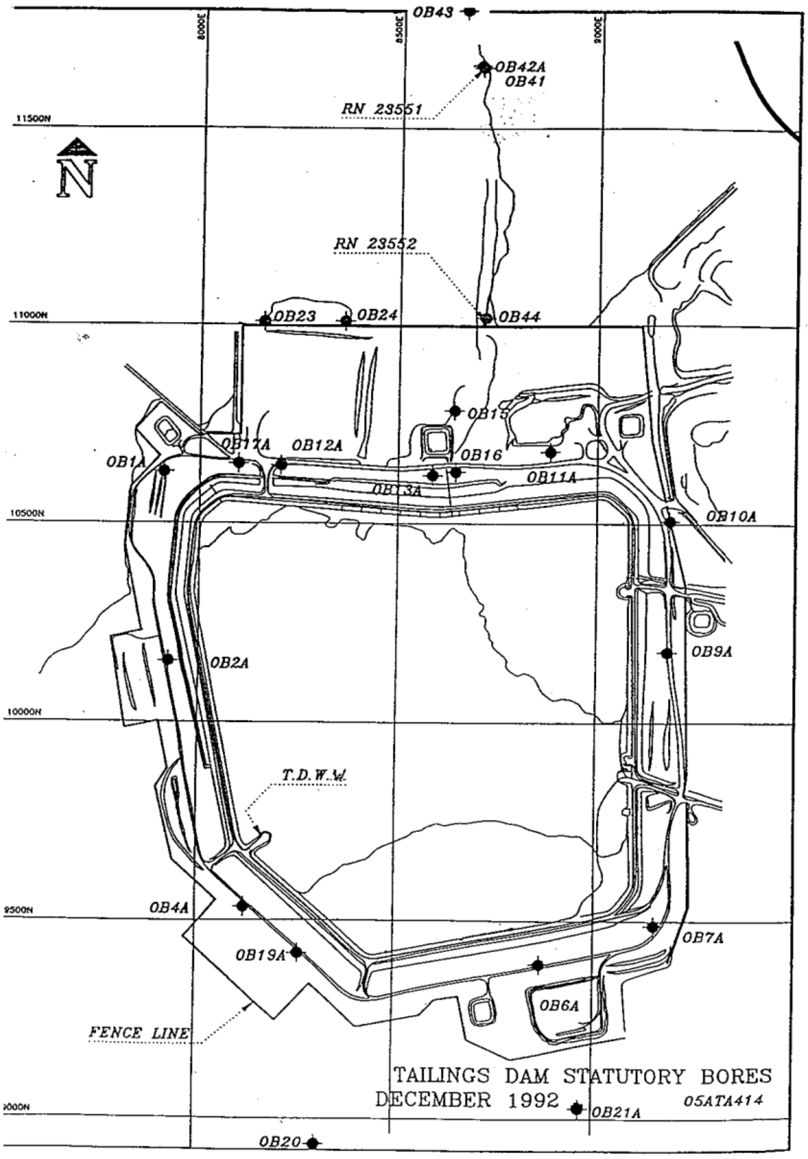


Figure 16 Map of early groundwater monitoring bores taken from Townley (1996) showing the locations of bores OB2A, OB4A and OB19A, where sulfate breakthrough had occurred.

The URS review (2010) identified that bores OB2A and OB4A were important for further defining the vertical extent of TSF seepage, recommending that these bores be re-established as nested sets of four monitoring bores and included into a routine, long-term monitoring programme. These nested bores have been installed as recommended, with the nest OBR222A, B, C and D installed in proximity to OB2A and the nest OBR216A, B, C and D installed in proximity to OB4A (Figure 13). Data from these bores and others installed based on the URS (2010) recommendations have been reported by ERA on a quarterly basis, and are summarised in the routine Annual Ranger Ground Water monitoring Reports (Deacon 2014, Deacon 2013). However these data are yet to undergo detailed analyses and interpretation and at this point in time:

"…The Tailings Storage Facility Groundwater Monitoring Programme (TSFGWMP) results to date show there are isolated areas of elevated parameters which may indicate potential tailings water seepage extending laterally from the TSF. These results will require further rounds of monitoring and analyses to confirm whether there is an overall increasing trend developing..."

The interpretation of these data, along with data collected from various other bores that are monitored as part of ERA's TSF routine monitoring programme, is critical in determining the current location of the seepage plume and the rate and direction of its progression. ERA is encouraged to progress this analysis as a priority. Klessa & Welsh (2003) reviewed previous studies of solute movement from the TSF, and the expression of solutes in the Gulungul Creek catchment. They specifically focused on implications of further toe loading of the southern wall of the TSF on future water quality in Gulungul Creek, and found that:

"the current flux of solutes in a westerly direction from the tailings dam is incompletely described and understood, and, in future, may modify water quality in Gulungul Creek non-linearly and to a greater extent than is currently estimated".

Over time ERA have implemented a number of systems to facilitate the management of poor quality water collected around the base of the TSF, including various collection sumps that can be pumped to other on-site waterbodies depending on their water quality. In the past these systems have prevented the poor quality water to the west of the TSF from having measurable off-site impacts. However these systems now need to be upgraded to ensure that the poor quality water to the west of the TSF is contained, regardless of its source. Ongoing monitoring will be required to ensure that the upgraded system is working as required and as expected.

Conclusion

This report presents the details and results of field investigations that were carried out by ERA and the Supervising Scientist in relation the high EC spikes measured in some locations in Gulungul Creek during the 2013/14 wet season. It also proved a summary of previous investigations into the increased solute concentrations in surface water and shallow groundwater to the west of the TSF. There is sufficient evidence showing that input of solutes from the western area of the TSF is causing the water quality in the GCT2 tributary to deteriorate and is now causing elevated EC in parts of Gulungul Creek. The report does not identify the specific source of these solutes, however it presents information to suggest that TSF seepage cannot be ruled out as a potential contributor, including:

- TSF seepage is occurring according to the estimated rates predicted using various groundwater modeling techniques, however analysis and interpretation of the TSF groundwater monitoring data collected to date is not sufficient for determining the full extent of groundwater seepage and understating the connectivity between deep and shallow groundwater;
- Evidence of sulfate breakthrough to the south-west of the TSF that has previously expressed at the surface and could contribute increased to Mg and sulfate loads in the catchment;
- The occurrence of a fault along the GCT2 tributary and the intersection of two major faults upstream of the GCT2RST site which could potentially provide a preferential flow pathway for contaminated groundwater; and

- Statistical analysis indicate the water occurring at GCT2 is only slightly similar to the water derived from rock-wall runoff and infiltrate and is possibly influenced by TSF seepage.

The Supervising Scientist proposes that ERA restrict the movement of solutes into the Gulungul Creek catchment by upgrading their water management systems to the west of the TSF. Further work should also be carried out to ensure the ongoing protection of the environment in Kakadu National Park downstream of the mine site, including but not limited to:

1. Developing a reliable method for detecting a tailings seepage signal and using it to delineate the source of solutes in the surface water and groundwater mixtures in the Gulungul Creek catchment. This method will not only be useful for gaining a better understanding of the current surface water/groundwater interactions in the Gulungul Creek catchment but is also important for future monitoring of the area.

2. Reviewing and collating all information available on groundwater contamination, hydrogeology and key potential groundwater flow pathways in the vicinity of the TSF. In particular, the preferential, highly permeable underground flow pathways associated with fault lines, old uncapped bore holes and the presence of sandy alluvial sediments associated with creek channels, such as the GCT2 tributary, should be investigated. This information should be assessed to design an updated conceptual model that clearly illustrates the lateral movement of solutes in groundwater in the vicinity of the TSF.

3. Development and implementation of specific Water Quality Objectives for Gulungul Creek that will enable the risk posed by the potential ongoing contribution of solutes to the catchment to be assessed. This will also provide a means by which active water management strategies can be implemented in the Gulungul Creek catchment in order to control the environmental risk.

A programme of possible work has been scoped by the Supervising Scientist and is presented in Appendix A. This work is only suggested at this stage and no commitment is made by the Supervising Scientist to undertake any of the potential projects or tasks except for those already underway, which include:

- Undertaking direct toxicity assessment of the GCT2 tributary surface water;
- Installing an additional continuous monitoring station near ERA's GCLB site;
- Assessing aerial imagery for signs of vegetation die back;
- Characterising the mineralogy of the evaporites using visible and near-infrared reflectance data coupled with x-ray diffraction data; and
- Working collaboratively with ERA to implement a wet season monitoring programme that (i) detects and accurately quantifies the affects of the solutes on water quality in the Gulungul Creek catchment; and (ii) assess the long-term effectiveness of the surface water management systems in place to the west of the TSF.

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Toxicity of ammonia to local freshwater biota

T Mooney, C Pease, A Hogan & A Harford

Background

Although ammonia (NH_3-N) is present at very high concentrations in Ranger process water (~1000 mg L^{-1} Total Ammonia Nitrogen, TAN (NH_3-N), to date it has presented negligible environmental risk as process water is not discharged to the off-site environment. However, these risks may increase in the future through (i) the potential presence of residual amounts of ammonia in treated process water distillate from the brine concentrator (~1 mg TAN L^{-1}, Harford et al. 2013); and (ii) potential seepage of ammonia from in-pit tailings deposits post-closure. Consequently, a site-specific water quality Guideline Value (GV) for ammonia in Magela Creek is required. This project aims to understand ammonia toxicity under physico-chemical conditions relevant to the off-site surface water environment (i.e. Magela Creek – channel and billabongs) to a range of local freshwater species.

The results from toxicity tests using six native local species will be used to derive site-specific GVs that can be incorporated into the regulatory regime. Until this study is completed, an interim 99% GV for ammonia of 0.7 mg TAN L^{-1} has been calculated and applied at MG009 as part of the approval for ERA to discharge distillate from the Ranger brine concentrator to Magela Creek (via Corridor Creek) during the 2013–2014 wet-season. The interim GV was derived using the large body of literature concerning ammonia toxicity, and was largely based on the recent USEPA (2013) updated water quality criterion for ammonia.

Ammonia is readily miscible with water and will undergo ionisation to form the ammonium ion (NH_4^+), the degree of ionisation being dependent on pH and temperature, with higher pH values leading to decreased ionisation. Ammonia toxicity is predominantly caused by the unionised from (NH3), thus, GVs should be adjusted for temperature and pH to account for these important changes in speciation and toxicity.

Preliminary toxicity testing of ammonia was undertaken for one local species, the green hydra, *Hydra viridissima*. This species was found to be highly sensitive to ammonia where test water was at pH 8 and at a temperature of 27.5°C, with EC10 and EC50 (i.e. concentrations causing 10 and 50% effect in population growth) values of 0.2 and 1.6 mg TAN L^{-1}, respectively (van Dam et al. 2011). Ammonia toxicity under typical physico-chemical conditions of Magela Creek water will, however, very likely be lower than this, as the pH of the creek typically ranges from 5.4 to 6.4, where the proportion of the more toxic NH_3 is approximately 30 times lower than at pH 8.0 (Emerson et al. 1975). To ensure toxicity estimates from laboratory tests are accurate, pH and temperature need to be closely monitored and controlled. Controlling pH to the level required in the poorly buffered, soft-waters of Magela Creek provides a particular challenge. Therefore, an appropriate pH buffering system for each test species required investigation. The bicarbonate buffering system was identified as a potentially appropriate system for at least half of the suite of test species in the present study. This system has been used successfully in the literature to control pH during whole effluent toxicity testing, and was our primary buffer choice (Elphick et al. 2005). However, this system may not be logistically and technically possible to use, depending on the toxicity test protocol. In cases where a carbonate buffer is not appropriate, the use of an organic pH buffer (HEPES, MES or MOPS) will be investigated.

To test and potentially modify the existing interim ammonia GV for Magela Creek, laboratory toxicity testing will be used to derive a site-specific GV. Six local native species will be tested at a pH and temperature relevant to the site-specific physico-chemical conditions of Magela Creek.

Methods

The toxicity of ammonia is currently being assessed using six local tropical freshwater species: the unicellular green alga (*Chlorella* sp.); the duckweed (*Lemna aequinoctialis*); the green hydra (*H. viridissima*); the cladoceran (*Moinodaphnia macleayi*); the aquatic snail (*Amerianna cumingi*) and the Northern trout gudgeon (*Mogurnda mogurnda*). Standard toxicity testing protocols detailed in Riethmuller et al. (2003) and Houston et al. (2007) were used unless otherwise stipulated (Table 1). To date, ammonia toxicity testing has been completed for *H. viridissima, M. mogurnda* and *Chlorella* sp. The diluent, Magela Creek Water (MCW), was spiked with ammonia using stock solutions of ammonium sulphate (which strongly dissociates and is very soluble in water; Weast et al. 1988). Actual concentrations of ammonia in solution were checked before and after the test exposure, at ***eriss***, using spectrophotometry (colourimetric method EPA 350.1). All ammonia concentrations are reported in total ammonia nitrogen (TAN).

A starting pH of 6.0 was the target for the toxicity tests and test solutions were kept within 0.3 units throughout the test. Temperatures were maintained as per testing protocol (Riethmuller et al. 2003, Houston et al. 2007), at 27.5°C or 29°C ± 1°C, depending on test species. To control pH throughout testing the bicarbonate buffering system was used where logistically and technically possible. This system involved conducting toxicity testing in air-tight chambers and increasing the atmospheric CO_2 concentration within the system, following the principles of Elphick et al. (2005). Specifically, as the partial pressure of CO_2 in the headspace increases, dissolution of CO_2 into the water is increased. Some of the dissolved CO_2 will react with water to form carbonic acid which disassociates into a carbonate ion and two hydrogen ions, resulting in a decrease in pH. The rate of dissolution of CO_2 and the aqueous concentration of carbonic acid depends on the partial pressure of CO_2 and the surface area of the solution. This system was used during ammonia toxicity testing on H. viridissima and M. *mogurnda* and we expect to use it to control pH during *M. macleayi* testing. This method has the dual advantages of 1) being able to decrease the initial pH of the test solution to the desired pH and; 2) maintain pH throughout testing, without having a toxic effect on the test organism. The carbonate buffering system was not required for *Chlorella* sp. because the standardised protocol includes the use of 1 mM HEPES buffer and this adequately maintained pH during toxicity tests (Riethmuller et al. 2003). Due to logistical or technical reasons the bicarbonate buffering system was not able to be used during toxicity testing with*, L. aequinoctialis* and *A. cumingi.* Specifically, the large volume of water and large testing vessels used in *A. cumingi* tests means that it is not possible to conduct tests in the air-tight chambers, while initial ammonia toxicity tests with *L. aequinoctialis* using bicarbonate buffering resulted in a marked reduction in pH (see the results section). Consequently, trials are currently underway to determine the most appropriate organic buffer (HEPES, MES or MOPS) to use during *L. aequinoctialis* and *A. cumingi* testing.

At least two valid toxicity tests were completed for each species and, for some of the toxicity tests, a modified treatment replication design was used (Table 1), whereby the number of different concentrations tested was increased by reducing treatment replication. Due to some species having higher variability in endpoint performance, the modified design could not be used for the *A. cumingi, M. macleayi* or *Chlorella* sp. toxicity tests. The *Chlorella* sp. testing protocol was modified from Riethmuller et al. (2003) to reduce the algal density to 3 x 103 cells mL^{-1} and nutrient concentrations (nitrate and phosphate were reduced to a 6^{th} of the original nutrient content). This had the advantage of providing better control of water quality parameters during testing.

Table 1: Details of the ammonia concentration-response tests conducted to-date.

Test ID	Date	Species name	Endpoint	Ammonia concentration range tested (mg L^{-1}) a	Comments
1382G 1394G	25/02/14 3/03/14	*Chlorella* sp.	Population growth	2.5 – 80 7.5 - 240	Lower nutrients/ lower density c; 1 mM HEPES
1391B 1401B	7/04/14 28/04/14	*H. viridissima*	Population growth	1 – 32 1.5 – 24	Modified design b; Bicarbonate buffer
1410E 1413E	14/06/14 23/08/14	*M. mogurnda*	Survival	10 – 320 15 - 60	Modified design b; Bicarbonate buffer
1421L	25/08/14	*L. aequinoctialis*	Surface area growth rate	2 -64	Modified design b; Bicarbonate buffer
1405S (HEPES) 1411S (MES)	26/05/14 23/06/14	*A. cumingi*	Reproduction	N/A	As per protocol

a Concentration range is based on the mean of start and end ammonia values
b A modified design of less replicates and more treatments was used.
c Protocol in Riethmuller et al. (2003) modified to reduce nutrients and density of algae required.

Results and Discussion

The relationship between pH, temperature and ammonia toxicity has been extensively characterised by the USEPA (1999) and the dissociation constants (pKa) for ammonia have been reported by Emerson et al. (1975). These data allow the guidance documents (ANZECC & ARMCANZ 2000) to provide algorithms that allow adjustment of GVs for a specific pH and temperature. In this study, instead of using an adjustment algorithm for the derivation of a site-specific GV for Magela Creek, tests were maintained within 0.3 of a pH unit at an ecologically relevant pH of 6.0 (Table 2), which is the median pH during the wet season in Magela Creek. The starting pH in the *M. mogurnda* tests were lower than pH 6.0 (Table 2), this was the result of increasing the concentration of CO_2 during testing to gain greater pH control for the duration of the test. Acceptable amount of pH drift was chosen based on what was technically possible to achieve. While the use of a buffer allowed for greater pH control, pH drift commonly still occurred. At this pH, the percentage of un-ionised ammonia in solution is extremely low and much of the toxicity can be attributed to ionised ammonia (ANZECC & ARMCANZ, 2000). Therefore, pH drift reported in the present study would have negligible effect on toxicity.

Table 2 Summary of control water physico-chemistry for toxicity tests completed to-date.

Species	Test	pH[c]		EC (µS/cm)		DO (%)		DOC (mg/L)	Alkalinity (mg/L CaCO3)
		new	old	new	old	new	old		
Chlorella sp.	1382G	6.0	6.3	22	31	107	96	3.6	2
	1393G	6.0	6.2	22	21	104	95	3.6	2
Mogurnda mogurnda	1410E	5.8[a]	6.2[a]	14	17	99	83	2.1	2
	1413E	5.7[a]	6.1[a]	14	18	110	86	2.1	2
Hydra viridissima	1391B	6.1	6.2	12	14	92	100	3.7	2
	1401B	6.1	6.3	15	17	95	100	2.5	<1
Lemna aequinoctialis	1421L	6.1	6.1	23	16	113	101	1.5	3
Buffer Trials									
Amerianna cumingi	1405S	6.3[b]	6.8[b]	12	33	99	80	1.5	1
	1411S	6.3[b]	6.9[b]	14	23	101	84	2.1	2

[a] Measured pH values were rounded to the nearest 0.1 of a pH unit, therefore pH drift appears to be >0.3 of a pH unit.

[b] Buffer trial, control test waters were not buffered.

[c] Reported pH values are the range over a whole test. Drift in pH was less for the majority of 24hr periods for the *H. viridissima* and *M. mogurnda* tests.

Of the species tested, ammonia toxicity varied markedly (Table 3; Figure 1). *Hydra viridissima* was the most sensitive species followed by *M. mogurnda* and *Chlorella* sp. (Table 3; Figure 1). Testing is currently underway for the three remaining species. Completed toxicity tests have yielded results consistent with the current literature in regards to species sensitivities (USEPA 2013). These have been detailed previously in (van Dam et al. 2011). However, in the international literature two species of unionid freshwater mussel are of particular note due to their high sensitivity to ammonia; *Lampsilis siliquoidea* had an EC20 of 3.2 mg TAN L^{-1} and *Lampsilis fasciola* of 1.4 mg TAN L^{-1} after normalisation to a pH of 7 and a temperature of 20°C (USEPA 2013). As a result, investigations are underway to develop a freshwater mussel toxicity test using a native species of unionid mussel, *Velesunio angasi*, which is known to inhabit Magela Creek and to be an important bush food for communities downstream of Ranger. Upon development of a successful test, ammonia toxicity results for the mussel will be included in future site-specific GV revisions for Magela Creek.

Table 3 Preliminary ammonia toxicity estimates (± 95% confidence limits) to 3 local freshwater species in Magela Creek Water

Species	IC10 (mg TAN L^{-1})a	IC50 (mg TAN L^{-1})b
Chlorella sp.	70 (45 – 85)	230 (200 - 260)
H. viridissima	1.5 (1 – 3)	8 (6 – 10)
	LC05 (mg TAN L^{-1})c	**LC50 (mg TAN L^{-1})c**
M. mogurnda	28 (1 – 33)	42 (40 – 44)

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

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

[c] Toxicity estimates for *M. mogurnda* are LC05 and LC50, that is the concentration that results in 10 and 50% reduction in the survival of the fish

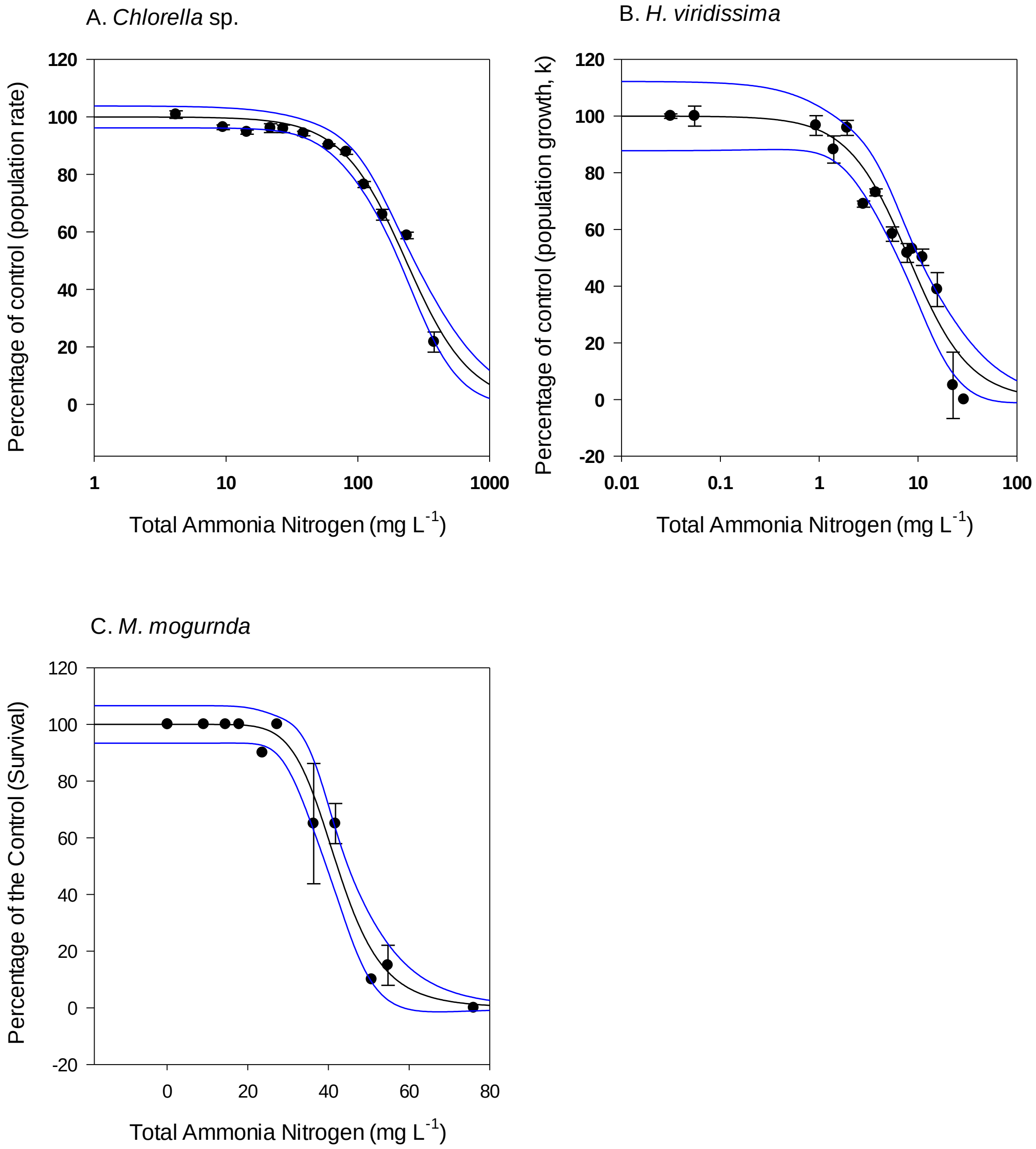

Figure 1 Effect of ammonia on A) *Chlorella* sp. growth rate B) *H. viridissima* population growth rate and; C) *M. mogurnda* survival. Data points represent the mean ± standard error of 2-3 replicates. 3-parameter logistic models were used to determine toxicity estimates for all species. Test were conducted at pH 6 ± 0.3 and temperatures of 27.5°C ± 1°C for *M. mogurnda* and *H. viridissima,* and 29°C ± 1°C for *Chlorella* sp.

Buffer trials

Logistically, it was not possible to implement the bicarbonate buffering system during *A. cumingi* ammonia toxicity testing due to the large volume of test water required and the size of the exposure vessels. Therefore, trials are underway to determine an appropriate buffer to use. Once appropriate pH buffers are identified ammonia toxicity testing can commence. To date, two buffers have been trialled for *A. cumingi,* HEPES and MES. Exposure to both buffers resulted in a concentration-dependent reduction in *A. cumingi* egg production (Figure 2). Furthermore, the pH control during the HEPES test was unacceptable (Figure 2a), ruling this buffer out as a potential option. While pH control using MES buffer was at an acceptable standard (Figure 2b), there was an 18% effect at the lowest concentration of 1 mM MES, indicating further testing is required.

The bicarbonate buffering system was trialled during ammonia toxicity testing with *L. aequinoctialis*. This buffering system was not effective and the test failed due to a decrease in pH to 4.5 at ammonia concentrations of 2 mg TAN L^{-1}, with pH concomitantly increasing as TAN concentrations increased (Figure 3). We believe this was caused by an interaction between *L. aequinoctialis* and ammonia, because there was no decrease in pH observed in the controls (Figure 3). The metabolic pathway of ammonia nitrification might be the cause of the pH decrease, because it results in the generation of H^+ ions. However, nitrifying bacteria would need to be present in order for this metabolic pathway to be complete. More testing is required before a definitive conclusion can be drawn. Since the bicarbonate buffering system was not effective in this instance, trials will be undertaken to find a suitable buffer that can effectively control decreasing pH.

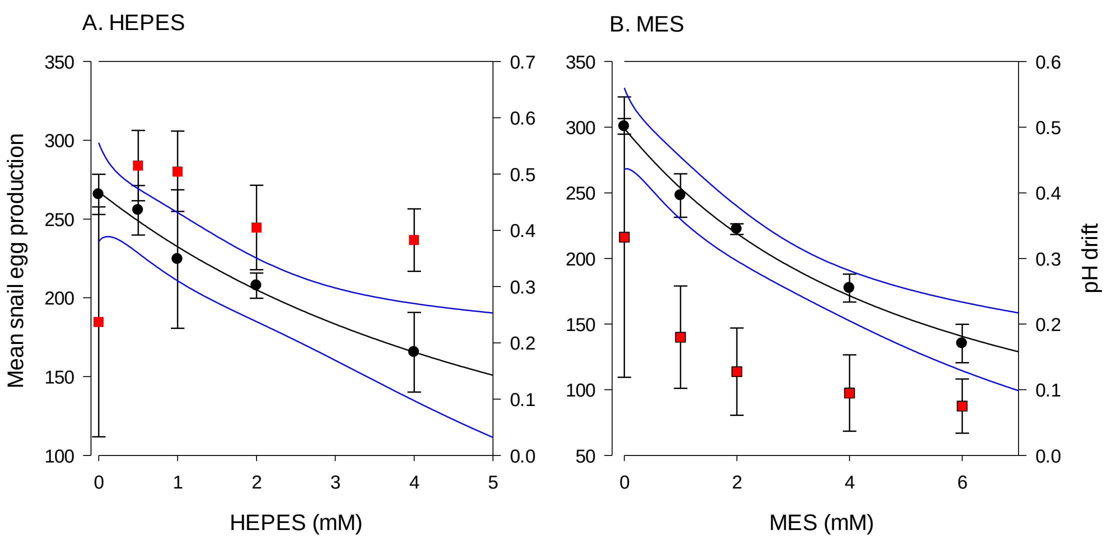

Figure 2 Effect of A) HEPES and B) MES on *Amerianna cumingi* egg production. Black data points represent the mean ± standard error of 2–3 replicates. Red data points represent pH drift ± standard error of 2–3 replicates and correspond with the right y-axis. 3-parameter logistic models were used to determine toxicity estimates.

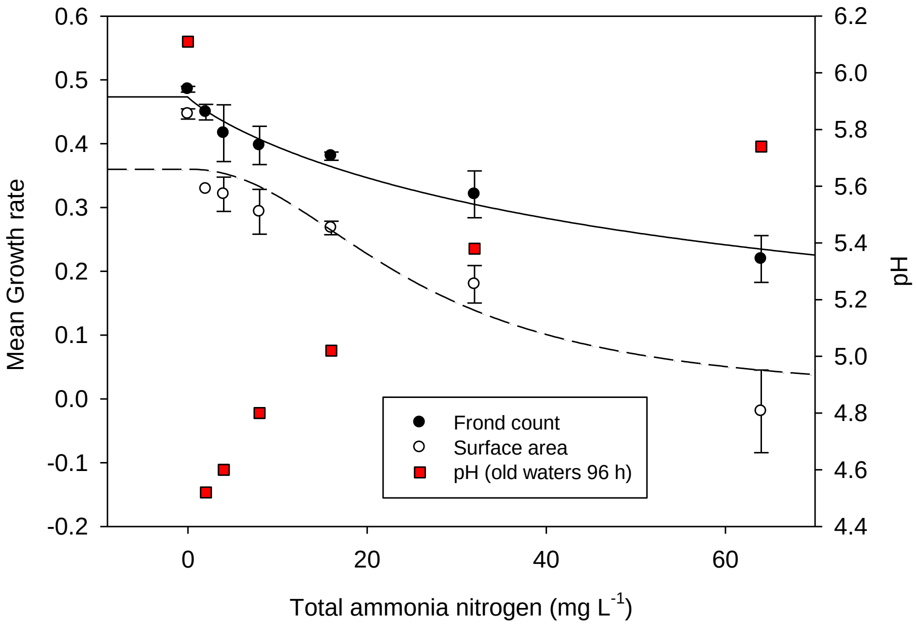

Figure 3 Effect of ammonia on *L. aequinoctialis* frond and surface area growth rates. Black (frond count growth rate) and white (surface area growth rate) data points represent the mean ± standard error of 2–3 replicates. Red data points represents end of test pH ± standard error of 2–3 replicates, corresponding with the right y-axis. 3-parameter logistic models were used to determine toxicity estimates.

In conclusion, ammonia toxicity testing has been successfully completed for three of the six local species. These species showed a wide range of sensitivities, with *H. viridissima* being the most sensitive and the second most sensitive species reported in the literature to-date (USEPA 2013), despite the low pH at which the tests were conducted. Toxicity testing is still required for three species: *L. aequinoctialis, A. cumingi and M. macleayi*. To control pH, the bicarbonate buffering system has been highly effective for some species, but was not suitable or practical for others. In these instances, other buffers, such as MES, HEPES and MOPS are currently being trialled.

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The sensitivity of *Moinodaphnia macleayi* to uranium

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Introduction

The cladoceran *Moinodaphnia macleayi* has been used by ***eriss*** as a standard toxicity testing organism to assess the toxicity of mine waters and their toxic constituents for over 20 years (McBride et al. 1991). The chronic toxicity of uranium (U) to *M. macleayi* has been assessed on several occasions (e.g. Hyne et al. 1993, Semaan et al. 2001, ***eriss*** unpublished data 1992, 2011 & 2012), and this species was found to be the most sensitive of the five local species used to derive a site-specific water quality guideline for U in Magela Creek (Hogan et al. 2005).

Cladocera require feeding during toxicity testing in order to maintain survival, growth and reproduction (Norberg & Mount 1985). Optimal culture health and environmental relevance of test results is more readily achieved through the provision of a ration that reflects the organism's natural diet. However, the need for standardisation in laboratory testing requires a compromise, usually in the form of a more standardised and/or defined diet. Hyne et al. (1993) hypothesised that the natural diet of *M. macleayi* is likely to consist of unicellular algae, small organic particles and bacteria, because other species of cladocera were shown to utilise these food sources (Kobayashi 1991) and *M. macleayi* were known to inhabit weed bed environments (Julli 1986). In our laboratory, *M. macleayi* are fed a fermented bacterial suspension that also contains organic particulates (Fermented Food with Vitamins; FFV), and a unicellular alga, *Chlorella* sp., during both culturing and testing. The introduction of the FFV in the early 90's resulted in a marked improvement in the survival and fecundity of *M. macleayi*, and was adopted as standard practice for this species (Hyne 1991). The FFV is prepared by fermenting ground fish food and alfalfa in water for several days before settling, straining and then adding vitamin B12 and calcium pantothenate (vitamin B5). The cladocera are fed 30 µL of this suspension, which is equivalent to 2–4.5 mg L^{-1} Total Organic Carbon (TOC). *Chlorella* sp. is cultured under axenic conditions in a standard nutrient medium and cell density is enumerated prior to calculating a consistent feeding ration of 6×10^4 cells mL^{-1}.

The establishment of a reference toxicity testing programme for the ecotoxicology laboratory in 2006 enabled the tracking of test organism performance over time (Cheng et al. 2012). In the first few years of this programme it was noted that the sensitivity of *M. macleayi* changed significantly on occasion. As this only occurred infrequently these tests were considered outliers, but as the dataset increased it became apparent that the shifts in sensitivity were a real issue for this species. The overall range of LC50s (median lethal concentrations) observed in reference toxicity testing over the past eight years has varied from 9 to 240 µg L^{-1} U (Pease et al. 2013). A similar level of variability has been found in chronic (3 brood reproduction) tests using this species (van Dam et al. 2012).

In trying to resolve this issue, several stand alone experiments, analyses and trials have been conducted. These include undertaking side by side comparisons of *M. macleayi* performance and sensitivity using different batches of FFV and characterising the bacterial assemblages and organic carbon content of different FFV batches. This project has been created to formalise the collation of these data and to undertake further laboratory work to elucidate the cause of the reduction in sensitivity and trial potential replacement food types.

This project aims to:

1. Document the temporal changes in sensitivity of *M. macleayi* to U and elucidate the cause for this shift in sensitivity; and
2. Alter the feeding procedure to create a more standardised testing environment where *M. macleayi* is found to be consistently sensitive to U and other toxicants.

Summary of other research for this project

Concurrent reference toxicity tests undertaken in previous years confirmed that the sensitivity of *M. macleayi* is strongly influenced by the batch of FFV they are fed during testing. However, sensitivity was not correlated with the organic carbon concentration of the FFV across 14 tests (r^2 = -0.07). There was also no correlation between test water pH and sensitivity (r^2 = -0.20, n = 29), nor the organic carbon concentrations of the algal ration and sensitivity (r^2 = 0.14, n = 6), indicating that these factors were unlikely to be responsible for the observed variability.

Nuclear magnetic resonance (NMR) analysis undertaken by the University of Melbourne suggested that the metal binding capacity of the FFV may have increased after a batch of dried alfalfa used for FFV preparation was replaced. Nuclear magnetic resonance spectroscopy provided a semi-quantitative measure of the functional groups on the organic carbon molecules. The carboxylic groups are known to have the highest affinity for metals and therefore considered the most influential with respect to metal complexation (Tipping 2002). When normalised according to TOC concentration of the sample, the carboxylic group peak heights were higher in FFV produced from the new batch of alfalfa compared to those produced with the older alfalfa. This corresponded with a reduction in cladoceran sensitivity in the reference toxicity testing programme and indicated that the structure of organic carbon may be more important than the overall TOC concentration. This hypothesis is supported by work done by Kolts et al. (2008) who directly related cladoceran sensitivity, not only to the quantity, but also the source of dissolved organic matter in the test system.

Microbial characterisation of nine different batches of FFV showed large differences in species composition and diversity across batches. While there was insufficient data to relate this information to sensitivity in reference toxicity testing, there appears to be no relationship between the dominant microbial species, or diversity of species, and cladoceran culture health. As such, we were unable to identify a preferred species of bacteria for isolation and use as an axenically cultured bacterial source.

Despite these investigations being unable to explain the cause of the variability in cladoceran sensitivity, the concurrent tests using different FFV batches clearly demonstrate that FFV batch was a key factor. Accordingly, recent efforts in the ecotoxicology laboratory have been focused on finding a replacement food source for *M. macleayi*. The current paper focuses on these food trials.

Methods

Off-the-shelf food products

Nutraplus

Nutraplus is an off-the-shelf red algal suspension that is used as a food source in the aquaculture industry. This product has successfully been used to maintain cultures of *Ceriodaphnia dubia* at the CSIRO Biogeochemistry Programme's Ecotoxicology Laboratory, when provided alongside a fresh unicellular algal ration.

Two dilutions of Nutraplus, 1:1000 and 1:2000 were chosen as they were equivalent to that provided to *C. dubia* at CSIRO. The Nutraplus stock was fortified with calcium pantothenate and Vitamin B12 at the same rate normally added to FFV and fed to *M. macleayi* in combination with the regular algal ration. The trial was conducted alongside the *M. macleayi* culture, which was used as the control treatment. All treatments consisted of twelve vials, each containing an individual female neonate. Natural Magela Creek water (NMCW) was used as the diluent. The cladocera were observed and transferred daily into freshly prepared waters containing food.

Yeast

The yeast ration trialled was based on attaining 4.5 mg L^{-1} TOC, which is equivalent to the upper end of the organic carbon concentration of FFV typically fed to *M. macleayi*. Trials were conducted as described above for Nutraplus, including the fortification of the feed ration with calcium pantothenate and Vitamin B12.

Probiotics

Orthoplex Multiflora is a probiotic formulated for human use containing twelve different species of lyophilised bacteria from the genera *Lactobacillus*, *Bifidobacterium* and *Streptococcus*. The amount of Orthoplex Multiflora provided during trial was also based on attaining a concentration of 4.5 mg L^{-1} TOC. Trials were conducted as described above for Nutraplus, including the fortification of the feed ration with calcium pantothenate and Vitamin B12.

Protexin Professional is a probiotic formulated for veterinary and livestock use containing seven different species of lyophilised bacteria from the genera *Lactobacillus*, *Bifidobacterium, Enterococcus* and *Streptococcus.* A similar product, Protexin Aquatic is used by the aquaculture industry to feed to finfish by bioencapsulating the probiotic by feeding to *Daphnia magna* which are subsequently eaten by the fish (Faramarzi et al. 2011). Unfortunately, Protexin Aquatic is unavailable in Australia.

Protexin Professional was found to contain 210 mg of TOC per gram. In order to estimate the contribution of bacterial cells to the overall organic carbon load, a 50 mgL^{-1} suspension (in Milli Q) was filtered through, 0.45, 0.2 and 0.1 μm pore size filters, with pore sizes of ≤0.2 μm being known to remove bacteria cells (Levy 2001). The findings of this exercise (see results below) resulted in a decision being made not to trial this product with *M. macleayi*, but rather to redirect efforts to another product called 'Platypus Aquaculture'.

Trials using commercially available *Bacillus* spore suspension

'Platypus Aquaculture' consists of dormant *Bacillus* spores suspended in distilled water with sodium salts and acetic acid. The spores are activated by heat shocking with boiling Milli Q water and then brewing for 4 hours in an aerated cone in the presence of a nutrient powder called 'Platypus Activate'. In the absence of any other guide for setting the feed ration, the concentration range of brew fed to *M. macleayi* during the trial was based on that recommended for use in aquaculture ponds. The brew was prepared following the manufacturer's instructions, after which a 1:4 dilution was undertaken using ultra-pure water. Six different volumes of this suspension (12, 24, 48, 96, 192 and 384 μL) were added to 30 mL NMCW in the presence of the regular unicellular algal ration to make up the different feed treatments. Each treatment consisted of ten vials, each containing an individual cladoceran. A FFV and algal control, as well as algal-only treatment, were included for comparative purposes. The trial was run until the day the cladocerans released their third brood. Cladoceran appearance, survival and neonate production were recorded daily.

Observation of the brew under ×400 magnification indicated that another larger cell type dominated the final suspension being fed to *M. macleayi*. These cells were identified as yeast. The manufacturer confirmed that yeast extract was a primary ingredient in Platypus Activate and that it was possible that some live cells were also present. In order to then establish whether *M. macleayi* were being sustained by yeast rather than *Bacillus* spp. cells, another trial was undertaken where *M. macleayi* were fed the nutrient powder, Platypus Activate, brewed following the same method but in the absence of Platypus Aquaculture. The feeding rate for this small-scale trial was based on the volume of brew that sustained the best cladoceran performance in the Platypus Aquaculture + Activate trial. An additional electronically homogenised treatment was added to determine if greater dispersion of cells made them more available to *M. macleayi*.

The viability of *Bacillus* spp. spores in the Platypus Aquaculture product was tested by heating shocking and inoculation into Nutrient Broth, a general microbiological medium. This exercise confirmed cell viability through good growth over ~72 h (Figure 1) and visual confirmation of rod and elliptical shaped cells under ×1000 magnification.

Figure 1 Platypus *Bacillus* spores cultured in nutrient broth. Note the clarity of the un-inoculated medium on the left and the cloudiness of the other two that were inoculated with spores.

A third trial was conducted where *M. macleayi* were fed Platypus Aquaculture grown in a standard microbial nutrient broth in the absence of Platypus Activate in order to determine the relative contribution of the *Bacillus* spp. cells to cladoceran health. Five mL of Platypus Aquaculture was heat shocked in 100 mL of boiling ultra-pure water and 100 μL of this suspension was added to 40 mL of nutrient broth. A 1:10 dilution of the ~50 h old culture was undertaken and a range of volumes (12, 24, 48, 96, 192 and 384 μL) trialled in the presence of the regular algal ration. Observations of *M. macleayi* health were recorded as described above.

Results and Discussion

Off-the-shelf food products

Nutraplus

All twelve *M. macleayi* died within 24 h in the 1:1000 dilution and within 48 h in the 1:2000 dilution.

Both the 1:1000 and 1:2000 Nutraplus feeding rates resulted in orange tinted waters with a high amount of large particulate matter and electrical conductivities of 295 and 564 µS cm^{-1}, respectively. Homogenisation using a hand held tissue grinder did not disperse the large particles.

Greater dilutions of Nutraplus could have been trialled but considering the amount of large particulate material and that *M. macleayi* are thought to require a bacterial component to their diet, it was decided to trial other products rather than continue further trials with this product.

Yeast

The *M. macleayi* fed on yeast appeared healthy (based on size, colour and mobility) for the first two days of the trial. By the third day, at which point the adults were having their first brood, they still appeared healthy but neonate counts were approximately 20% lower than the controls. At the time of the second brood, neonate counts were similar to control but the health of the adults was poor, with only 5/12 surviving. The following day the remaining adults had also died.

Probiotics

Moinodaphnia macleayi fed Orthoplex Multiflora appeared healthy until they had their first brood, at which point they were paler and smaller than the FFV fed culture. Only 3/12 survived and none had produced a second brood when the control treatment had all had their second brood. The following day the remaining probiotic-fed adults had also died.

The exercise involving the filtration of Protexin Professional demonstrated that the contribution of bacteria cells to the overall TOC load was negligible (i.e. 100% of the TOC passed through all filters down to 0.1 µm pore size). This initiated a conversation with the manufacturer (International Animal Health Products, IAH) who confirmed that the bulk of the product consisted of a soluble carbohydrate carrier. Given that one of the primary aims of this study was to find a food source that would have minimal effects on the bioavailability of toxicants during testing, a decision was made not to trial Protexin Professional in the *M. macleayi* culture. On the advice of the IAH microbiologist an alternative product, that consisted of suspended dormant spores of *Bacillus* spp., was trialled instead.

Trials using commercially available *Bacillus* spore suspension

Platypus *Bacillus* spores in Activate medium

Survival of *M. macleayi* was good up to a ration of 192 µL of Platypus Aquaculture + Activate (Figure 2). Neonate production was greatest at the 48 µL treatment where the average number of neonates per individual was approximately 20% lower than the control treatment. The total organic carbon content of Platypus Aquaculture + Activate at this ration was only 0.4 mg/L TOC, suggesting that the potential for this food product to ameliorate U toxicity may be lower than FFV. Neonate numbers in the 12–96 µL treatments were higher than in the algal–only treatment indicating that the Platypus brew was nutritionally beneficial. While performance of *M. macleayi* fed Platypus did not meet the acceptability criteria for neonate production in a three brood reproduction test using this species (i.e. control reproduction ≥ 30 neonates per individual), they did reasonably well overall and it was thought that further optimisation of the diet (e.g. homogenising the brew to improve cell dispersion) may further improve cladoceran health.

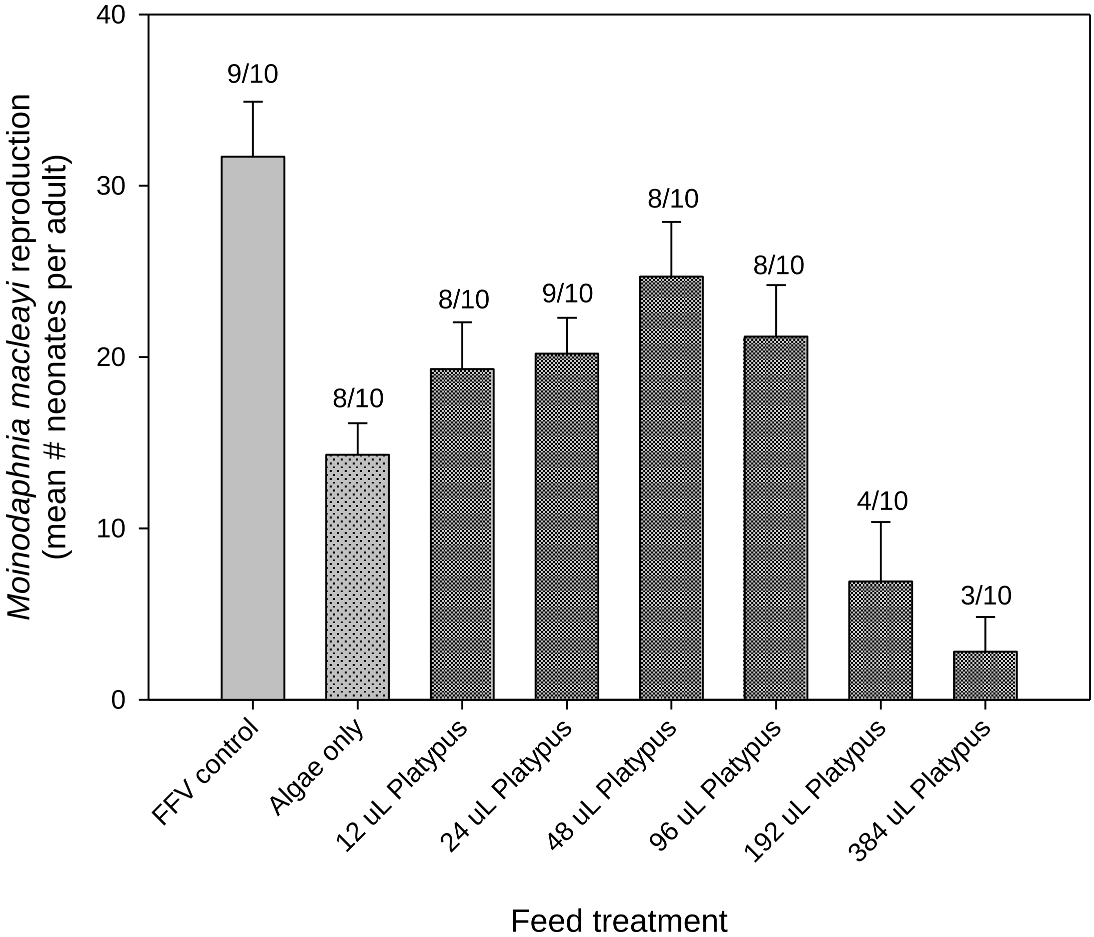

Figure 2 Response of *M. macleayi* to different food rations. The light grey bar represents the neonate production of the cladocera fed the regular food ration of FFV and unicellular algae; the dotted grey bar represents the regular algal ration in the absence of FFV; and the dark bars represent different volumes of Platypus *Bacillus* spores in activate medium provided in combination with the regular algal ration. The numbers above the bars indicate the number (out of ten) cladocera surviving until the end of the test.

Activate medium only

A Platypus Activate-only trial was conducted to determine if the yeast cells identified in the nutrient powder were being utilised by the cladocera rather than the *Bacillus* spp. cells in the Platypus Aquaculture product. None of the *M. macleayi* survived through to their third brood when only fed Platypus Activate (Figure 3). Similarly, neonate production was very low with 6 and 7 neonates per individual for the unmodified and homogenised treatments, respectively. Note that cladoceran culture health in the ecotoxicology laboratory had declined and the control performance did not meet the acceptability criteria (see section 'Current culture condition' below). As such, this experiment will be repeated, with a greater number of treatments, when the culture health becomes more reliable.

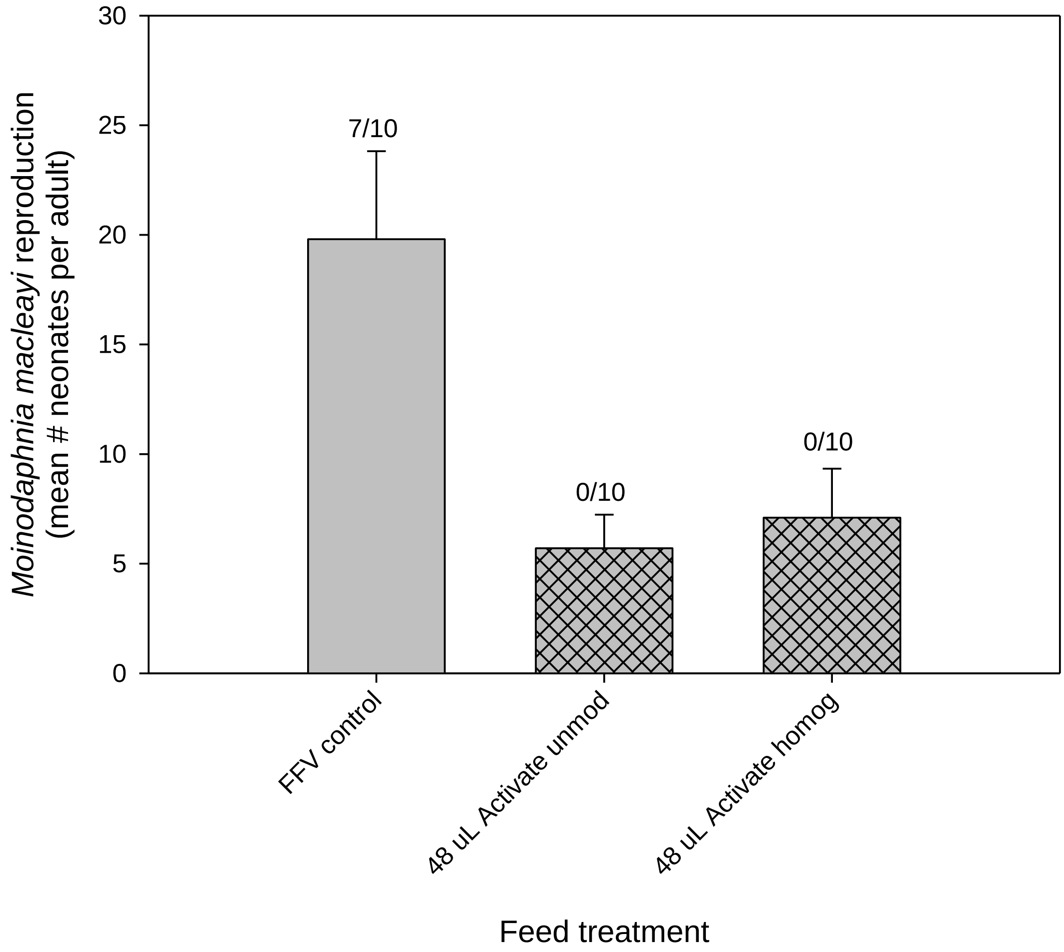

Figure 3 Response of *M. macleayi* to Platypus Activate medium only, prepared in the same manner as the activated *Bacillus* spp. spores. The light grey bar represents the neonate production of the cladocera fed the regular food ration of FFV and algae; the cross-hatched bars represent the volume of Activate medium that was equivalent to the best performing treatment of activated *Bacillus* spp. spores in the previous experiment. 'Unmod' indicates that the brew was fed to the cladocera in an unmodified form; 'homog' indicates that the Activate medium was electronically homogenised prior to feeding. The numbers above the bars indicate the number (out of ten) cladocera surviving until the end of the test.

Platypus *Bacillus* spores cultured in nutrient broth

Moinodaphnia macleayi fed Platypus Aquaculture *Bacillus* spp. grown in nutrient broth performed similarly to the FFV control (Figure 4), indicating that *Bacillus* cells may support cladoceran survival and reproduction. However, control performance did not meet the acceptability criteria for neonate production or survival so this test requires repeating. Interestingly, *M. macleayi* fed the algal-only diet out-performed all other treatments, possibly because the provision of sub-optimal FFV for an extended period had caused the cladocera to adapt to this food source alone.

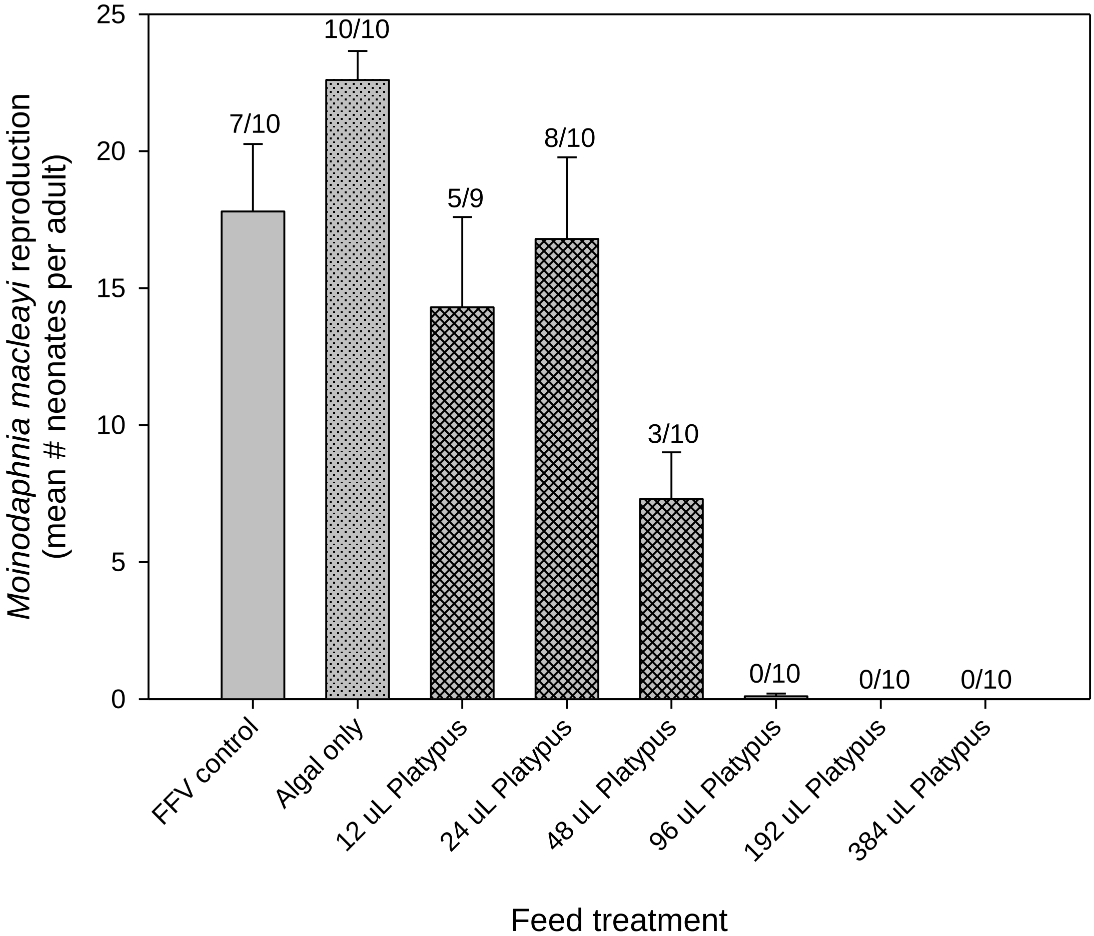

Figure 4 Response of *M. macleayi* to different food rations. The light grey bar represents the regular FFV control. The numbers above the bars indicate the number (out of nine or ten) cladocera surviving until the end of the test.

Current culture condition

The health of the ***eriss*** cladoceran culture periodically experiences short-term declines in culture health. However, since April 2014, an extended period of less than optimal neonate production and percent survival at their third brood has been experienced with this culture (Figure 5). Despite a large amount of effort to find a batch of FFV that improves cladoceran performance, a resolution to this issue has not yet been found.

Many more tests have been conducted than those reported above (e.g. 8 toxicity tests to investigate the Platypus products while only the results of three have been presented in this summary), however, their quality has not been sufficient to draw any conclusions.

A decision has recently been made to re-stock the laboratory culture with wild *M. macleayi*, as it is possible that the culture is so nutritionally deplete that it is not responding when provided with suitable conditions.

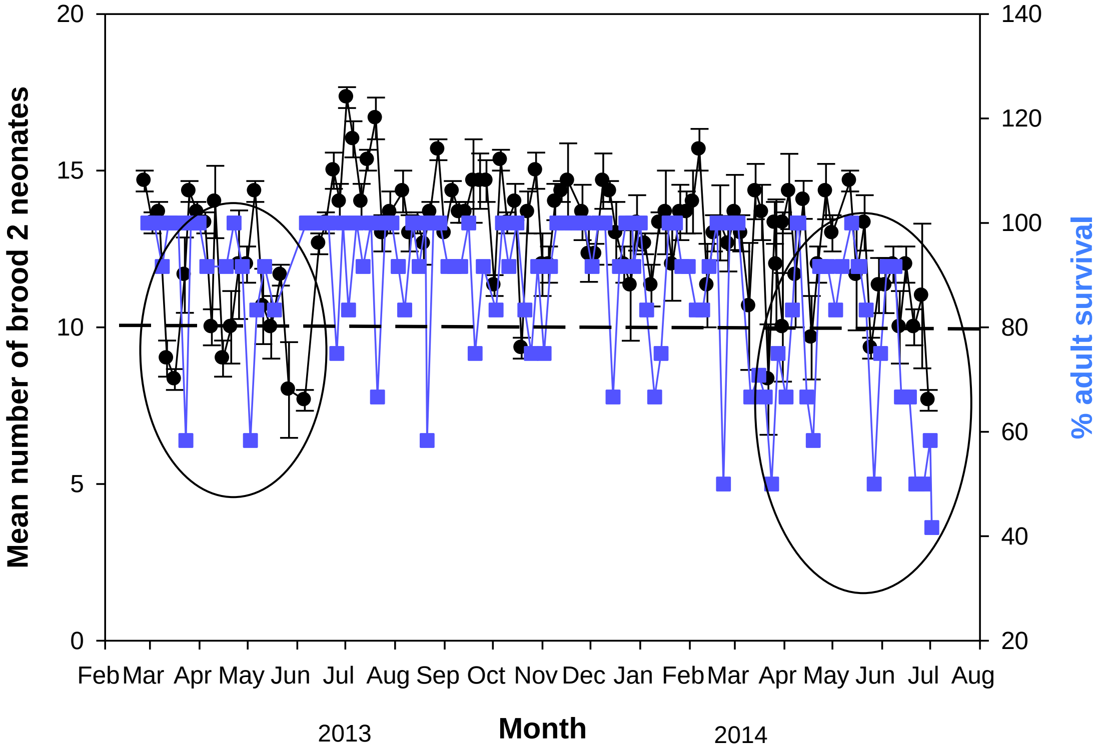

Figure 5 Neonate production and percent adult survival (after their third brood) in the ecotoxicology laboratory since February 2013. The dotted line indicates the acceptability threshold for both metrics. Periods of poor culture health are circled.

Steps for completion

Efforts in 2014–15 will be focused on continuing to restock the laboratory *M. macleayi* culture with wild animals, and return the culture to a reliably healthy condition.

Further trials will be conducted using the Platypus Aquaculture *Bacillus* spp. grown in different microbiological media (including 'Platypus Activate'). Where control treatments did not meet test acceptability criteria in previous experiments, this work will be repeated.

Depending on the success with Platypus Aquaculture, there may be a need to trial an axenically cultured bacterium.

Consideration is also being given to moving away from a standardised food source in favour of a more natural diet that may more reliably sustain good culture health. Research undertaken by Loh et al. (2009) has demonstrated the viability of fish faeces as a food source for large cultures of *Moina macrocopa* in aquaculture. However, the implications of this type of approach on the variability of U bioavailability needs to be considered.

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The toxicity of uranium to sediment biota of Magela Creek backflow billabong environments

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Background

A project that aims to derive a sediment Guideline Value (GV) for uranium (U) has been on-going since 2009. Following an initial site characterisation during the 2008–09 wet season, two pilot studies were conducted during the 2009–10 and 2010–11 wet seasons. The methods and results of the pilot studies have been previously reported in Annual Research Summaries (van Dam et al. 2010, Harford et al. 2011, Harford et al. 2012). Briefly, sediments spiked with U were deployed in an unimpacted billabong (Gulungul) for the duration of the wet season. They were retrieved and sub-sampled for the analysis of bacteria (prokaryotes), microinvertebrates (eukaryotes) and macroinvertebrates using a combination of ecogenomic and traditional taxonomic methods.

The main experiment, focusing on macroinvertebrate and microinvertebrate communities, commenced at the beginning of the 2012–13 wet season, with the experimental design revised on the basis of the findings of the two pilot studies. The results of the chemical analyses were reported in Harford et al. 2013, and showed that there was a strong positive correlation of the total recoverable fraction and porewater concentrations of U. High porewater U might have resulted in higher U bioavailability and toxicity than would be found in the environment, which could cause conservative toxicity estimates. Nonetheless, the spiking method produced a suitable gradient of sediment U concentration across the treatments with up to 90% being extractable with weak acid indicating it was likely to be bioavailable. Porewater U concentrations had also reduced markedly by the end of the deployment period.

Sub-samples of the 12 U treatments produced during the main experiment were also used in a laboratory-based midge larva (*Chironomus tepperi*) toxicity test using. This test found a median Effective Concentration (EC50) of 610 mg/kg U for inhibiting the larvae's development to an adult (i.e. an emergence endpoint). However, the observed effects may have been due to exposure of the larvae to metals in the overlying water and/or porewater because freshly spiked sediments with high porewater U concentrations were used. Hence, the result might not reflect an environmentally realistic scenario and the toxicity estimate is likely to be overly conservative (Harford et al. 2013).

Progress has been made on the analyses of the eukaryotic and macroinvertebrate datasets from the main experiment, as is reported below, along with a discussion regarding the derivation of a sediment GV for U.

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Methods

Methods for sediment spiking, deployment and retrieval have been described previously (Harford et al. 2011, Harford et al. 2012). The main experiment employed a reduced number of replicates (4) compared to the pilot studies in order to increase the number of U treatments (12), which enabled better characterisation of the concentration-response relationship. The concentration range chosen for the final experiment was: control (~8 mg kg^{-1} U), 50, 100, 200, 400, 600, 800, 1000, 1500, 2000, 3000 and 4000 mg kg^{-1} (nominal U). The passive absorption method of U spiking described in Harford et al. (2012) was used to spike 2 kg x 4 replicates of the each treatment. Deployment, retrieval and field and laboratory processing were the same as previously described (Harford et al. 2012).

Eukaryote genomic analysis was conducted by CSIRO's research facility at Lucas Heights through mass sequencing of Polymerase Chain Reaction (PCR) amplified gene sequences derived from the 18S rDNA gene. One sample in the 4000 mg kg^{-1} treatment failed to amplify and could not be included in the analyses. Both 48 pre-deployment and 48 post-deployment samples were processed for ecogenomic analysis, but the pre-deployment batch may have been compromised because the samples thawed during transit to the CSIRO Lucas Height laboratory. Hence, analyses of the post-deployment samples were the focus of further study.

A number of univariate and multivariate statistical analyses were conducted on the genomic datasets. For the ecogenomics, the Operational Taxonomic Unit (OTU) abundance data were converted to presence/absence data in each sample in order to remove any biases created by the DNA amplification and sequencing method. A Jaccard similarity matrix was calculated as a measure of the similarity amongst all samples, and metal concentrations were log-transformed for their use in multivariate analyses. Acid Extractable Metals (AEM, 1M hydrochloric acid), rather than Total Extractable Metals (TRM) were primarily used as environmental factors in the analyses because this fraction is considered to be bioavailable to organisms (Simpson & Batley 2007) and, in any case, the AEM U correlated strongly with TRM U. Primer 6 (V6.1.13, Primer-E, Plymouth) was used to produce multi-dimensional scaling plots (MDS) and Canonical Analysis of Principle Co-ordinates (CAP) analyses, which characterise the differences amongst the treatments (MDS) and identify the factors most related to differences (CAP). Permutational Multivariate Analysis of Variance (PERMANOVA) and Analysis of Similarity (ANOSIM) were used to measure separation and test for significance of separation between control and successive treatments. Threshold Indicator Taxa Analysis (TITAN) (Baker & King 2010) and Gradient Forests (Ellis et al. 2011), available as R statistical packages, were used to determine concentration-response relationships for the whole community, and were also able to determine the response of individual species. These statistical tools have been described in more detail in Harford et al. (2012).

Results of the pilot studies lead to an expectation that there would be low diversity of macroinvertebrates in the sediment, and so only samples from the three lowest (0, 50 and 100 mg/kg) and the three highest treatments (2000, 3000, 4000 mg/kg) were processed. Processing of the macroinvertebrate samples followed an internal protocol developed for these sediment samples.

Differences amongst treatment means of macroinvertebrate abundance and taxa richness were tested for, using Analysis of Variance (ANOVA). Abundance data required a Kruska-Wallis ANOVA on ranks due to a non-normal distribution in the dataset, and a

Dunnett's post-hoc test was used to determine statistical differences from the control. For the multivariate analyses, the abundance data were dispersion-weighted (to account for patchiness in samples) and log transformed. Macroinvertebrate data were analysed using PERMANOVA, ANOSIM, MDS and CAP. TITAN and Gradient Forests could not produce a result from the macroinvertebrate datasets.

Results

Microinvertebrate Eukaryotes

The number of OTUs in each treatment was consistent at ~400 OTUs across the U gradient, which indicated that the richness of eukaryotes did not respond to the spiked-U contamination (Figure 1).

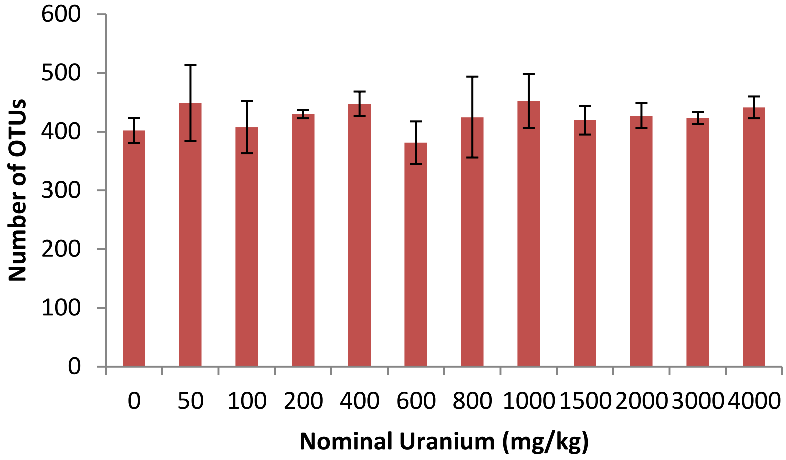

Figure 1 The number of eukaryote Operational Taxonomic Units (OTU) in each U treatment (n = 4 except for 4000 mg/kg where n = 3).

MDS ordination of all the post-deployment samples showed separation of the high (≥1500 mg kg^{-1}), medium (200 – 1000 mg kg^{-1}) and low (0 – 100 mg kg^{-1}) samples (Figure 2a). The differences between the high, medium and low treatments were more apparent when the centroids of each treatment were analysed, especially for those samples with ≥1500 mg kg^{-1} U (Figure 2b).

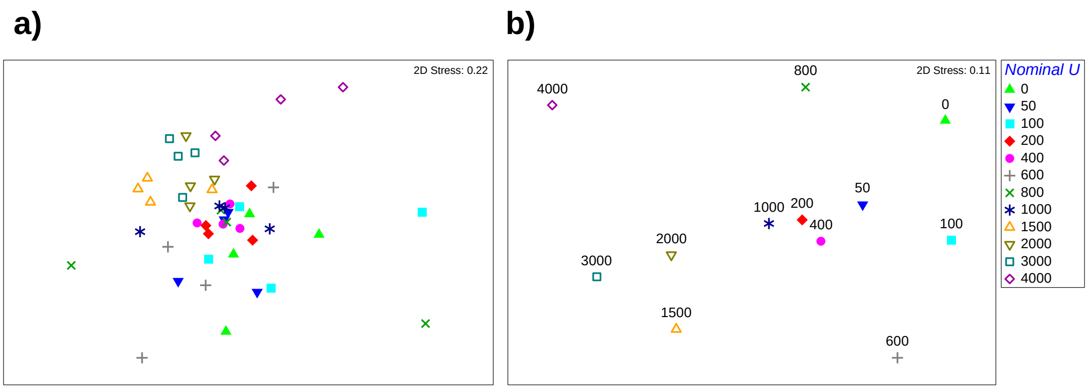

Figure 2 Multi Dimension Scaling of Jaccard resemblance for the eukaryotes. a) all post-deployment treatments and b) the centroids of the treatments.

There were statistically significant differences (PERMANOVA, $P < 0.05$, Figure 3a) in community composition for all post-deployment treatments >200 mg kg^{-1}, except for the 800 mg kg^{-1} treatment, compared to the control treatment. The 50 and 100 mg kg^{-1} treatments were not significantly different from the control treatment ($P > 0.05$, Figure 3a). There was a positive correlation between ANOSIM R values and U concentration (r^2=0.63, P = 0.0019; Figure 3b) for the post-deployment samples. A plateau in the ANOSIM R values at ~1000 mg kg^{-1} suggested that the difference from controls had reached a maximum in samples >2500 mg kg^{-1}. The 800 mg/kg treatment appeared to be an outlier and various analyses showed that it did not follow the trends of the other treatments.

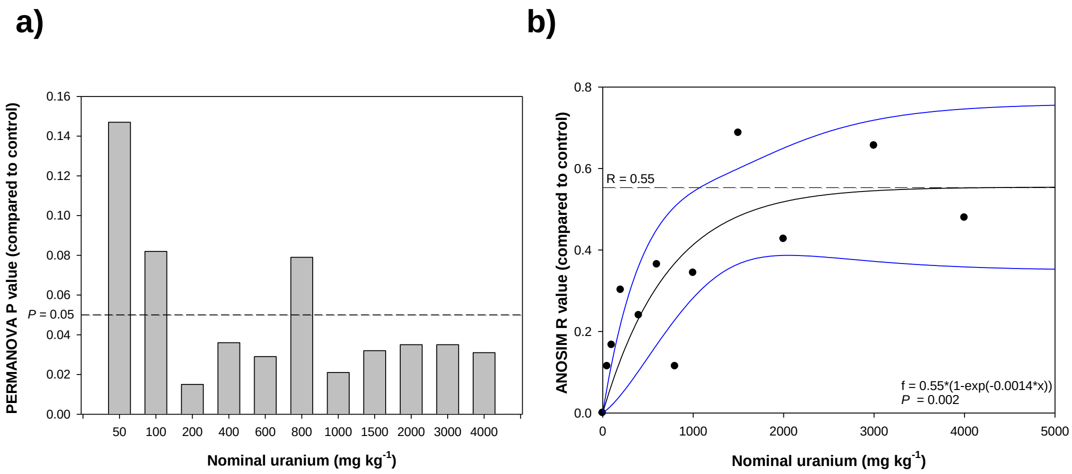

Figure 3 a) PERMANOVA pairwise comparison P values, and b) ANOSIM R value comparisons of the post-deployment eukaryote Operational Taxonomic Units (blue lines show the 95% confidence intervals).

A Distance-based Linear Model (DistLM) using AEM as the predicting factor separated the >1000 mg/kg from the other treatments and identified U as the key factor influencing the changes. Manganese was also identified as a secondary factor influencing the differences between treatments (Figure 4). This analysis also showed a clear delineation between the treatments <1000 mg kg^{-1} and those higher.

TITAN results indicated that there was a change point for species appearing (z+) at 955 mg kg^{-1} and a change point for species disappearing (z-) at 70 mg kg^{-1}. However, the initial sumz- scores appeared anomalous because the change point was taken from the low end of the confidence interval (10th percentile) and the result was sensitive to changes in algorithm parameters, which should not have altered the change points (Table 1). The reason for this will be investigated further because it may help understand the nature of the ecogenomic dataset. Regardless, a beta-version of TITAN 2.0 was recently provided by the programmers and allowed analysis with only the OTUs with high purity and reliability scores, i.e. 500 bootstraps are used to identify the Indicator Values that have a high proportion (95%) that return the correct directionality (purity) and a statistically significant result (reliability). Using OTUs that were filtered for purity and reliability resulted in a change point for species disappearing of 1000 mg kg^{-1} U, which was not sensitive to the aforementioned parameter changes. It was also similar to traditional change points of 1100 mg kg^{-1} based on TITAN analyses using traditional Bray-Curtis and Euclidian distances (Table 1). There were 26 pure and reliable indicator species disappearing across the gradient (z-) and 21 appearing (z+), of which many were unclassifiable (Figure 5). The strongest indicator species in terms of taxa disappearing appear to be from the class *Colpodea* (a ciliated protozoan bactrivore) and two OTUs from the phylum *Stramenopiles* (aka *Heterknota*, which is a major phylum of eukaryotes including all algae and diatoms, Figure 5).

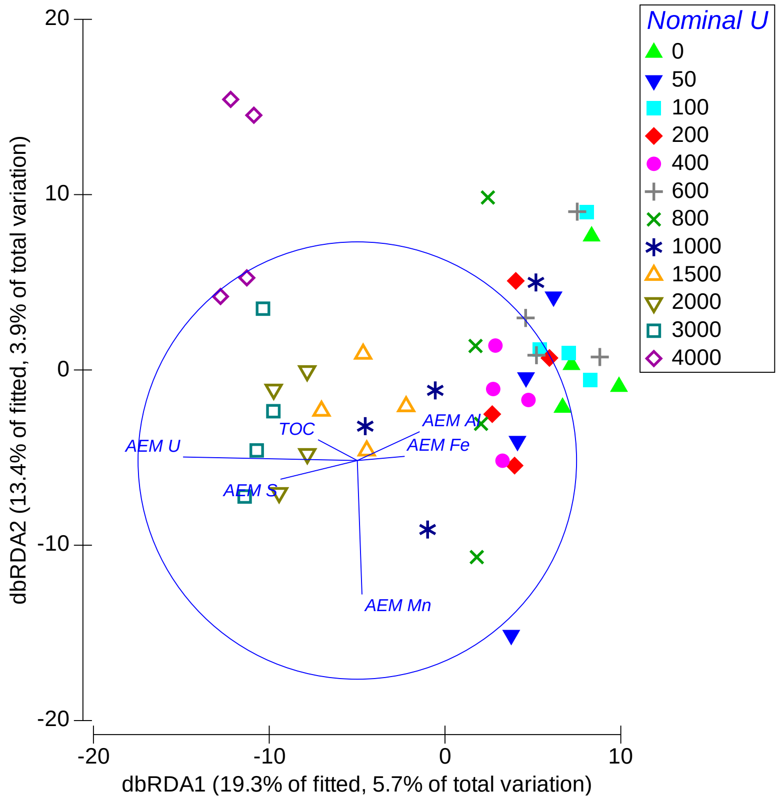

Figure 4 Distance-based linear model of the post-deployment eukaryote treatments.

Table 1 Eukaryote Threshold Indicator Taxa Analysis (TITAN) calculated change points for uranium.

Analysis	Change point (U, mg kg-1)	Confidence intervals (mg kg^{-1})				
		0.05	0.10	0.50	0.90	0.95
sumz-	70	5	70	180	650	650
sumz+	950	650	820	1110	1940	2270
Filtered sumz-	1000	420	520	1000	1180	1210
Filtered sum z+	680	650	650	960	1210	1240
ncpa.bc	1110	50	55	955	2145	2430
ncpa.euc	1110	52	54	920	2430	2640

sumz- = taxa disappearing from samples; sumz+ = taxa appearing in samples; ncpa.bc = Change point analysis using Bray-Curtis; npca.euc = Change point analysis using Euclidian distance.

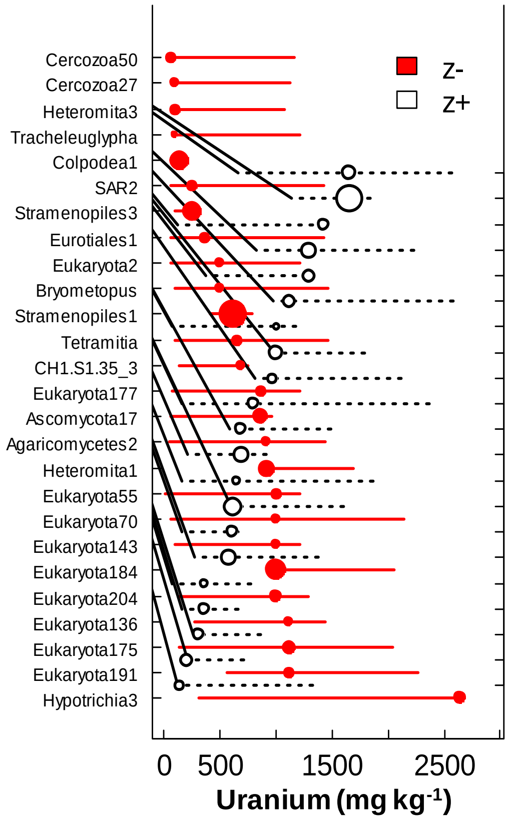

Figure 5 Reliable and pure OTUs identified from the TITAN analysis. Dots represent the OTUs change point; the larger the dot, the higher their Indicator Value score. The bars and dotted lines represent the confidence intervals calculated with the bootstrapping method.

Gradient Forest (GF) analysis calculated that U was clearly the most important predictor of changes across the gradient. Manganese was the second most important, followed by S. This concurs with the DistLM analysis. The rate of change in species composition was fairly consistent across the U gradient but there were three notable step-ups at 50, 1100 and 1600 mg kg^{-1} (Figure 6). This concurs with the unfiltered TITAN result but, unfortunately, GF has no mechanism with which to filter out OTUs that might contribute to noise (e.g. OTUs that do not respond to the U treatments) in the dataset.

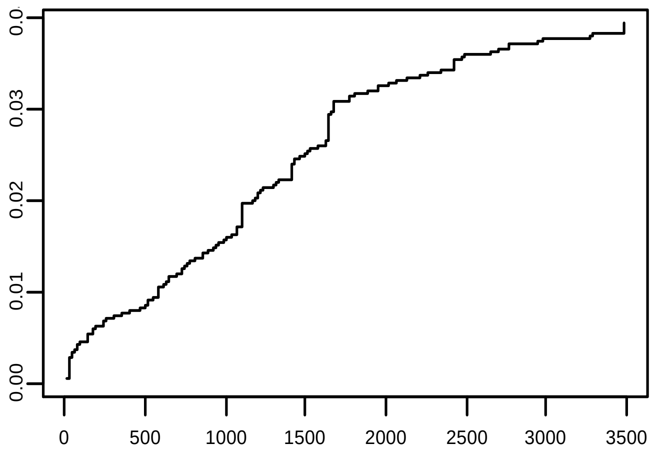

Figure 6 Species compositional turn-over across the U gradient calculated by Gradient Forest Analysis.

Macroinvertebrates

The abundance and taxa richness of macroinvertebrates was lowest in the controls and significantly higher in the 50 mg kg^{-1} U treatments (Figure 7). The abundance and richness of the other treatments were not significantly different from the controls. Analysis by PERMANOVA (Figure 8) and ANOSIM (data not shown) indicated that 2000 mg kg^{-1} U treatment was not significantly different from the control, with the 50 and 100 mg kg^{-1} being more different than the 3000 and 4000 mg kg^{-1} treatments. MDS ordination of treatment replicates showed that 3 of the 4 control replicates were markedly different from the other samples and there was no pattern of separation in the U-treated replicates (Figure 9). The results suggest that the U treatments might have encouraged colonisation, possibly through altering the physical structure of the sediments, e.g. through particle agglomeration due to higher salinity in the treatments. However, the results may also reflect anomalous variability in the data due to difficulties in processing the fine grained samples and low diversity and patchiness of benthic macroinvertebrate communities previously reported in Gulungul Billabong (Harford et al. 2012). Regardless, the very high concentrations of U did not appear to adversely affect the macroinvertebrate abundance and richness of the samples. Most of the taxa recorded in the samples are those known to be not particularly sensitive to sediment contamination, with all of the abundant taxa being identified as tolerant by Chessman (2003).

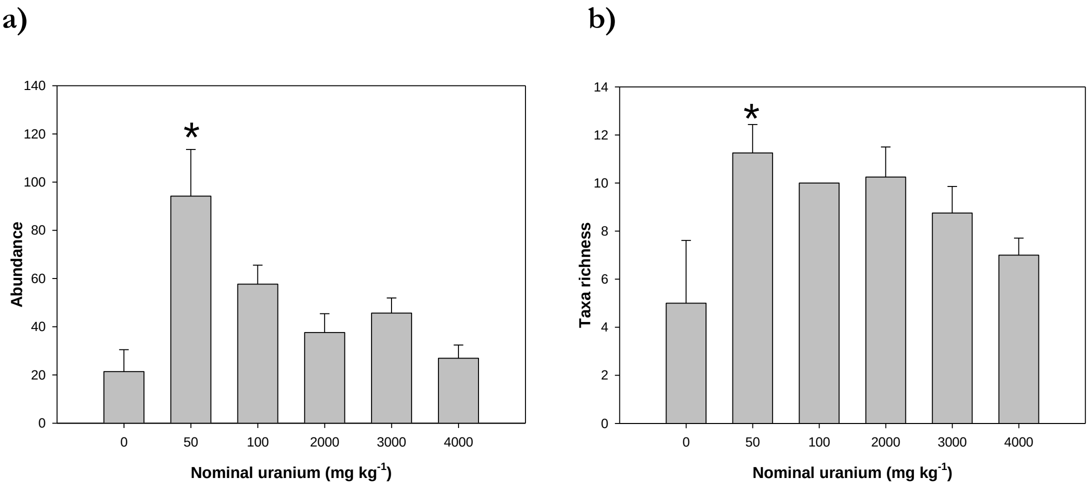

Figure 7 Effect of uranium on a) abundance and b) taxa richness of macroinvertebrates in 6 of the treatments. * represents treatments that are significantly different from the control (P < 0.05) using Dunnett's post-hoc test.

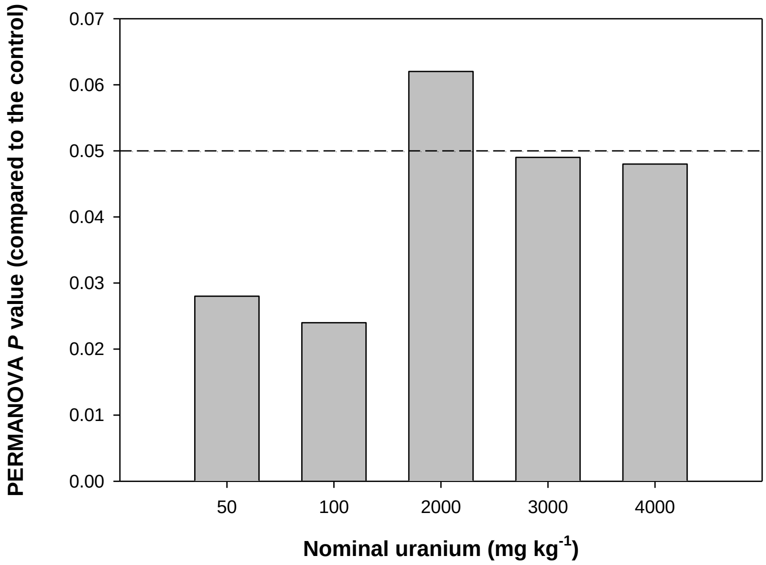

Figure 8 PERMANOVA pairwise comparison *P* values of the post-deployment macroinvertebrate species.

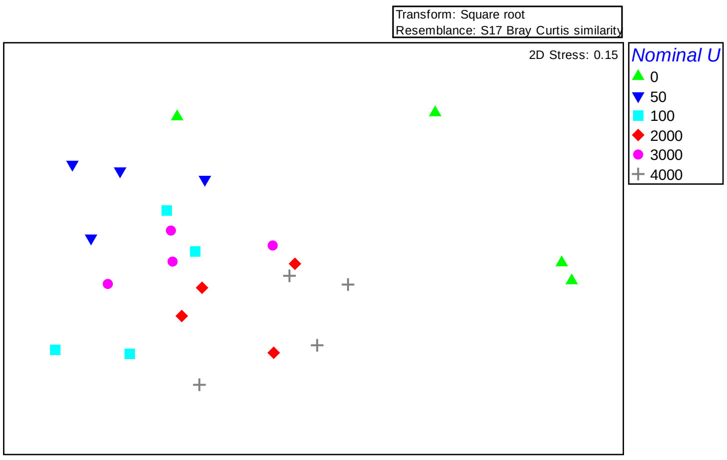

Figure 9 MDS ordination of the macroinvertebrate communities in U-spiked sediment samples

Derivation of a Sediment Quality Guideline Value for uranium

The ultimate objective of this project was to determine a sediment quality GV for U. The analyses of the macroinvertebrate data show the largest difference was between the control and 50 mg kg^{-1} U, although the effect was stimulatory. The macroinvertebrate data may be further limited due to the low diversity, tolerant fauna present and, as such, they are unlikely to be used to inform the derivation of a sediment quality GV. The various statistical analyses of the eukaryote genomic dataset showed that there were treatments with statistically significant differences from the control, and that concentration-response relationships could be observed as changes in community composition across the U gradient (Table 2). If a sediment quality GV was to be derived from TITAN then the 0.05 Confidence Interval (CI) of 420 mg kg^{-1} for species disappearing (sumz-), would represent a conservative value protecting species. However, PERMANOVA (Figure 3a), examining changes in community composition, indicated that most treatments ≥200 mg kg^{-1} U were significantly different from the control. The No Observed Effect Concentration (NOEC) of 100 mg kg^{-1} identified by

PERMANOVA would be more conservative than the TITAN value. An allied multivariate approach would be to use the ANOSIM R v U concentration relationship to derive a GV. If the plateau (shown in Figure 3b as R = 0.55) was regarded as a maximum possible change in community composition, then arbitrary, but conservative, 5% or even 10% change (i.e. R = 0.03) could be chosen as “protective concentrations”. These calculations would produce conservative GVs of between 40 and 80 mg kg^{-1} U, for a 5% and 10% change respectively. The derivation of sediment quality GVs using these types of datasets and analyses needs further investigation, as this is still an emerging area of research. The preliminary analysis indicates that that a sediment quality GV might be between 40 and 420 mg kg^{-1}.

Table 2 Potential sediment quality Guideline Values identified in the analyses of the eukaryote dataset.

Method	**Median and Confidence intervals (5, 50 and 95th mg kg^{-1} U)**	**Potential Guideline Value (mg kg^{-1} U)**	**Comment**
TITAN (sumz-)	420, 1100 and 1210	420	Filtered analysis using pure and reliable OTUs and using lower 5th percentile.
Bray-Curtis Change point analysis	50, 1100 - 2500	50	Calculated by TITAN and using lower 5th percentile
Gradient Forests	80, 1100 and 1600 a	80	80 mg kg^{-1} Highest peak appears at 1600
PERMANOVA	NC	100	100 mg kg^{-1} is similar to control, and would equate to a No Observed Effect Concentration (NOEC)
ANOSIM	20, 40 and 130	40	5% of maximum observed change
ANOSIM	10, 80 and 240	80	10% of maximum observed change

* Gradient Forests calculations do not produce confidence intervals and the numbers are the three peaks observed in the analysis.
NC = Not calculable. PERMANOVA analyses do not provide uncertainty around the values provided.

Conclusions

The analyses showed that benthic macroinvertebrate taxa typically colonising fine silt-clay sediments of backflow billabongs are only likely to be directly impacted by high U contamination. However, all the multivariate analyses indicated a compositional change of microinvertebrate eukaryotes across the U concentration range, as well as effects at lower concentrations. Numerous statistical analyses found statically significant results and thresholds of change between 40 – 420 mg kg^{-1} U. Future work will include further analysis of the datasets and publication of the results in peer-reviewed journals. The results will be used in discussions with stakeholders in order to derive a sediment quality Guideline Value (GV) for U for current operations and closure of the mine

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The effect of uranium on the structure and function of bacterial sediment communities

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Background

Microbial communities are a vital component of an ecological system, carrying out important biological processes like nutrient cycling as well as providing a food source for higher organisms. A decrease in the biodiversity and/or alteration in microbial composition may indicate a shift in the ecological health of a site (Sims et al. 2013). Therefore, many studies target microbial identification for ecological assessments. Sediment bacteria are important biological factors capable of modifying the chemistry of sediment and, hence, the speciation and toxicity of uranium (U) in sediments. While some preliminary work has been performed on understanding the interplay between U contamination and bacterial community structure (Mondani et al. 2011) in sediments, large gaps in our knowledge still exist.

Metagenomics is a relatively new molecular tool being adopted by a number of scientific disciplines. It is capable of sequencing whole genomes of individuals within a sample, allowing researchers to identify organisms by regions, such as the DNA sequence that codes for small subunit ribosomal RNA, but also to elucidate the functional potential of each organism by sequencing all the genes in a sample (Tringe & Rubin 2005). Metagenomics has the potential to provide pertinent ecotoxicological information with respect to sediment communities, as the functional diversity of the system can be assessed in addition to the biodiversity. In doing so, it enables the development of conceptual models, and generation of associated hypotheses, on the interactions and relationships within the system. In addition, through manipulative studies, biologically significant thresholds can be determined, i.e. points at which environmental contaminants (e.g. U) affect the function of an ecosystem component. This project is applying molecular tools such as Next Generation Sequencing, metagenomics and advanced bioinformatics to assess the biological and functional diversity of microbial sediment communities exposed to varying concentrations of U.

During the project, "*The toxicity of uranium to sediment biota of Magela Creek backflow billabong environments*", the analysis of bacterial community assemblages did not occur during the main experiment because of increasing questions surrounding the ecological significance of a change in bacterial community structure. Specifically, these questions related to the facts that an unknown functional redundancy was likely to exist in bacterial communities,

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and also that it was not possible to determine how a community change may affect the sediment biogeochemical function (Harford et al. 2012). Nevertheless, sub-samples of the spiked sediments were archived during the main experiment, and have been made available for use in this metagenomic analysis.

The primary hypothesis of the current study is that increasing U concentration will affect the function of bacteria in sediments and, ultimately, the biogeochemistry of the sediment. Moreover, it is hypothesised that this change in function can be detected in the metagenome of DNA extracted from U exposed sediments. The study will be undertaken by a Macquarie University PhD student (Brodie Sutcliffe), who commenced in January 2014, and will also involve collaborators from CSIRO. This investigation will contribute to our understanding of acceptable U limits, and associated Guideline Values (GVs), in billabong sediments, for Ranger operations and rehabilitation, based on meaningful data directly related to ecological processes.

Methods

Sub-samples of DNA were extracted from U-spiked sediments and archived during the eukaryote analysis for the project "*The toxicity of uranium to sediment biota of Magela Creek backflow billabong environments*".

The structure of the bacterial communities was first characterised. This involved amplifying the small subunit ribosomal RNA gene (16S rRNA) of the DNA using Polymerase Chain Reaction (PCR). The 16S rRNA amplicons were sequenced by an Illumia MiSeq sequencer (Ramacoitti, University of NSW) using both forward and reverse primers. The sequences were aligned and cleaned using the Usearch bioinformatics pipeline (Edgar 2014) and the Operational Taxonomic Units (OTUs, which are a proxy used for species identification) were compared against the Ribosomal Database Project (RDP) database to describe the taxonomy of the bacteria.

The metagenomes of the samples were determined by fragmenting the unamplified DNA to ~550 bp segments. These DNA fragments were sequenced using an Illumina platform, which produced 100 bp sequences using both forward and reverse reads. These short sequences were aligned and assembled into contigs (i.e. overlapping regions of DNA) using the Velvet algorithms (Namiki et al. 2012). The genes present in the sample contigs were identified with the Prokka pipeline (Seemann 2014).

Results

A total of 15,098 OTUs were present in the generated dataset. A Multi-Dimensional Scaling (MDS) plot of each community profile (48 samples from 12 treatments × 4 biological replicates) suggests a relationship between community profiles and U concentrations, with a notable separation of samples ≤1,000 mg kg^{-1} and those ≥1,500 mg kg^{-1} (Figure 1). PERMANOVA analysis indicated that community profiles of sediments with U concentrations ≥1,500 mg kg^{-1} were significantly different from the control ($P < 0.05$), as was the 600 mg kg^{-1} treatment (Figure 2).

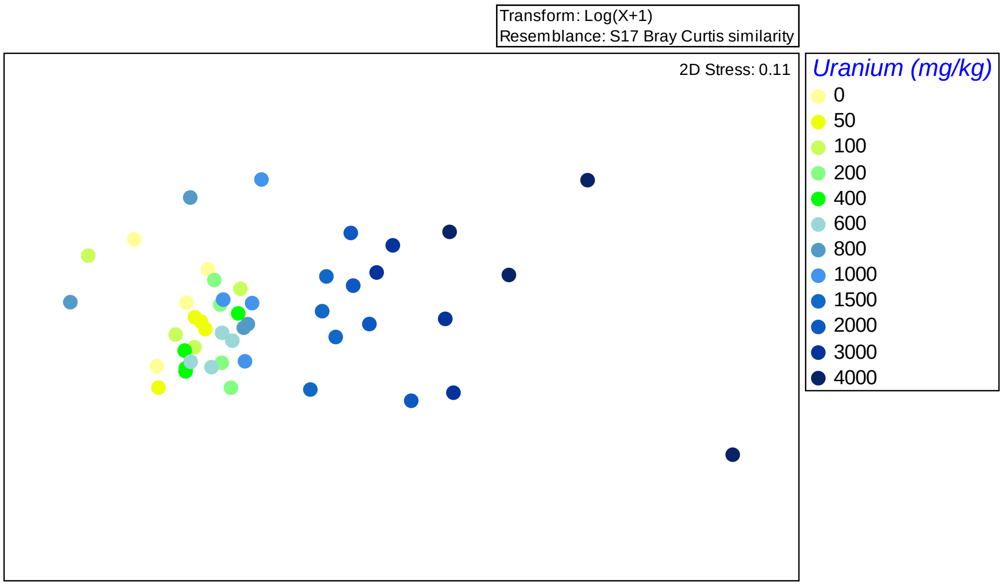

Figure 1 Multi-Dimensional Scaling of the bacteria community similarities.

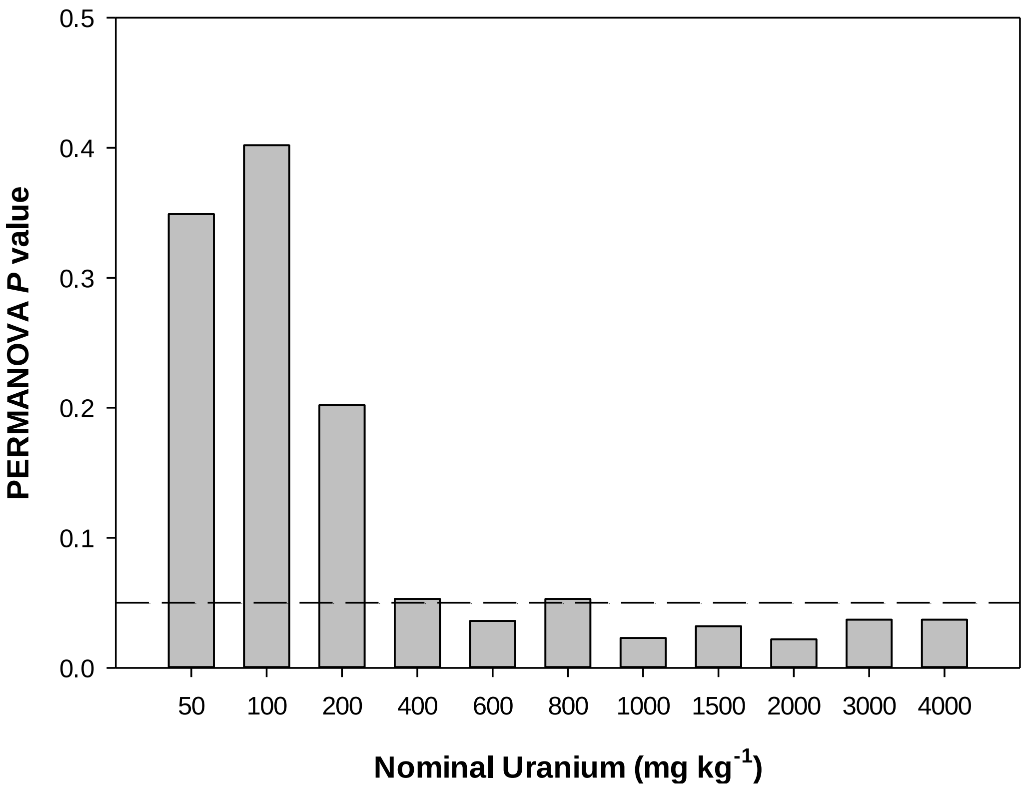

Figure 2 PERMANOVA pair-wise comparison P values against the control treatment.

Similar to the eukaryotes, an unfiltered TITAN analysis indicated a low change point for species disappearing (sumz-) of 250 mg kg^{-1} U. However, when the analysis was filtered to use only pure and reliable OTUs (i.e. Indicator Values that return a high (95%) correct directionality (purity) and a statistically significant result (reliability) during analysis) sumz- change point was 920 mg kg^{-1} U with 95% confidence limits of 380 – 1210 mg kg^{-1}.

Table 1 Change points for bacteria communities calculated by TITAN

	Change Point (mg kg-1 U)	Confidence Intervals				
		0.05	0.1	0.5	0.9	0.95
sumz-	250	10	20	90	250	730
sumz+	1110	830	1070	1470	2030	2150
Filtered sumz-	920	380	580	920	1040	1210
Filtered sumz+	1110	940	990	1110	1390	1440

Unlike eukaryote ecogenomics, it is widely accepted that prokaryote sequence read counts can be used as a proxy for OTU abundance. Analysis of OTU abundance in controls compared to the treatments shows a clear shift in the community composition with increasing U concentrations (Figure 3). It shows that as U concentration increases bacteria that are rare become more abundant and abundant OTUs become rare. Noteworthy increasing bacteria were *Holophaga*, *Geobacter* and *Geothrix*, which are known to reduce metals for their respiration. Also increasing across the U gradient were methanogenic bacteria, *Methanobacteria* and *Methanomicrobia*, and sulfate reducing bacteria, *Desulfuromondales*, *Desulfovibrionales* and *Desulfobacterales*. Noteworthy decreasing bacteria were nitrogen fixing plant associated bacteria *Rhizobiales*, and *Burkholderiacles*, and chitinolytic bacteria, *Chitinophagaceae* and *Sphingobacteriales*.

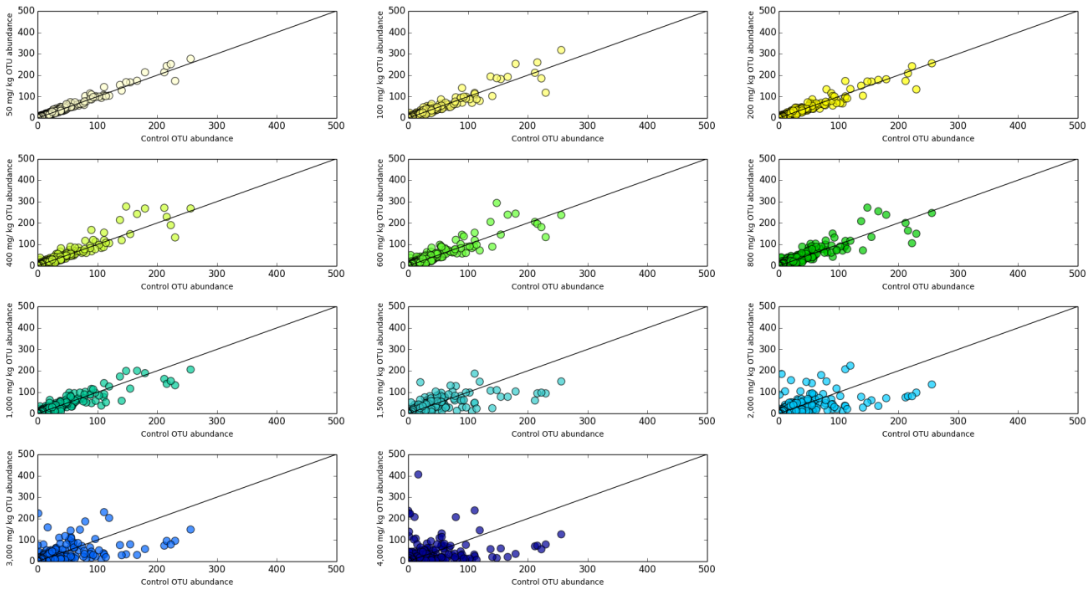

Figure 3 Correlation of OTU abundance in treatments with control OTU abundances.

The assembly of the metagenomes for each sample was completed using the velvet algorithms. Analyses of the metagenomes indicated that the quantity and quality of contigs varied markedly between samples. Namely, samples from the high U treatments produced better prokaryote contigs of higher quality and quantity compared to those from the control. This was due to the high U treatment samples being less biodiverse than the controls. This may also mean that comparison between the treatments and the controls might be unreliable due to unresolved prokaryote gene sequences that might be present in the controls. Consequently, an alternative approach of reads analysis is being perused using the EBI pipeline for metagenomes. This analysis uses direct matches with the 100 bp reads to the database, bypassing contig assembly.

Future work

In the immediate future, alternative statistical analysis will be investigated in order to appropriately interpret and integrate these complex datasets. Planning for the next phase of experiments is also currently underway. Future experiments may involve samples collected from Ranger along gradients of contamination in order to compare these with the laboratory spiked experiment. Future experiments are also likely to involve the use of Stable Isotope Probing (SIP), which involves feeding the sediment bacteria specific isotopically labelled substrates, e.g. C^{13} labelled cellulose. This will enhance the resolution of bacteria actively participating in key metabolic processes, e.g. denitrification and methanogenesis. Linking these metagenomic data with standard chemical analyses is also being considered.

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Reference toxicity testing for routine toxicity test species

C Pease, T Mooney, A Hogan, C Costello, M Trenfield & A Harford

Background

Reference toxicity testing using uranium (U) has been a routine part of the ***eriss*** Ecotoxicology programme's quality assurance system since 2004–05. The methods were developed in accordance with formal guidance on reference toxicant testing (Environment Canada 1990). The aim for 2013–14 was to continue with the established reference toxicity testing programme, using five locally sourced tropical species; *Moinodaphnia macleayi*, *Chlorella* sp., *Hydra viridissima*, *Lemna aequinoctialis* and *Mogurnda mogurnda*. The *Lemna aequinoctialis* protocol required further development due to issues with the control performance and excessive changes in water quality during the toxicity tests.

Methods

Descriptions of the toxicity testing procedures are provided in Riethmuller et al. (2003). The *L. aequinoctialis* protocol in Riethmuller et al. (2003) has since been modified. Specifically, the diluent was altered to use 1% modified Hoaglands E (Cleland & Briggs 1969) and K medium (Maeng & Khudari 1973) and growth rate calculated from total surface area change was included as an endpoint. However, it was found that this diluent required high concentrations of U to be effective, which caused concomitant reductions in pH that potentially confound the toxicity estimates. Additionally, large and variable fluctuations in pH were observed in the controls and some treatments. In order to reduce the amount of U required for reference toxicity testing and minimise changes in water quality during the toxicity tests, modifications of the *L. aequinoctialis* test medium were investigated.

Progress

In total, 23 reference toxicants tests (*Chlorella* sp. – 8; *H. viridissima* – 4; *M. macleayi* – 5; *M. mogurnda* – 4; and *L. aequinoctialis* – 2) were completed during 2013–14. Of these tests, 21 provided valid results, as summarised in Table 1. In order to generate a reference control chart, a sufficient number of properly conducted tests must be included in order to capture a representative range of variability. Environment Canada (1990) recommend the inclusion of effect concentrations (ECs) from at least 15–20 reference toxicity tests in order to calculate reliable warning limits (± 2 Standard Deviations, SD) and 99 % confidence limits (± 3 SDs). The associated control charts are presented in Figures 1–6. The Ecotoxicology laboratory aims to complete four tests per species per annum. However, this was not achieved for *L. aequinoctialis* because of the need to prioritise method development in response to the aforementioned issues.

A summary of the issues identified and method development undertaken in 2013–14 for each component of the reference toxicity test programme is provided below.

Chlorella sp. (green alga)

Of the eight *Chlorella* sp. tests performed, all were valid for this reporting period. The tests were divided between two methods; the normal method (3 x 10^4 cells mL^{-1}, normal nutrients) and a new modified method (3 x 10^3 cells mL^{-1} and reduced nutrients). For the normal method, control growth rates were within the acceptability criterion of 1.4 ± 0.3 doublings day^{-1}. The running mean EC50 for the normal method is 42 μg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 7 and 76 μg L^{-1} U, respectively (Figure 1). This year we completed the method development work to reduce the Chlorella sp. starting cell density to a more ecologically relevant level. *Chlorella* sp. density was successfully reduced to 3 x 10^3 cells mL^{-1} and the nitrate and phosphate was reduced to a 1/6th of the original nutrient content. For the modified method, control growth rates were also within the acceptability criterion of 1.4 ± 0.3 doublings day^{-1}. However, these acceptability criteria may change because growth rates may vary due to lower starting density and modified nutrient concentrations. For this new method, the running mean EC50 is 42 μg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 33 and 52 μg L^{-1} U, respectively (Figure 2). An internal report will be published for the new protocol.

Hydra viridissima (green hydra)

All four reference toxicity tests for *H. viridissima* were valid. There were no issues associated with this protocol. The running mean EC50 was 85 μg L^{-1} U with all results within the lower and upper warning limits (± 2 standard deviations) of 47 and 123 μg L^{-1} U, respectively (Figure 3).

Moinodaphnia macleayi (cladoceran)

The five reference toxicity tests performed for *M. macleayi* were valid. The running mean EC50 was 96 μg L^{-1} with all but one of the tests within the lower and upper warning limits (± 2 standard deviations) of 0 and 265 μg L^{-1} (Figure 4). Test 1369I resulted in an EC50 of >270 μg L^{-1} U which exceeded the upper warning limit of 265 μg L^{-1}.

Past tests have indicated wide variation in *M. macleayi* U sensitivity. This continued into the 2013-14 period, with two tests yielding quite sensitive results (1371I and 1407I) and two that were insensitive (1369I and 1396I). As previously reported, differences in the sensitivity of *M. macleayi* to U have been attributed to different batches of fermented food with vitamins (FFV) with varying organic components that may result in variable binding of U to these components. This year we performed a side by side *M. macleayi* reference toxicity test using a batch of FFV that previously gave a sensitive result (test 1371I) and the batch of FFV being used in the cultures at the time to test this hypothesis (1369I). There was a dramatic difference in *M. macleayi* sensitivity between the two batches of FFV (1369I EC50 = >270 μg L^{-1} and 1371I EC50 = 22 μg L^{-1} U), thus, confirming our hypothesis about FFV affecting sensitivity to U. There is currently a project underway dedicated to resolving the inconsistencies associated with *M. macleayi* diet, which might help to resolve this issue (see related paper in this report).

Mogurnda mogurnda (fish)

The four toxicity tests for *M. mogurnda* were valid, with the EC50 values within the lower and upper warning limits of 930 and 2023 µg L^{-1} U, respectively. There were no problems associated with this protocol. The current running mean EC50 is 1477 µg L^{-1} U (Figure 5).

Lemna aequinoctialis (duckweed)

The current test medium of 1% modified Hoaglands E (Cleland & Briggs 1969) and K medium (Maeng & Khudari 1973) ameliorates U toxicity and, thus, increases the effective concentration of U. Hence, the *L. aequinoctialis* method was problematic in that very high concentrations of U are required to obtain an EC50, large variations in pH were frequently observed (>1 pH unit) and there were reductions in pH associated with U concentration in the treatments. *Lemna aequinoctialis* observations made during the brine concentrator distillate toxicity project (RES-2013–013) showed suitable growth rates in the ultra-clean distillate amended with minimal major ions. Due to these problems and observations, the finalisation of the toxicity test protocol was delayed in order to explore some alternative testing media. Simple media compositions were explored to see whether acceptable growth rates for both surface area and frond count could be obtained. These consisted of ultrapure water and minimal salts selected from the composition of the Synthetic Soft Water (SSW), which is used for all other reference toxicity tests (Figure 6). All trialled media met the acceptability criteria for frond count but transparent patches were observed on the fronds in these treatments, suggesting that the plants were not at optimal health. The next steps are to investigate if nutrient concentrations and ratios can improve the overall health of the plants while minimising the amount of nutrients added. The addition of carbon dioxide to the test atmosphere will also be evaluated.

Uranium toxicity estimates are now being calculated using two different endpoints, i.e. *L. aequinoctialis* growth rates are measured based on frond number (Figure 7) as well as frond surface area (Figure 8). The sensitivities of these two different methods are similar, with the frond number method having a running mean EC50 of 11,807 µg L^{-1} U. All results are within the lower and upper warning limits of 3186 and 20,428 µg L^{-1} U, respectively (Figure 7). The surface area method current running mean EC50 is 8962 µgL^{-1}, with all results within the lower and upper warning limits of 5115 and 12,810 µg L^{-1} (Figure 6). One test this reporting period (1372L) failed due to a pH increase of >1 pH unit.

Table 1 Summary of the results from reference toxicity tests.

Species & endpoint	Test Code	EC50 (μg L^{-1})	Valid?	Comments
Chlorella sp (72-h cell division growth rate)	1363G	NC[a]	No	No nutrients – test terminated at 48 h
	1365G	58 (56, 59)	Yes	
	1384G	75 (60, 90)	Yes	Repeated due to proximity to warning limit
	1388G	46 (42, 50)	Yes	
Chlorella sp. (as above with reduced density and nutrients)	1383G	40 (35, 45)	Yes	
	1389G	41 (37, 45)	Yes	
	1404G	39 (37, 40)	Yes	
	1415G	49 (46, 55)	Yes	
Hydra viridissima (96-h population growth)	1370B	77 (71, 86)	Yes	
	1390B	98 (88, 111)	Yes	
	1408B	86 (71, 94)	Yes	
	1417B	99 (75, 162)	Yes	
Moinodaphnia macleayi (48-h immobilisation)	1369I	>270 (NC[a], NC[a])	Yes	
	1371I	22 (19, 41)	Yes	Run side by side with 1369I using a different batch of FFV
	1396I	187 (167, 209)	Yes	
	1407I	31 (21, 49)	Yes	
	1423I	26 (22, 32)	Yes	
Mogurnda mogurnda (96-h sac fry survival)	1374E	1548 (1237, 1669)	Yes	
	1386E	1416 (1324, 1519)	Yes	
	1409E	1061 (985, 1124)	Yes	
	1422E	1273 (612, 1555)	Yes	
Lemna aquinoctialis (96-h growth rate - surface area)	1372L	7360 (6959, 7804)	No	Greater than 1 pH unit drift
	1385L	12690 (11950, 13420)	Yes	

Values in parentheses represent 95 % confidence limits

a NC: Not calculable

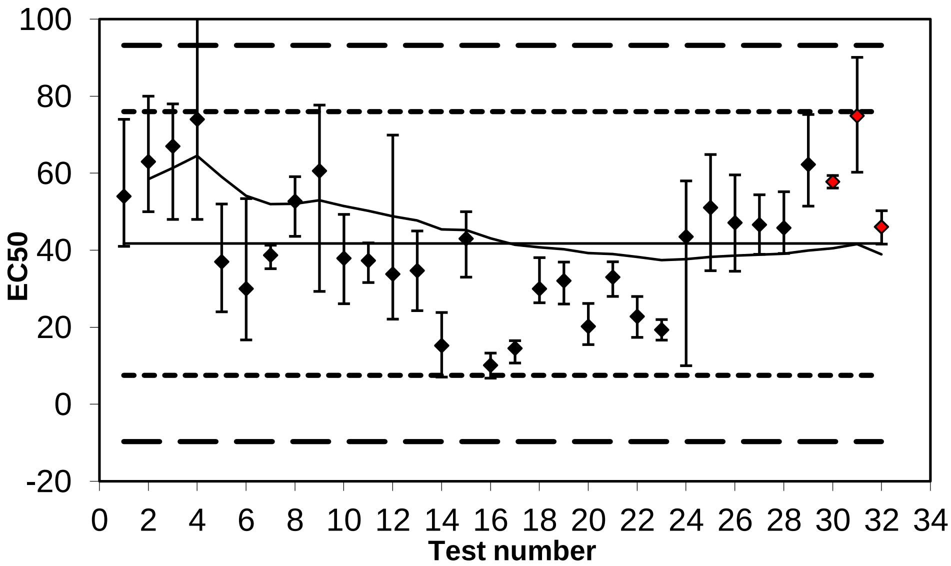

Figure 1 Reference toxicant control charts for *Chlorella* sp., as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

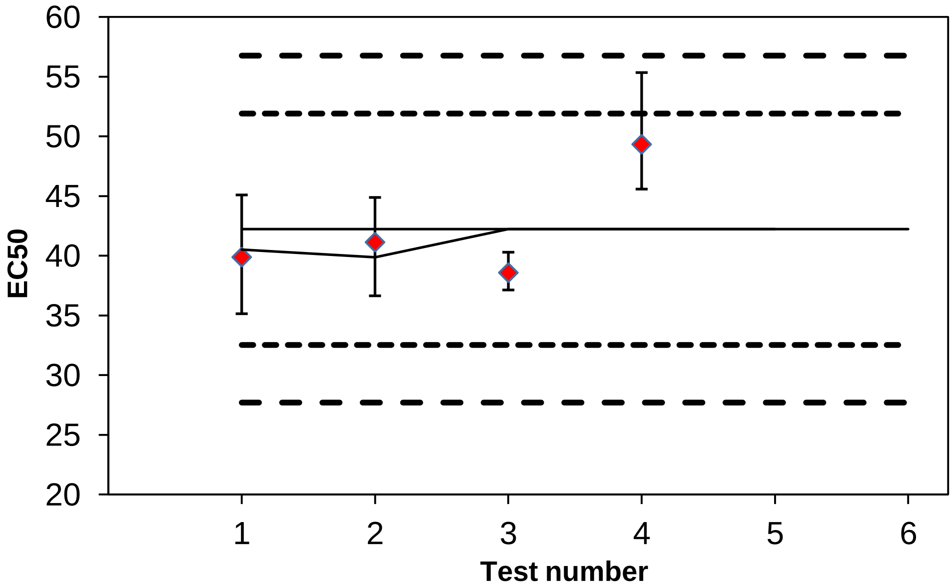

Figure 2 Reference toxicant control charts for the modified *Chlorella* sp. method, as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

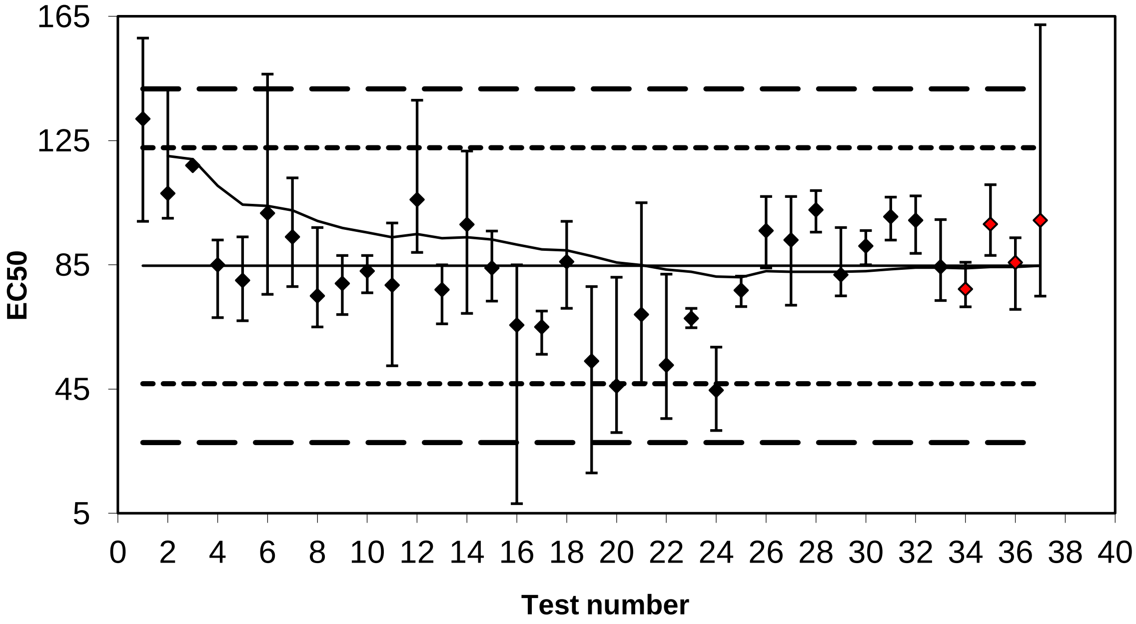

Figure 3 Reference toxicant control charts for *H. viridissima* as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

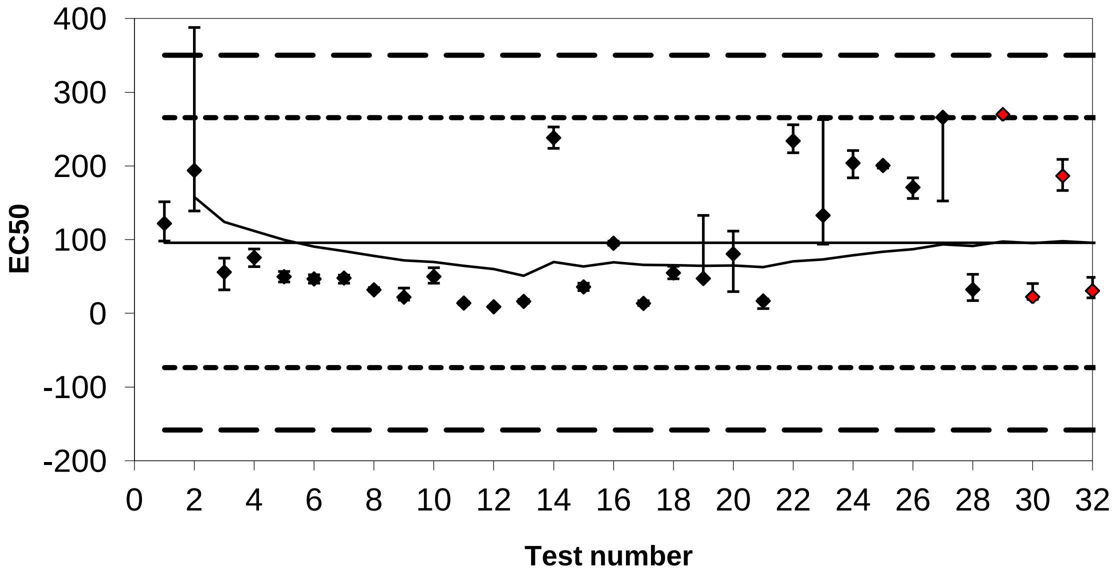

Figure 4 Reference toxicant control charts for *M. macleayi* as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

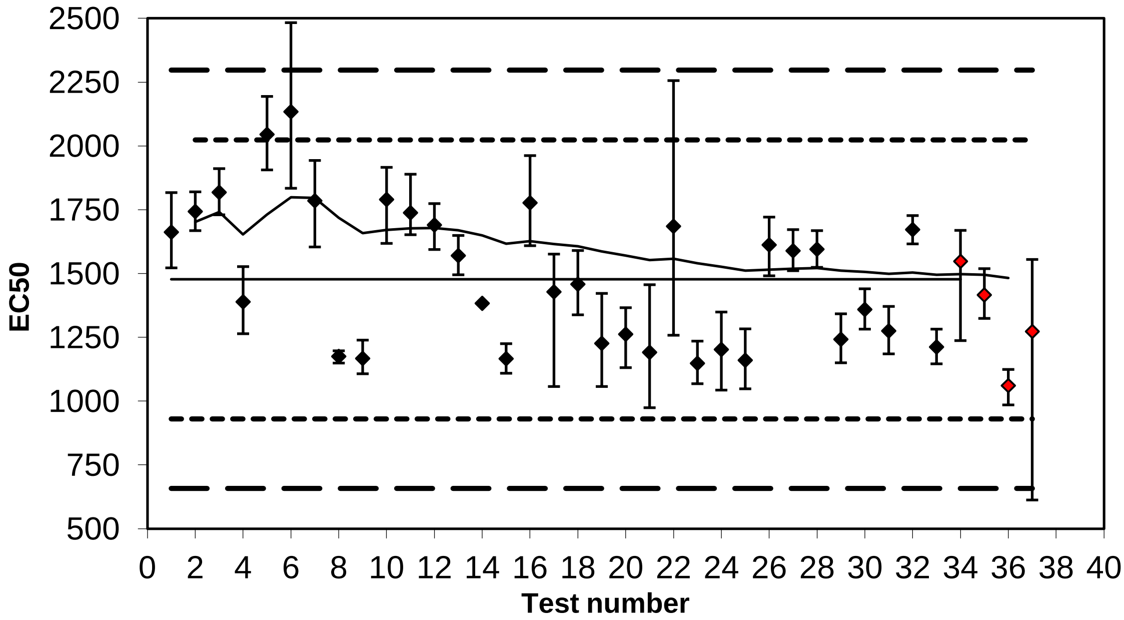

Figure 5 Reference toxicant control charts for *M. mogurnda* as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

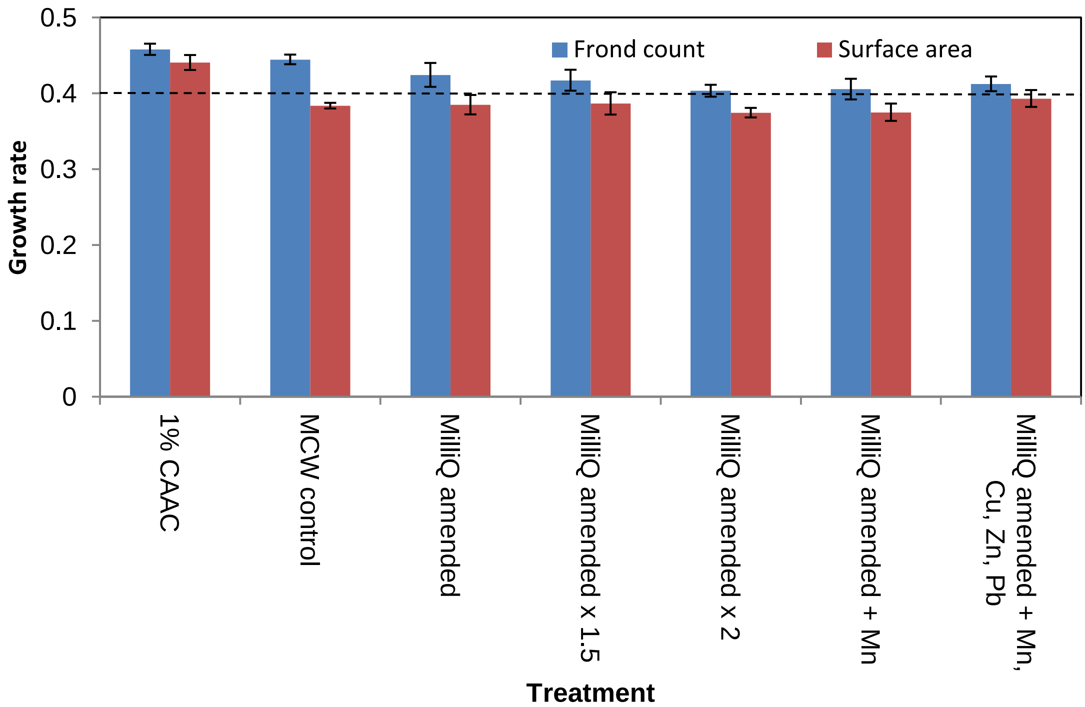

Figure 6 Trialled modified test media for *L. aequinoctialis* compared to the current media 1% CAAC and a Magela Creek water (MCW) control. 'MilliQ amended' is made up of ultra pure MilliQ™ water with sodium, potassium, calcium, magnesium, aluminium and iron salts added at the same concentrations as in SSW preparation. The red bar indicates growth rates based on surface area and the blue bar represents growth rates based on frond number (± S.E.). The broken line indicates the frond count acceptability criteria, an acceptability criteria for surface area has not been decided.

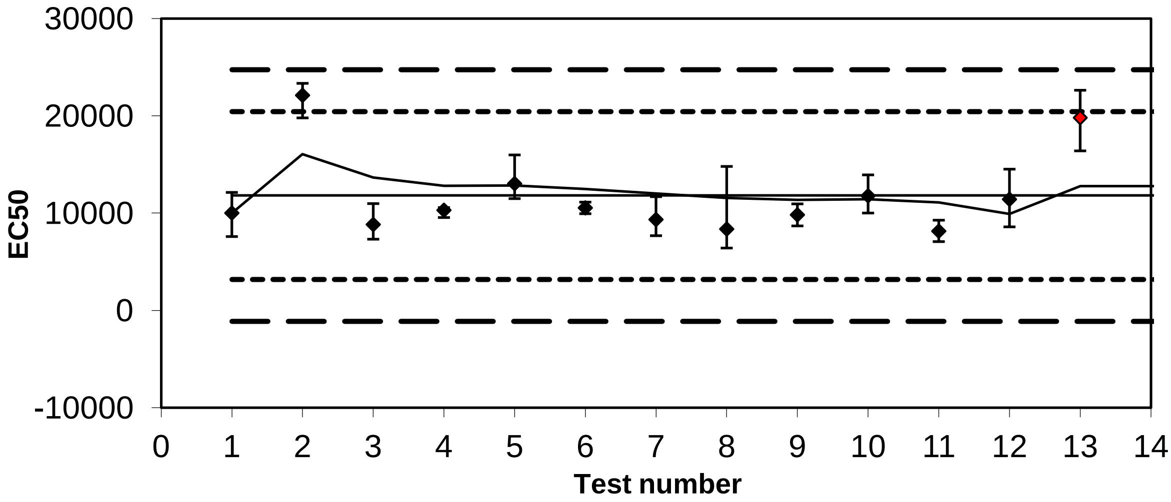

Figure 7 Reference toxicant control chart for *L. aequinoctialis* based on frond number as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

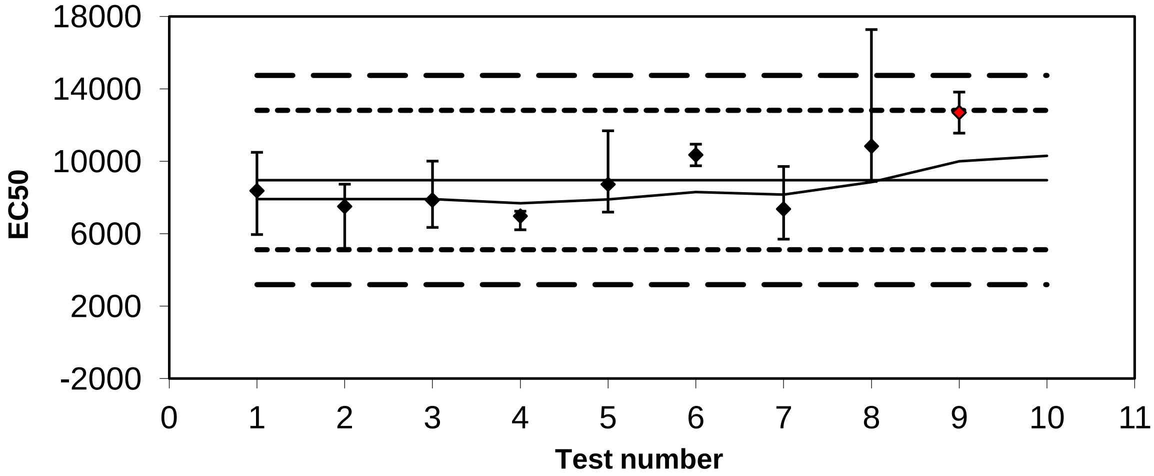

Figure 8 Reference toxicant control chart for *L. aequinoctialis* based on total surface area as of October 2014. Data points represent EC50 µg L^{-1} U toxicity estimates and their 95% confidence limits (CLs). Reference lines represent the following: broken lines – lower and upper 99% confidence limits (± 3 standard deviations) of the whole data set; dotted lines – lower and upper warning limits (± 2 standard deviations); unbroken line – running mean. Coloured data points represent tests conducted within this reporting period.

Planned testing in 2014–15

The reference toxicity testing programmes for all five species will continue in 2014–15, with the aim of completing at least four tests per species to further improve the accuracy of our reference control charts. Investigations will continue into the reduced sensitivity of *M. macleayi* to U and refining the *L. aequinoctialis* testing method.

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Key Knowledge Need 1.3: Monitoring

Results of the stream monitoring programme in Magela Creek and Gulungul Creek catchments, 2013–14

Progress under this KKN for the stream monitoring programme in the Magela Creek and Gulungul Creek catchments is reported by way of (i) results of the routine monitoring programme conducted for the 2013–14 period, and (ii) monitoring support tasks for the same period, including research and development, reviews and reporting.

Since 2001, ecotoxicity and routine water quality monitoring and programmes have been deployed by the SSD for environmental assessment of aquatic ecosystems in the ARR. The objective of this work has been to provide independent assurance that the aquatic environment remains protected from current and past mining-associated activities in the region. The monitoring programme incorporates chemical, physical and biological components.

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

Monitoring techniques adopted by the SSD that meet these requirements are:

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

- Water physico-chemistry:
    - Continuous monitoring: through the use of multi-probe data sondes and data loggers, continuous measurement of pH, electrical conductivity (EC), turbidity and temperature in Magela and Gulungul Creeks;
    - Event-based automatic sampling: The upstream and downstream monitoring sites in both Magela and Gulungul Creeks are equipped with auto-samplers, programmed to collect a 1 L water sample in response to the occurrence of pre-specified EC or turbidity conditions. The samples are analysed for total concentrations of uranium, magnesium, calcium, manganese and sulfate;
    - Ongoing quality control sampling: Routine site visits for spot *in situ* measurement of pH, EC, turbidity and temperature (fortnightly), periodic grab sampling for measurement of uranium, magnesium, calcium, manganese and sulfate (monthly) and radium (samples collected fortnightly but combined to make monthly composites).
- Two forms of biological monitoring:
    - *Toxicity monitoring* of reproduction in freshwater snails (four-day tests conducted *in situ*, at fortnightly intervals): for detection at a weekly timescale of effects arising from inputs of mine waters during the wet season.
    - *Bioaccumulation* – concentrations of chemicals (including radionuclides) in the tissues of freshwater mussels 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).

(ii) Assessment of changes in biodiversity

- This involves two types of assemblage-based biological monitoring. Data from late wet season sampling at sites in Magela and Gulungul creeks are compared with historical data and data from control sites upstream, and/or in streams unaffected by contemporary mining:
    - *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).

In accordance with the concepts of best practice and optimisation, the routine monitoring programme has evolved through time as technologies (e.g. continuous physicochemical monitoring using data sondes and telemetry) have evolved, and improved methodologies for biological assessment (e.g. in situ monitoring using snails) have been developed under the SSD research programme.

The results from the stream chemical and biological monitoring programme for 2013–14 are summarised below.

Surface water quality monitoring: Ranger

A Sinclair & K Turner

Under the Authorisation, ERA is required to monitor and report on water quality in Magela and Gulungul Creeks adjacent to Ranger mine. Specific water quality objectives must be achieved in Magela Creek.

The Authorisation specifies the sites, the frequency of sampling and the analytes to be reported. Each week during the wet season ERA reports the water quality at key sites, including Magela and Gulungul Creeks upstream and downstream of the mine, to the major stakeholders (SS, DME, NLC and GAC). A detailed interpretation of water quality across the site is provided at the end of each wet season in the ERA Ranger Annual Wet Season Report.

In addition to ERA's monitoring programme, the Supervising Scientist conducts an independent surface water quality monitoring programme that includes measurement of chemical and physical variables in Magela and Gulungul Creeks, and biological monitoring in Magela and Gulungul Creeks as well as other reference creeks and waterbodies in the region.

Key results (including time-series charts of key variables of water quality) are reported on the Internet by the Supervising Scientist throughout the wet season (see www.environment.gov.au/science/supervising-scientist/monitoring). The highlights of the monitoring results from the 2013–14 wet season are summarised below.

Chemical and physical monitoring of Magela Creek

Flow was first recorded at the Magela Creek upstream and downstream monitoring stations on 28 November 2013, which is also when the water quality sensors at the downstream site were submerged. Only the turbidity sensors at the upstream site were submerged on 28 November 2013 (Figure 1), with the EC and pH sensors, which are located above the turbidity sensors, becoming submerged on 2 December 2013 (Figures 2 and 3).

On 7 December 2013 there was a leach tank failure on the Ranger mine site. In response to this SS increased the monitoring frequency within Magela Creek as part of an investigation into the incident to confirm that there was no impact on the external environment or human health, and that the surrounding environment, including Kakadu National Park, remained protected. Results received show that concentrations of uranium, manganese and magnesium remained significantly below the ecotoxicologically derived water quality objectives.

On 22 December 2013 an isolated heavy rainfall event occurred in the Georgetown Creek catchment with 64 mm recorded by ERA at GC2 compared to only 22 mm at the Jabiru Airport. This resulted in an influx of surface water down the Georgetown system, with an EC of around 130 µS cm^{-1} at ERA's monitoring site GC2 and 94 µS cm^{-1} in Georgetown Billabong. The flow in Magela Creek was less than 5 cumecs and the pulse of surface water from Georgetown Billabong was observed at the downstream site where the EC peaked at 31 µS cm^{-1} (Figure 2). The corresponding EC measured at the upstream site during this period was 16 μS cm^{-1}. During this EC event an automatic sample was triggered at MCDW, which had elevated magnesium (1.7 mgL^{-1}) and sulfate (3.0 mgL^{-1}) concentrations. The event remained well below the chronic exposure limit for magnesium of 3.0 mg^{-1}. Uranium concentration was low at 0.19 μgL^{-1}, which is approximately 3% of the ecotoxicologically derived limit of 6 μgL^{-1} (Figure 4). Manganese concentrations were also below the ecotoxicologically derived trigger values (Figure 5)

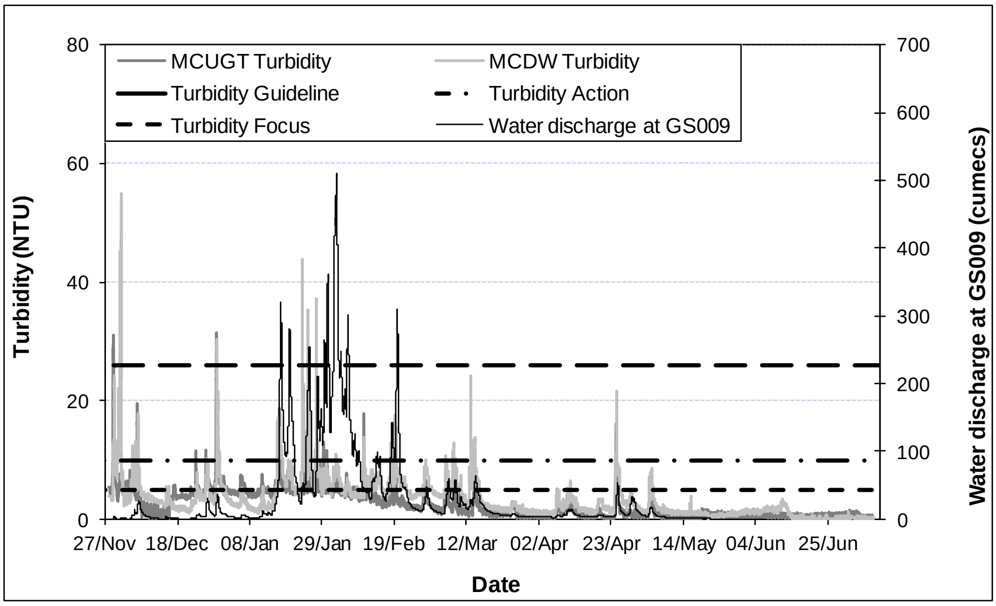

Figure 1 Continuous monitoring of turbidity and water discharge in Magela Creek during the 2013–14 wet season.

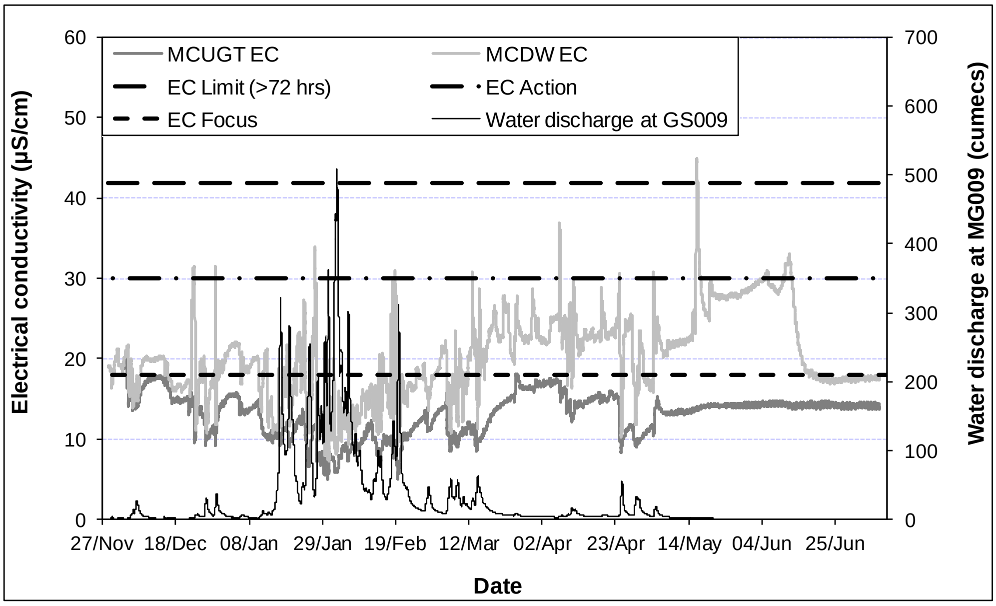

Figure 2 Continuous monitoring of EC and water discharge in Magela Creek during the 2013–14 wet season.

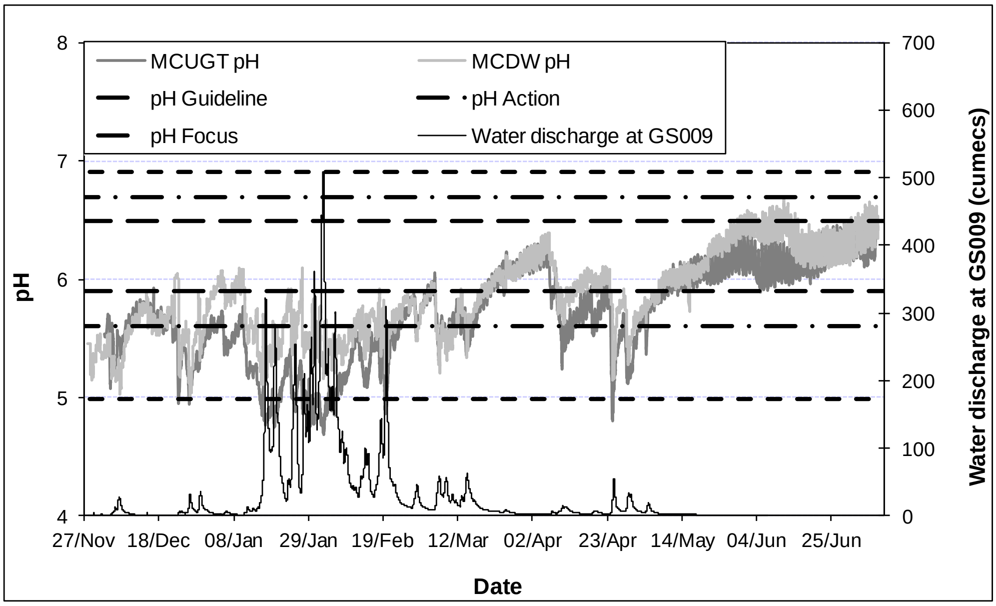

Figure 3 Continuous monitoring of pH and water discharge in Magela Creek during the 2013–14 wet season.

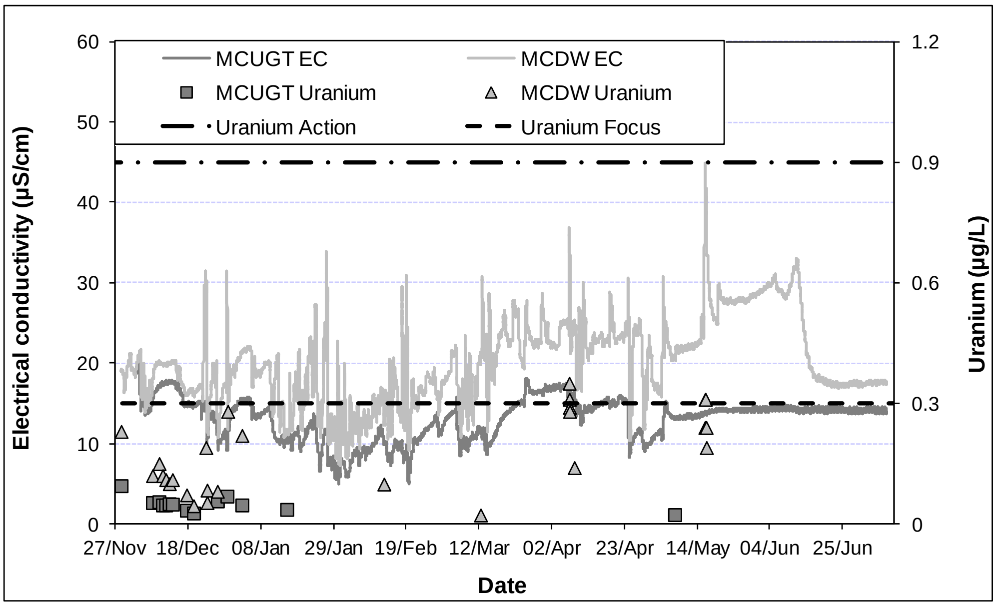

Figure 4 Continuous EC and total uranium concentrations in Magela Creek during the 2013–14 wet season.

On 29 December 2013 elevated EC was observed at the downstream site, with EC peaking at 36 µS cm^{-1} with a corresponding upstream EC of 12 µS cm^{-1} (Figure 2). This increase in EC downstream of the mine was also observed by ERA, who have continuous monitoring sites located in all three channels of Magela Creek. The fact that the EC signal occurred prior to managed release of RP1 waters and it was apparent across all three channels downstream of the mine (indicating thorough mixing), suggests that the likely source is surface water from Georgetown Billabong which at the time, had an EC of around 95 µS cm^{-1}.

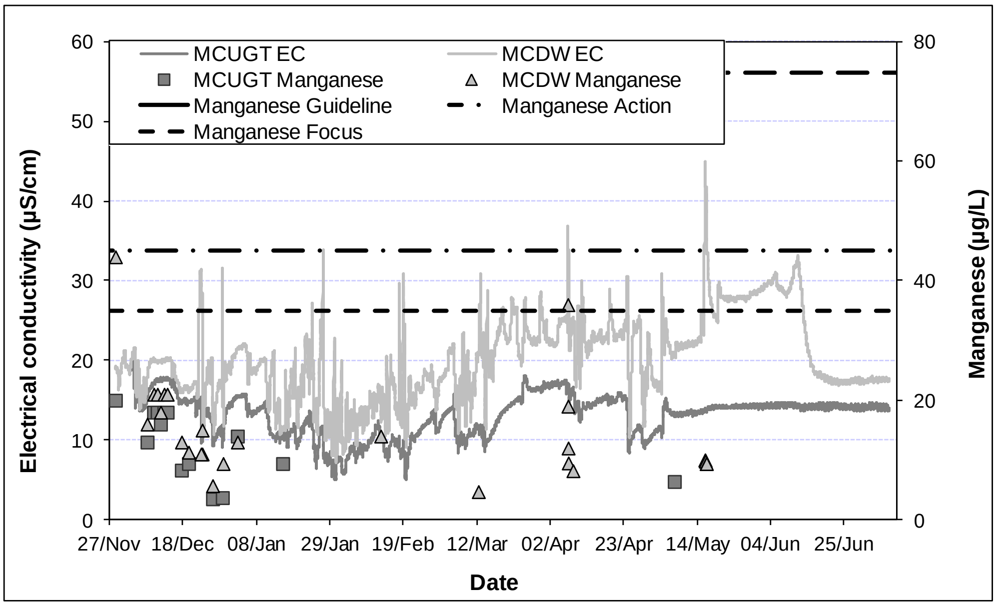

Figure 5 Continuous EC and total manganese concentrations in Magela Creek during the 2013–14 wet season.

Samples were manually triggered over the Christmas period on 29 December 2013, as shown in Figure 6. The upstream sample was triggered at 03.33 am and the downstream sample was triggered at 09.15 am. The magnesium and sulfate concentrations were higher in the downstream sample as would be expected with the higher EC at the time. The uranium concentration downstream was marginally higher at 0.28 µg L^{-1} however this is still low at approximately 5% of the ecotoxicologically derived limit.

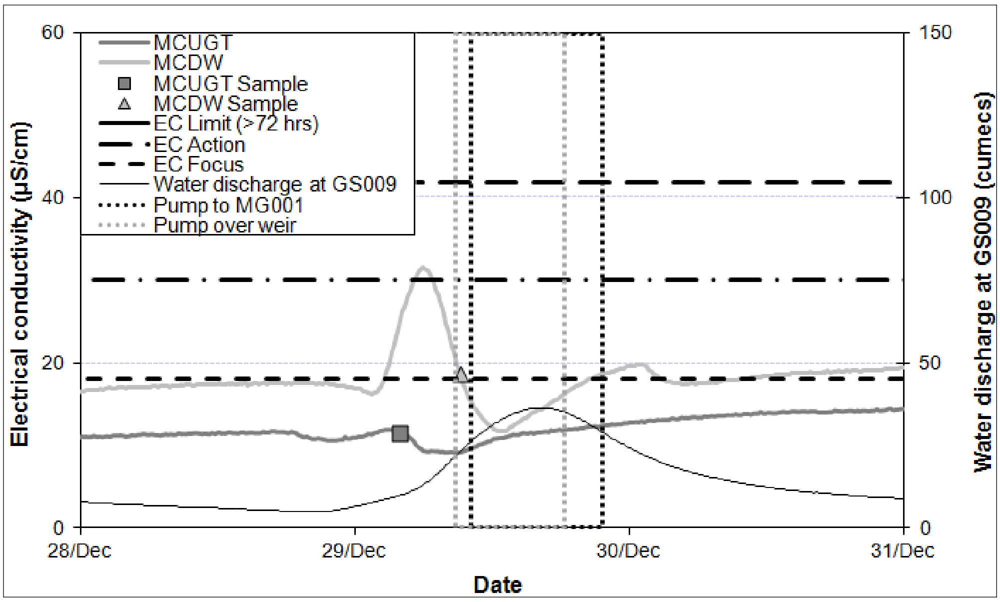

Figure 6 Magela Creek continuous monitoring data showing a peak in EC at the downstream monitoring site (MCDW) during late December 2014.

From late December 2013 the EC downstream of the mine diverged from the background EC measured upstream of the mine, which occurs each wet season once flows from Coonjimba and Georgetown billabongs into Magela Creek become established.

On 27 January 2014 there was an EC peak of 34 µS cm^{-1} at the downstream monitoring site (Figure 7). The peak occurred during a decrease in Magela Creek water level and is likely to be due to Coonjimba Billabong inflow to Magela Creek. At the time, the EC of Coonjimba Billabong was around 150 µS cm^{-1} and the EC of RP1 was around 270 µS cm^{-1} (ERA monitoring data). As the EC peaks were below the autosampler activation trigger of 42 µS cm^{-1} no samples were collected.

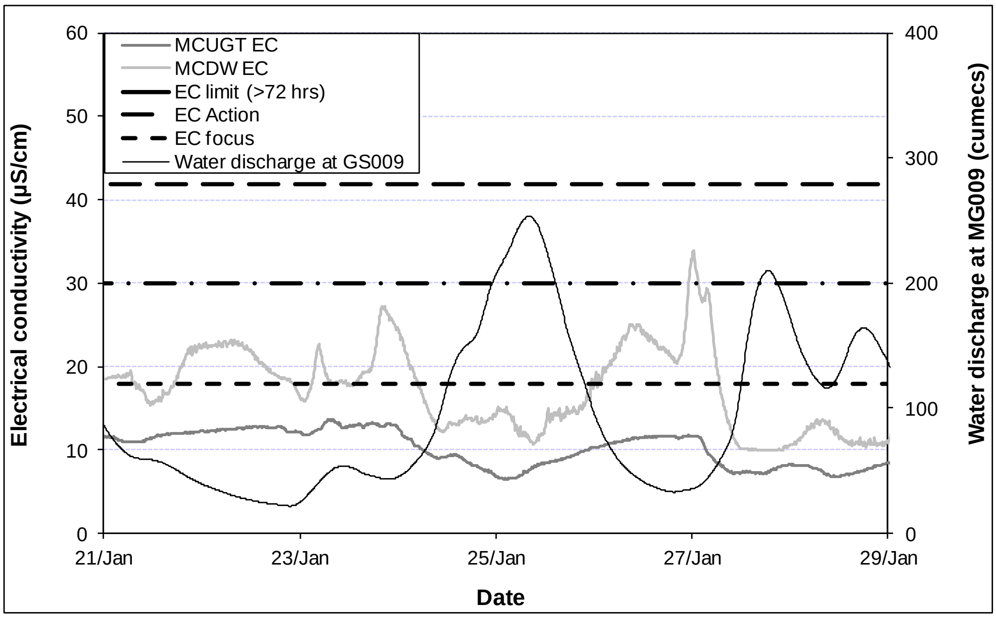

Figure 7 Magela Creek continuous monitoring data showing peaks in electrical conductivity at the downstream monitoring site (MCDW) during late January 2014.

High turbidity levels are common in Magela Creek during the early wet season due to first flush effects. Historically, the mine site does not contribute substantial amounts of suspended sediment to Magela Creek and therefore has little influence on the turbidity at the downstream site. Occasionally turbidity spikes are observed at the downstream site without an accompanying spike at the upstream site, indicating that the source of suspended sediment lies between the two sites. From 23–28 January 2014 there were three turbidity peaks which occurred at MCDW but were not detected upstream at MCUGT (Figure 8). These coincided with intermittent rainfall events associated with the monsoonal trough over northern Australia during late January and are due to surface run-off from localised rainfall. Water quality objectives are currently being developed for continuously monitored turbidity.

There was little rainfall over the first two weeks of February 2014 and the decrease in stream discharge led to increased pH levels. The inverse relationship between pH and water level in the creek is explained by the slightly acidic nature of the incident rainfall in the region (Noller et al. 1990). Water quality objectives are currently being developed for continuously monitored pH.

Uranium concentrations recorded within Magela Creek were all below the Focus trigger value of 0.3 µg L^{-1} with the exception of two samples collected on 6–7 April 2014, which contained 0.35 µg $L^{-1}$and 0.31 µg L^{-1}respectively (Figure 4). This is approximately 6% of the ecotoxicologically derived limit.

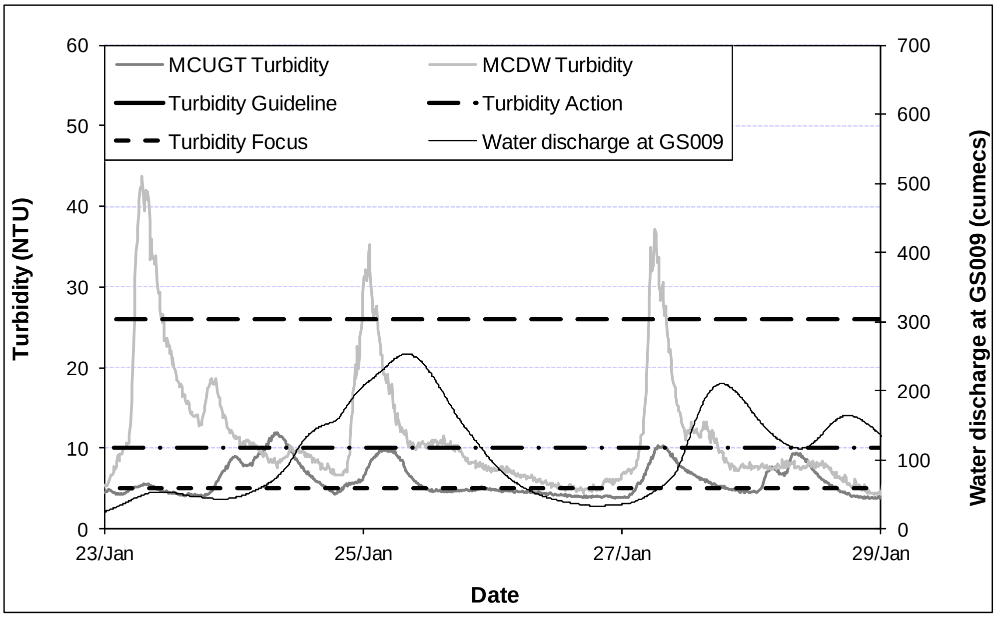

Figure 8 Magela Creek continuous monitoring data showing peaks in turbidity at the downstream monitoring site (MCDW) during late January 2014.

Manganese concentrations were also very low (Figure 5). Only two samples at MCDW were recorded above the Focus trigger value of 35 µg L^{-1} with concentrations of 44 µg L^{-1}on 28 November 2013 and 36 µg L^{-1} on 6 April 2014. The manganese trigger values only apply when creek flow is above 5 m^3 s^{-1}. At the time of sample collection on both occasions creek flow was < 3.5 m^3 s^{-1} and thus the trigger values are not relevant.

During late February and March the water level in Magela Creek had decreased leading to increased EC levels, generally fluctuating with each rainfall event. Automatic samples were triggered during two EC events in April and May 2014.

EC peaked at 37 µS cm^{-1} (above the *Action* trigger value) at 00.40 am on 7 April 2014 in response to surface water run-off from a rainfall event flushing solutes into the creek. A total of 41 mm of rainfall was recorded at Jabiru Airport. Despite the EC peak remaining below the chronic (> 72 hour) Limit four samples were collected based on the rate of rise of EC sample trigger. These samples contained magnesium concentrations above the Action trigger of 2 mg L^{-1}, with a maximum magnesium concentration of 2.4 mg L^{-1}.

A localised rainfall event occurred during the night of 15 May 2014 with 76 mm of rainfall recorded at the Jabiru Airport. This rainfall flushed higher conductivity water from Georgetown and Coonjimba Billabongs (95 µS cm^{-1} and 182 µS cm^{-1} respectively, as sampled on 12 May 2014) resulting in an increase of EC at the Magela Creek downstream site. The EC event lasted for 5 hours and peaked at 45 µS cm^{-1}. Given the EC toxicity limit for an event of 5 hours is 580 µS cm^{-1} it is unlikely this event was of environmental significance. Five samples were triggered by the EC event, two of which contained magnesium concentrations equal to or above the Chronic Limit trigger of 3 mg L^{-1} with a maximum magnesium concentration of 3.1 mg L^{-1} (Figure 9). Sulfate concentrations were also high during this EC event (Figure 10). A composite of the five EC-triggered samples was analysed for ^{226}Ra and an activity concentration of 6.11 mBq L^{-1} ^{226}Ra was recorded (Figure 14).

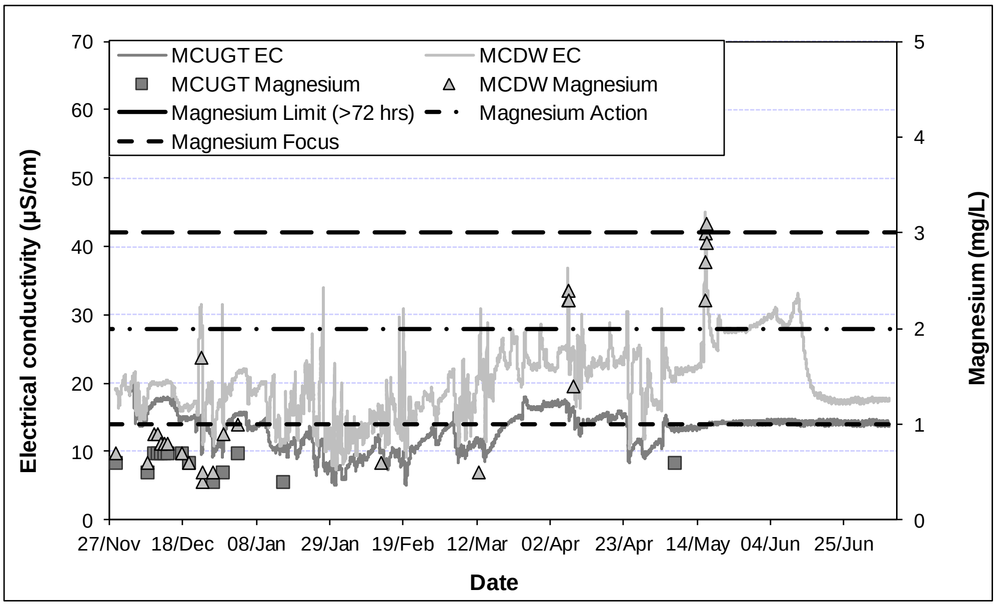

Figure 9 Continuous EC and total magnesium concentrations in Magela Creek during the 2013–14 wet season.

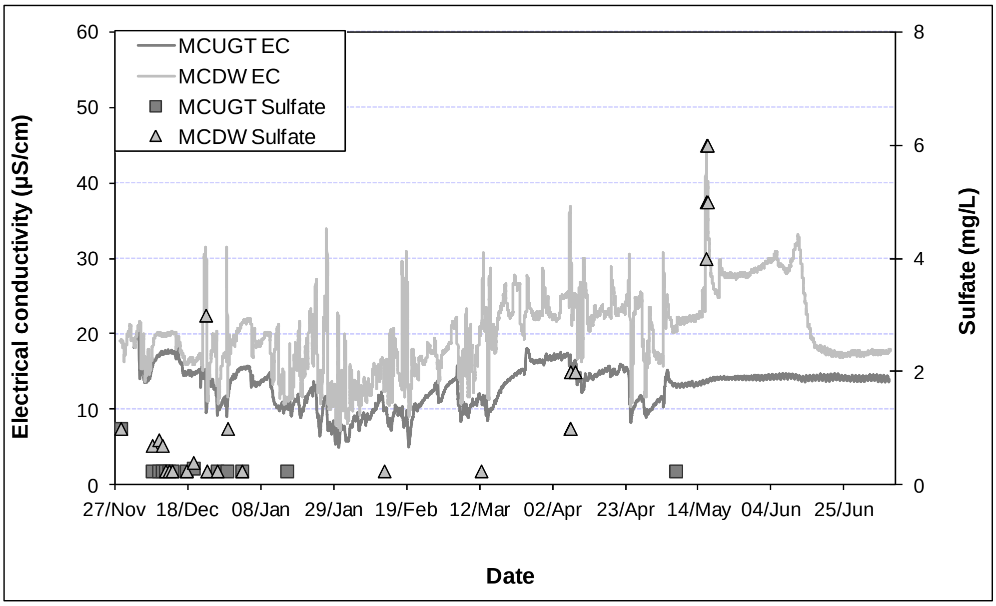

Figure 10 Continuous EC and total sulfate concentrations in Magela Creek during the 2013–14 wet season.

In December 2013 ecotoxicologically derived water quality objectives for total ammonia nitrogen (TAN) were formally adopted for Magela Creek as part of the regulatory approval for brine concentrator distillate release. Results received to date have shown that levels of TAN in Magela Creek are very low with a highest recorded concentration of 0.016 mg L^{-1} at MCDW on 12 December 2013, which is approximately 2% of the 0.7 mg L^{-1} Guideline trigger value (Figure 11).

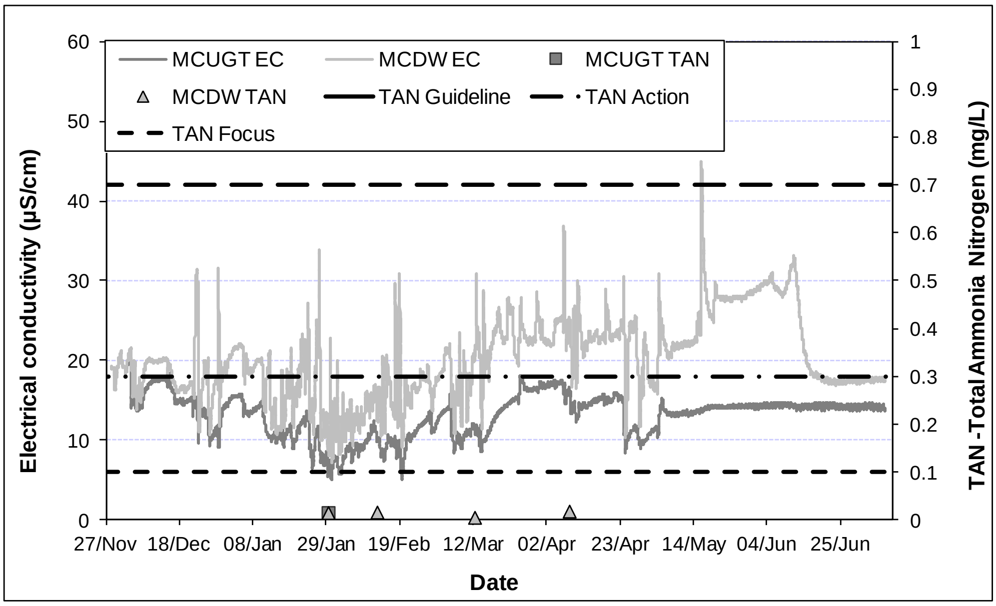

Figure 11 Continuous EC and total ammonia nitrogen concentrations in Magela Creek during the 2013-14 wet season.

Recessional flow conditions commenced in late May. Historically recessional flow has resulted in a gradual convergence of EC at MCUGT and MCDW. It was noted on 10 June 2014 that the EC at the downstream site was still diverging from the upstream site. The higher EC at MCDW and ongoing divergence with MCUGT was caused by low flow in Magela Creek coupled with water entering from Georgetown Billabong. This late flow from Georgetown Billabong resulted from WTP system permeate being released to the CCWLF. Following a query from the Supervising Scientist ERA ceased the release of WTP permeate on 10 June 2014 resulting in the almost immediate convergence of EC at MCUGT and MCDW. Continuous monitoring continued until cease to flow was agreed by stakeholders on 21 July 2014.

Overall, the water quality measured in Magela Creek for the 2013–14 wet season showed higher EC at the downstream monitoring site compared to 2012–13; however, the increase was within ranges observed in previous wet seasons. Analysis of the current and historic EC peaks at the downstream monitoring site show that the EC peaks are substantially below the Limit for EC pulses < 72 hours, which indicates that the aquatic environment in the creek has remained protected from mining activities (Figure 12 and Sinclair et al. 2013).

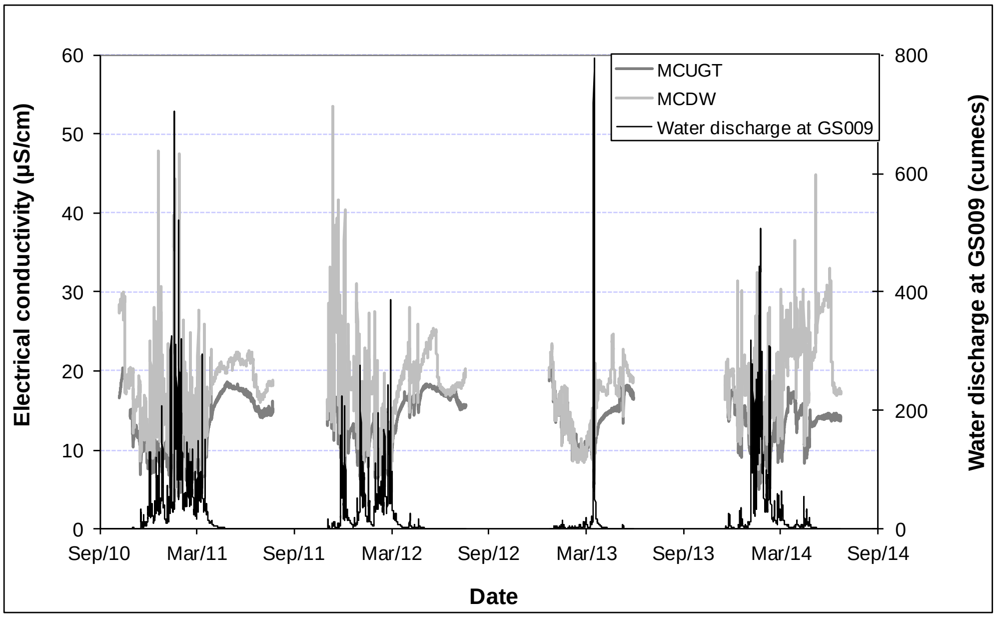

Figure 12 Continuous electrical conductivity and discharge in Magela Creek for each wet season between September 2010 and June 2014 (values averaged over a 90 minute period of measurement).

Radium in Magela Creek

Surface water samples are collected fortnightly from Magela Creek upstream and downstream of the Ranger mine. The fortnightly samples are combined to give monthly composite samples for each site. Total radium-226 (^{226}Ra) is measured in these samples and results for the 2013–14 wet season are compared with previous data ranging back to the 2001–02 wet season in Figure 13. The sample results for 2013–14 were within the historic range observed in Magela Creek since 2001.

During 7–13 December 2013 ^{226}Ra samples were collected in response to the leach tank incident and analysed individually. A routine composite sample was not obtained during this time. Consequently the average ^{226}Ra activity concentration of these samples has been included as a proxy composite sample for December 2014 and used in the calculation of the 2013–14 wet season median difference.

Since 2010, ^{226}Ra analyses on composited event-based samples (collected during EC-triggered events) have also been performed. A single composite sample, comprising five samples collected during an EC event on 15–16 May 2014, was analysed during the 2013–14 wet season. The result is shown in Figure 14, together with the results from the incident sampling and routine radium analyses. The EC-triggered event data are not included in the calculation of the wet season median difference, because these EC events are short-lived and their impact on seasonal ^{226}Ra loads is very small.

The data from monthly sample composites show that the levels of ^{226}Ra are very low in Magela Creek, both upstream and downstream of the Ranger mine. An anomalous ^{226}Ra activity concentration of 8.8 mBq L^{-1} measured in a sample collected from the control site upstream of Ranger in 2005 was probably due to a higher contribution of ^{226}Ra -rich soil or finer sediments that are present naturally in Magela Creek. This has previously been discussed in the 2004–05 Supervising Scientist's annual report.

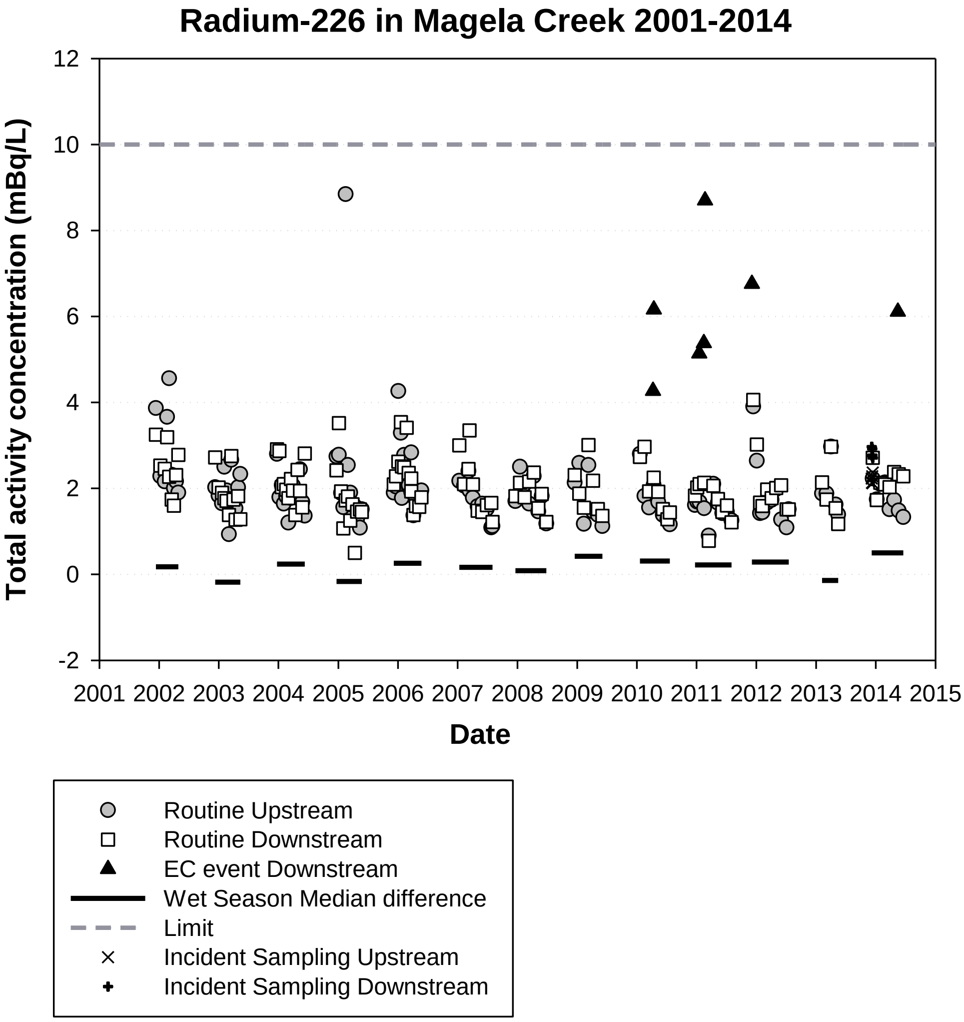

Figure 13 ^{226}Ra in Magela Creek 2001–2014.

The limit value for total ^{226}Ra activity concentrations in Magela Creek has been defined for human radiological protection purposes, and is based on the median difference between upstream and downstream ^{226}Ra activity concentrations over one entire wet season. The median of the upstream ^{226}Ra data collected over the current wet season is subtracted from the median of the downstream data. This difference value, called the wet season median difference, quantifies any increase at the downstream site, and should not exceed 10 mBq L^{-1}.

A wet season median difference of 10 mBq L^{-1} would result in a mine origin ingestion dose from ^{226}Ra bioaccumulated in mussels of about 0.3 mSv, if 2 kg of mussels were ingested by a 10 year old child. Wet season median differences (shown by the horizontal lines in Figure 14) from 2001 to 2014 are close to zero, indicating that the great majority of ^{226}Ra is coming from natural sources of ^{226}Ra located in the catchment upstream of the mine. The wet season median difference for the entire historical monitoring period (2001–14) is 0.1 mBq L^{-1}.

The wet season median difference for the 2013–14 wet season is 0.5 mBq L^{-1}, indicating a slightly higher median ^{226}Ra value for the downstream monitoring site than for the upstream monitoring site. This result is approximately 5% of the 10 mBq L^{-1} *Limit*. Whilst being the greatest wet season median difference since 2001 the result was only marginally increased compared to the range of variation observed in previous years and the individual sample results for 2013–14 were well within the historic range.

Chemical and physical monitoring of Gulungul Creek

Flow was first recorded at the Gulungul Creek upstream monitoring site (GCUS) on 28 November 2013. Flow was not recorded at the downstream monitoring site (GCDS) until 4 December 2013. The multi-probes at GCDS were only partially submerged in mid-December, which resulted in gaps in the continuous EC data (Figure 14).

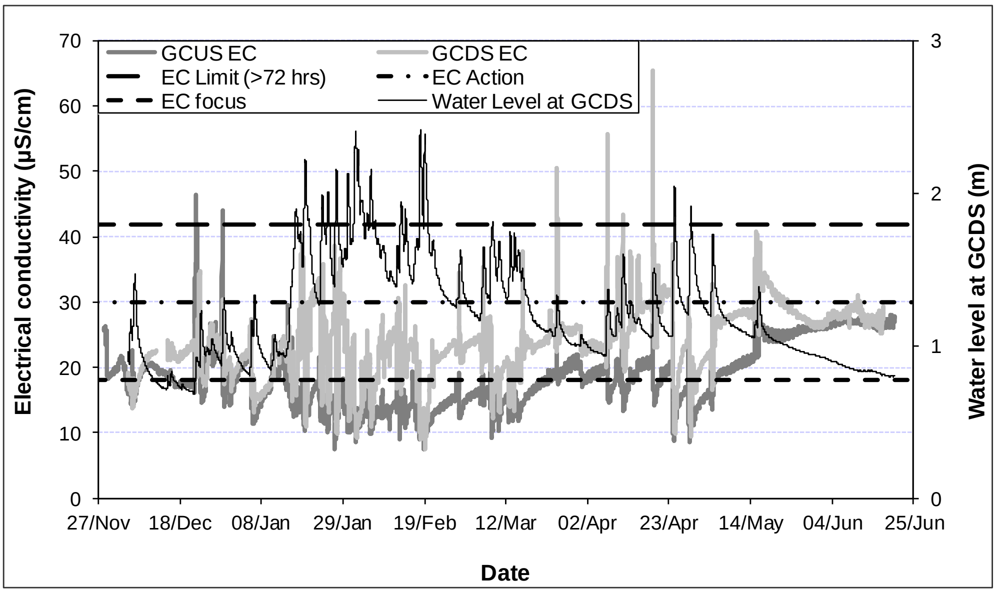

Figure 14 Continuous monitoring of EC and water level in Gulungul Creek during the 2013–14 wet season.

During 21–23 December 2013 Jabiru Airport received 22.4 mm of rain. This resulted in increased surface run-off with EC and turbidity peaks recorded at both GCUS and GCDS. This is typical of first flush effects observed early in the wet season and the increased water level submerged all water quality sensors at GCDS. The EC peaked at 46.5 µS/cm at GCUS and 34.8 µS/cm at GCDS. Turbidity fluctuated reaching 11.3 NTU at GCUS and 6.1 NTU at GCDS (Figure15). A number of event-based samples were triggered at the upstream site and the analysis results show corresponding peaks in magnesium and manganese concentrations but little change in uranium and sulphate concentrations. In distinction from the dominant mine site signature of magnesium sulphate, the major ions (anions and cations) of the upstream Gulungul catchment are dominated by magnesium hydrogen carbonate [or magnesium bicarbonate – $Mg(HCO_3)_2$] hence the lack of change in the sulphate concentrations during this event indicates a natural catchment influence rather than a mine site input.

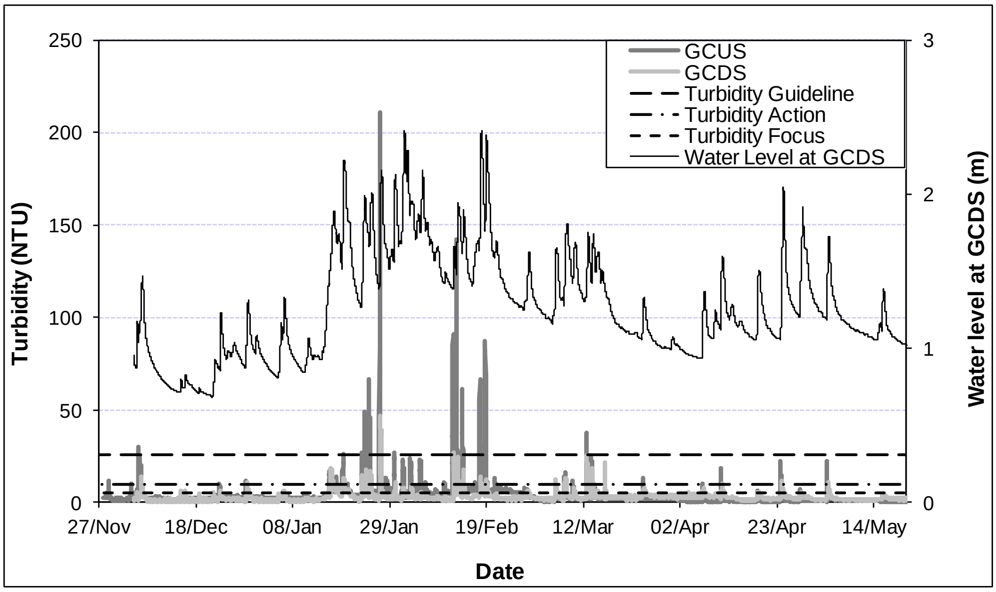

Figure 15 Continuous monitoring of turbidity and water level in Gulungul Creek during the 2013–14 wet season.

An EC event was measured at GCDS on 18 January 2014, with a peak of 41.1 µS/cm. The event-based samples collected showed a maximum uranium concentration of 0.22 µg/L, which is approximately 4% of the ecotoxicologically derived limit. Further EC peaks > 30 µS/cm were observed at GCDS from 20–28 January 2014 (Figure 17) and 12-14 February 2014 (Figure 18). On 19 January 2014 the ERA continuous monitoring at Gulungul Creek left bank (GCLB) recorded a 3 hour EC event that peaked at 184.2 µS/cm (Figure 16). The ecotoxicologically-derived framework for EC pulses provides an EC limit of 1040 µS/cm for a 3 hour EC event, therefore the environment is unlikely to have been impacted.

The SS continuous monitoring station GCDS, which is located approximately 1 km downstream of the ERA station, GCLB, typically detects all EC events that are detected at GCLB (Figure 16). However the peak of 184.2 µS/cm observed at GCLB on 19 January 2014 was not detected at GCDS, which recorded an EC of <25 µS/cm throughout the duration of the event. This indicates poor mixing of Gulungul Creek waters, as SS's monitoring station is located on the western bank of the creek and GCLB is located on the eastern bank. It also indicates that the source of the high EC waters is from the mine side of the creek and after entering the creek, the water may have flowed downstream close to the eastern bank and thus was not detected at GCDS. The poor mixing in the creek may have been exacerbated by incoming flows from the left bank tributary located upstream of GCDS (Figure 16).

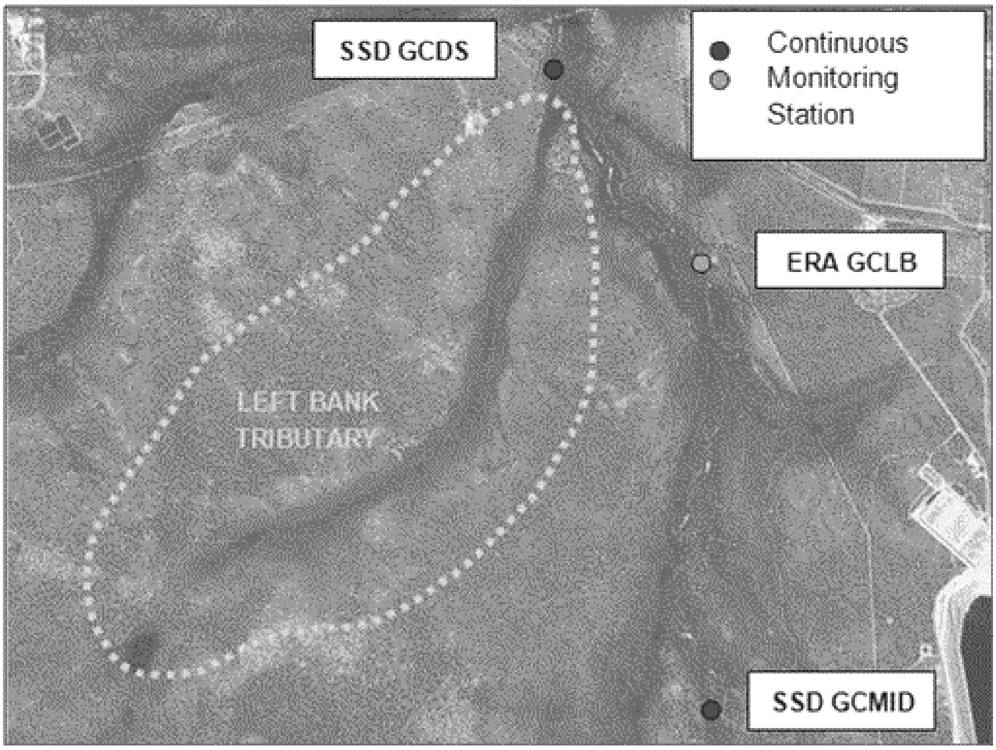

Figure 16 Gulungul Creek continuous monitoring stations.

SS is undertaking a number of investigations to determine the source of the solutes causing the EC peaks at the downstream sites and to better understand the extent and nature of the cross channel variation in EC at GCDS. To inform these investigations a number of small data loggers, measuring conductivity, temperature and depth (CTDs), were deployed at key sites in the Gulungul catchment during the 2013–14 wet season.

The first CTD logger was installed on the eastern bank, opposite the GCDS monitoring station (GCDS East Bank) on 23 January 2014. The recorded data showed that Gulungul Creek water is generally well mixed, with similar EC measured at both banks. However, occasionally the east bank EC was up to 15 µS/cm greater than the west bank EC (Figures 17 and 18). This indicates that the source of solutes is likely to be from the eastern bank of the creek, or the mine side of the creek.

On Monday 3 March 2014, SS staff conducted a field visit to the Gulungul Creek area to the west of the Ranger TSF in order to investigate the source of recent EC events noted at the ERA downstream site.

EC readings were taken at a number of locations in the standing water along the Radon Springs track, with a range of 15-25 µS/cm observed. At location 1 in Figure 1.19, being the intersection point of the Gulungul Creek Tributary Site 2 (GCT2) drainage line and the Radon Springs track, EC in the range of 1250-1300 µS/cm was recorded in the small flowing stream. This stream was tracked to its confluence with Gulungul Creek (location 2, Figure 19) and an EC reading of approximately 500 µS/cm was obtained at that location. The low volume of in-flow from the high EC tributary was quickly diluted once it entered the main Gulungul Creek channel. SS collected water samples and deployed a CTD logger at locations 1 and 2.

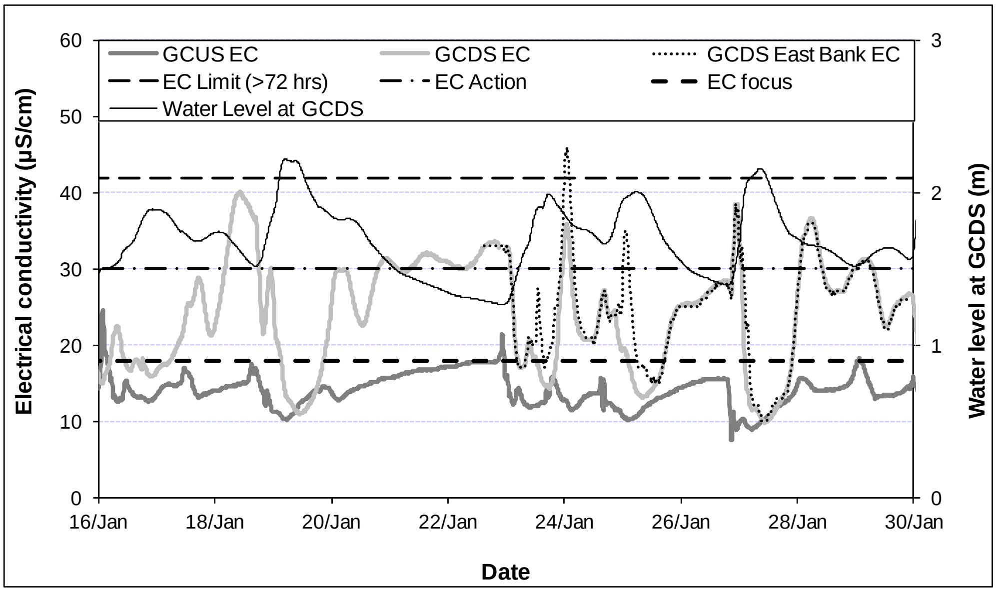

Figure 17 Gulungul Creek continuous monitoring data showing peaks in electrical conductivity at GCDS and GCDS East Bank during mid-late January 2014.

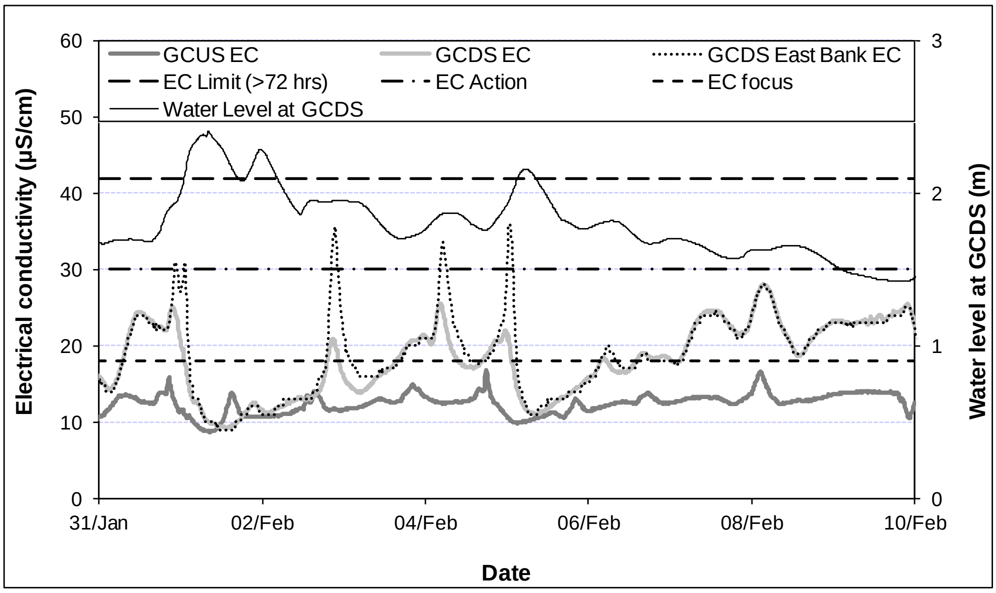

Figure 18 Gulungul Creek continuous monitoring data showing peaks in electrical conductivity at GCDS and GCDS East Bank during early February 2014.

The data collected during these investigations are currently being analysed and will be reported in the near future. To ensure that the surface water chemistry monitoring programme carried out by the SS detects and samples all EC events occurring downstream of the mine, a new continuous monitoring station is going to be installed on the eastern bank of the creek, just downstream of GCLB. This station will be equipped with duplicate EC sensors and an autosampler.

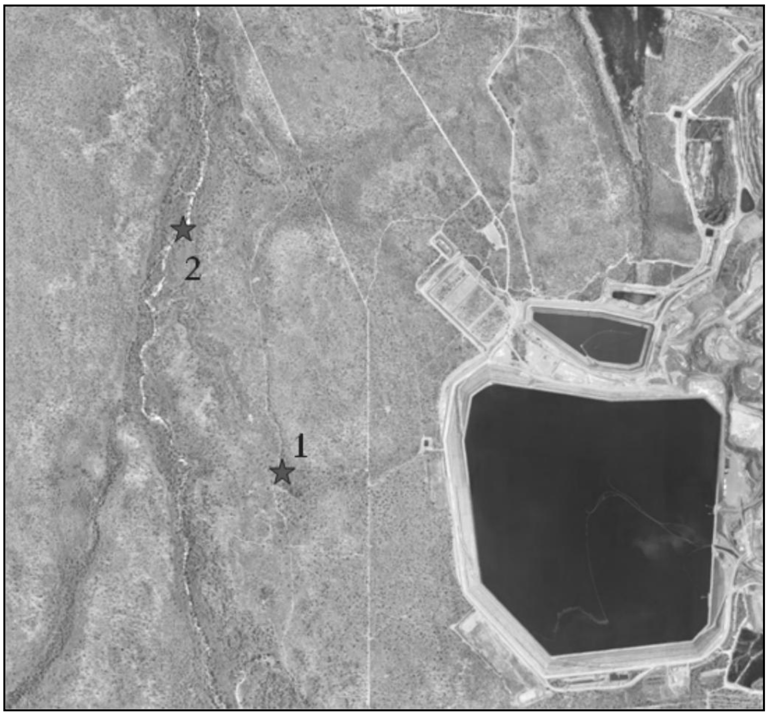

Figure 19 Location of SS EC loggers installed on 3 March 2014 to investigate the water quality of GCT2.

On 26 January 2014 a large turbidity event occurred at the upstream monitoring site GCUS in response to a rainfall event, with 68 mm recorded at GCUS, and subsequent rising creek flow. The magnitude of the turbidity peak at GCUS was 211 NTU occurring at 20:40 hours (Figure 20). This is the sixth time that a turbidity peak >150 NTU has been recorded at GCUS since continuous monitoring of turbidity began in the 2003–04 wet season. At 02:20 hours on 27 January 2014 the turbidity signal from the event reached GCDS and was substantially diluted, with a maximum turbidity of 49 NTU. The SS are currently undertaking a geomorphological assessment to identify the source of the turbidity.

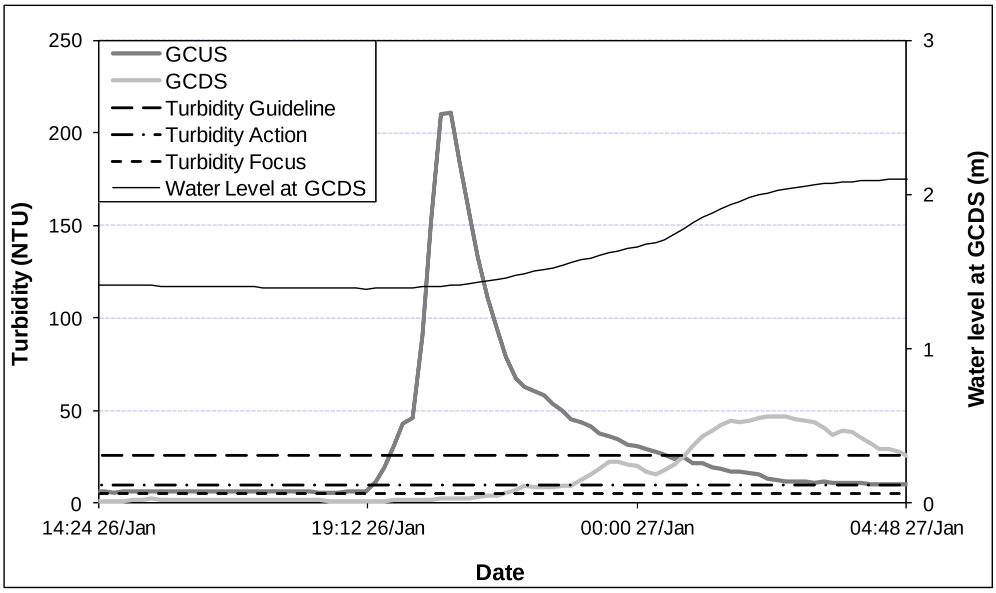

Figure 20 Gulungul Creek continuous monitoring data showing a peak in turbidity at the upstream monitoring site (GCUS) on 26 January 2014.

Uranium concentrations recorded within Gulungul Creek were all below the Focus trigger value of 0.3 µg L^{-1} (Figure 21). The maximum uranium concentration was 0.29 µg L^{-1} recorded at GCUS on 21 December 2013 at 20:25 hours. This is approximately 5% of the ecotoxicologically derived Limit.

Manganese concentrations were also very low in Gulungul Creek with all samples recording concentrations below the Focus trigger value of 35 µg L^{-1} (Figure 22). The maximum manganese concentration was 12 µgL^{-1} recorded at GCDS on 12 December 2013 at 10:48 hours. This is 16% of the ecotoxicologically derived limit.

Magnesium concentrations measured during 2013–14 were generally low with the exception of automatic samples triggered for an EC peak at GCDS on 18 April 2014. The GCDS EC event had two samples with magnesium concentrations above the chronic (>72 hour) Limit trigger value of 3 mg L^{-1}. The maximum magnesium concentration was 4.0 mg L^{-1} at 20.30 pm (Figure 23). The corresponding sulphate concentration was 12 mg L^{-1} (Figure 24). EC remained above the chronic (>72 hour) Limit of 42 µS/cm for 3 hours duration. The ecotoxicologically-derived framework for EC pulses provides a magnesium limit of 94 mg L^{-1} for pulses under 4 hours duration so this magnesium concentration of 4.0 mg L^{-1} is unlikely to have impacted the environment.

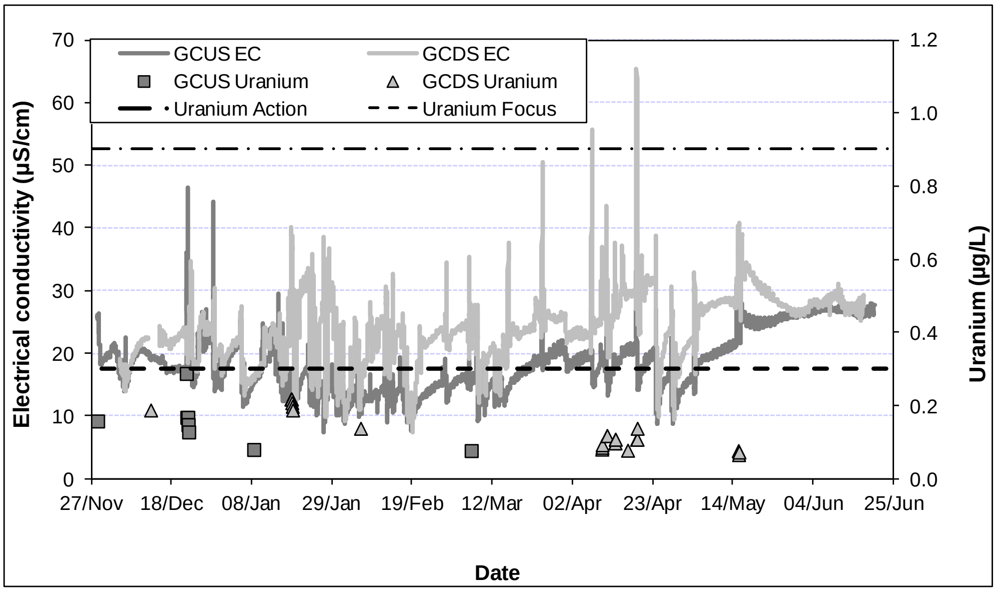

Figure 21 Uranium concentration and continuous monitoring of EC in Gulungul Creek during the 2013–14 wet season.

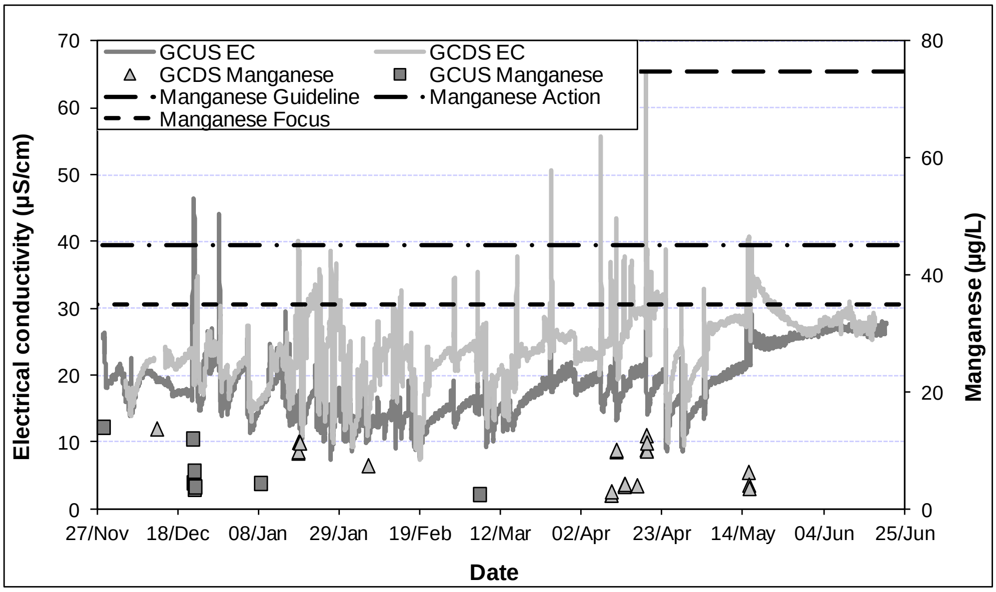

Figure 22 Manganese concentration and continuous monitoring of EC in Gulungul Creek during the 2013–14 wet season.

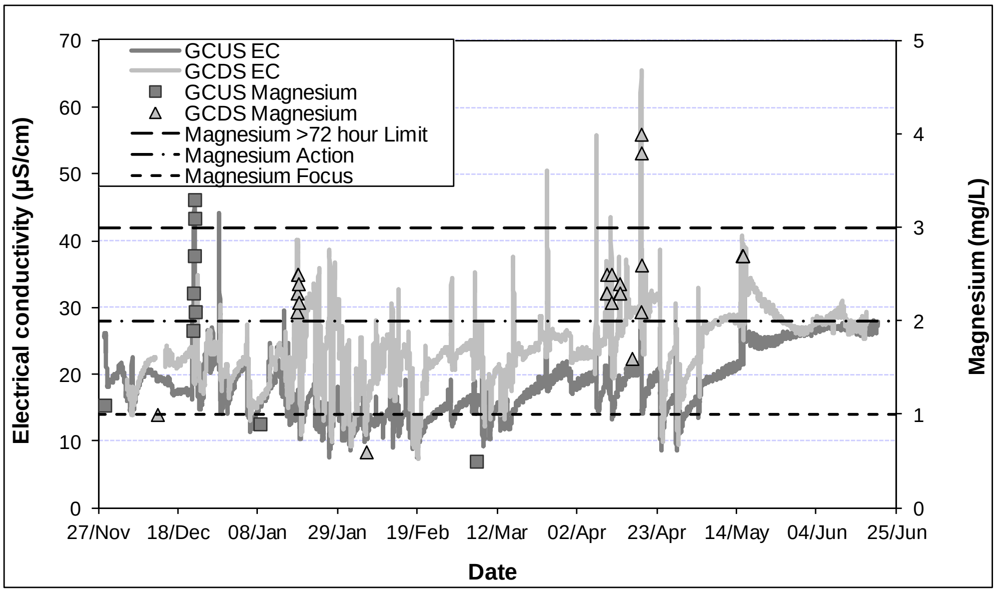

Figure 23 Magnesium concentration and continuous monitoring of EC in Gulungul Creek during the 2013–14 wet season.

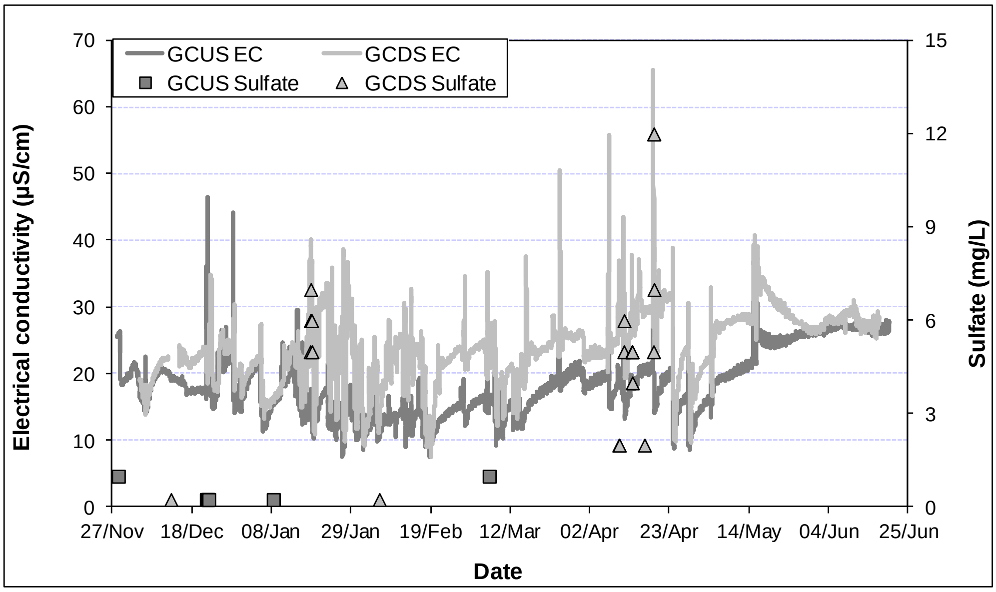

Figure 24 Sulfate concentration and continuous monitoring of EC in Gulungul Creek during the 2013–14 wet season.

During late February and March the water levels within Gulungul Creek decreased leading to gradually increasing EC levels, fluctuating with each rainfall event.

EC peaked at 51 µS/cm at 03.30 am on 25 March 2014 and at 55 µS/cm at 02.00 am on 7 April 2014 in response to rainfall events flushing solutes into the creek. The total rainfall recorded at Jabiru Airport for these events was 20 mm and 41 mm respectively. During late March and early April hardware malfunctions occurred in the sample trigger mechanism and then the turbidity probe, which meant that no water samples were collected during these EC events. SS is looking to increase the capacity of the autosampling equipment at GCDS by installing an additional sampler for the 2014–15 wet season so there are dedicated EC and turbidity samplers, as used in Magela Creek downstream.

The duration of the EC peaks above the chronic (>72 hour) Limit of 42 µS/cm was 4 hours 10 minutes on 25 March 2014 and 3 hours 15 minutes on 7 April 2014. The ecotoxicologically-derived framework for EC pulses provides EC limits of 1040 µS/cm for pulses under 4 hours EC peak duration and for the 4 hour 10 minute duration the derived Limit is 1010 µS/cm. Thus, with recorded EC peaks of 51 µS/cm and 55 µS/cm respectively, the environment is unlikely to have been impacted.

On 18 April 2014 at 21.00 pm EC peaked at the downstream monitoring site (GCDS) at 65 µS/cm and remained above the chronic (>72 hour) Limit of 42 µS/cm for 3 hours duration (Figure 25). The ecotoxicologically-derived framework for EC pulses provides EC limits of 1040 µS/cm for pulses under 4 hours EC peak duration so this EC peak of 65 µS/cm is unlikely to have impacted the environment. The source of these EC events is likely to be rainfall flushing solutes from the GCT2 drainage line as discussed above.

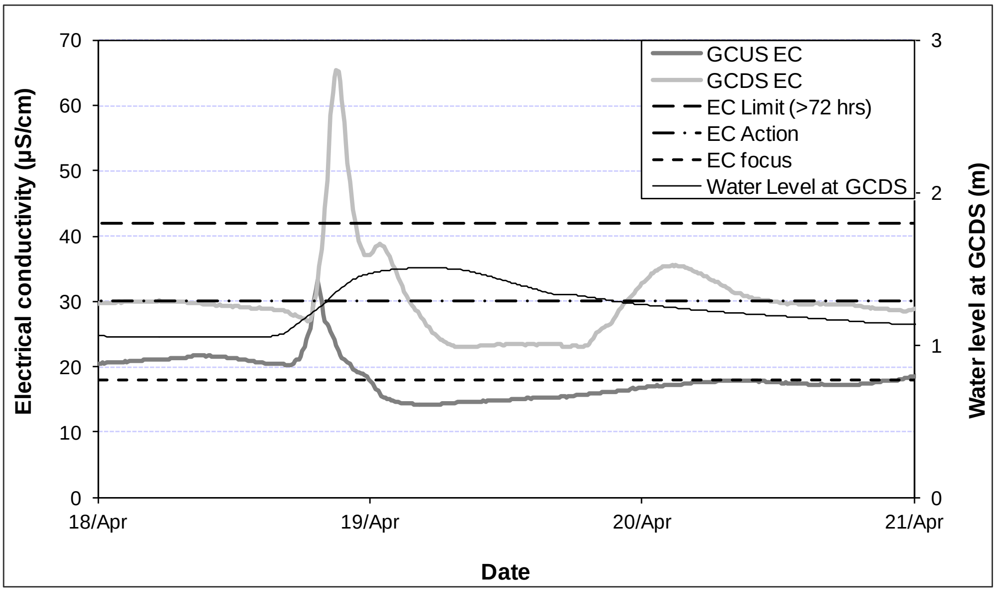

Figure 25 Gulungul Creek continuous monitoring data showing a peak in EC at the downstream monitoring site (GCDS) on 18 April 2014.

Rainfall events during April and early May increased flow levels and resulted in small decreases in EC and increases in turbidity.

A localised rainfall event occurred during the night of 15 May with 76 mm of rainfall recorded at the Jabiru Airport. Runoff from this rainfall event caused an increase in EC at the Gulungul Creek downstream site, peaking at 41 µS/cm. This is below the chronic toxicity limit of 42 µS/cm and not of environmental significance.

Recessional flow conditions became established in May 2014. Continuous monitoring continued until 20 June 2014 when the multi-probes were out of the water and could not be lowered any further. Cease to flow was agreed by stakeholders on 23 June 2014.

Overall, the water quality measured in Gulungul Creek for the 2013–14 wet season showed higher EC at the downstream monitoring site compared to previous wet seasons due to high EC inputs from the GCT2 tributary. However, analysis of the EC peaks at the downstream monitoring site show that the EC peaks are substantially below the Limit for EC pulses < 72 hours, which indicates that the aquatic environment in the creek has remained protected from mining activities (Figure 26).

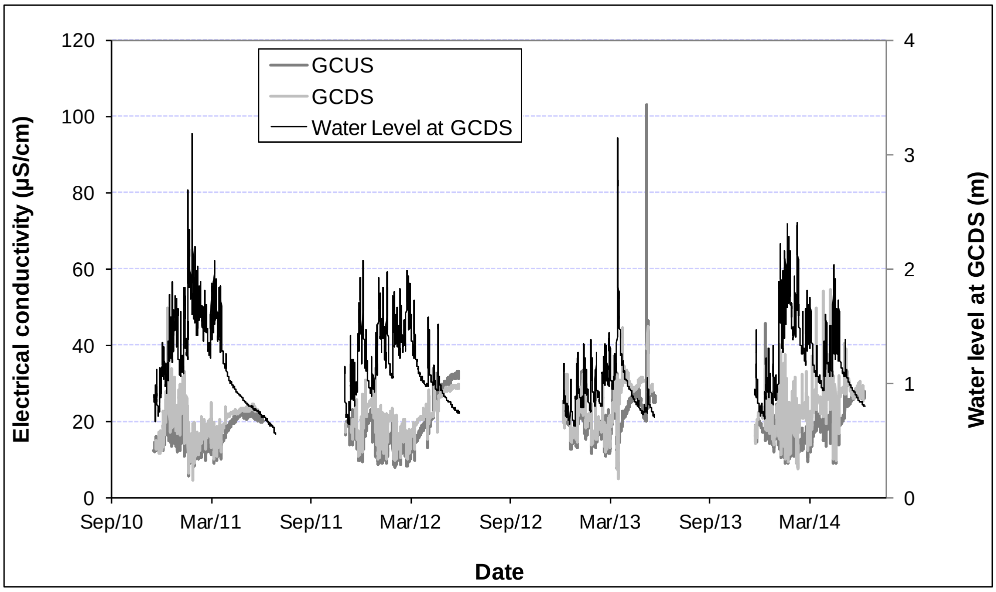

Figure 26 Continuous electrical conductivity and discharge in Gulungul Creek for each wet season between September 2010 and June 2014 (values averaged over a 90 minute period of measurement).

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Sinclair A, Tayler K, van Dam R & Hogan A 2014. Site-specific water quality guidelines: 2. Development of a water quality regulation framework for pulse exposures of mine water discharges at a uranium mine in northern Australia. *Environmental Science and Pollution Research* 21, 31–140.

In situ toxicity monitoring in Magela and Gulungul Creeks (routine monitoring)

CL Humphrey, M Ellis & J Hanley

Background

In this form of monitoring, effects on receiving waters of water dispersed from the Ranger minesite are evaluated using responses of aquatic animals exposed in situ to creek water. The response measured is reproduction (egg production) in the freshwater snail, *Amerianna cumingi*. Each test runs over a four-day (96-hr) exposure period. In such sub-chronic, continuous exposure situations, this species has been shown to be among the most sensitive, to both uranium and magnesium, of SSD's suite of six local species, as determined using standardised laboratory toxicity test protocols.

For the 1990–91 to 2007–08 wet seasons, toxicity monitoring was carried out using the 'creekside' methodology (Figure 1A). This involved pumping a continuous flow of water from the adjacent Magela Creek through tanks containing test animals located under a shelter on the creek bank. In the 2006–07 wet season, an in situ testing method commenced, in which test animals are placed in floating (flow-through) containers located in the creek itself (see section 3.2 of the 2007–08 Supervising Scientist Annual Report for details). Thus, for the 2006–07 and 2007–08 wet seasons, creekside and in situ testing were conducted in parallel, to evaluate the effectiveness of the in situ method. For current data analyses, creekside data up to and including the 2005–06 wet season and in situ data from the 2006–07 wet season onward (Figure 1) are combined. The most recent refinement to this programme has been the extension of toxicity monitoring to Gulungul Creek, with testing commencing in the 2009–10 wet season (Figure 1B).

Analysis of results

The first of 12 tests in Magela Creek commenced on the 5th December 2013, seven days after the establishment of continuous flow in both creeks. The leach tank failure that occurred on 7 December 2013 (see references to the incident in earlier sections of this report) thus coincided with the first test in Magela Creek. Following the incident, three more weekly tests in Magela Creek followed before the commencement of the first of nine Gulungul tests (from 9 January 2014). Thereafter, the Gulungul tests alternated weekly with Magela tests. A combined total of 21 tests were completed over 22 wet season weeks, the highest number of tests yet for a wet season, with the final test completed in Magela Creek on the 28 April 2014. Upstream and downstream egg production and difference values for both creeks are displayed in Figure 1B.

In the previous (2012–13) wet season, a marked increase in overall egg production was observed compared to the previous five wet seasons. This above-average increase in egg production continued in both creeks in the 2013–14 wet season (Figure 1B). As reported previously and elsewhere (Humphrey et al. 2014), Humphrey & Ellis 2014, 2015), a significant factor contributing to this appears to be a more effective culturing regime for the snails at the laboratory aquaculture facility.

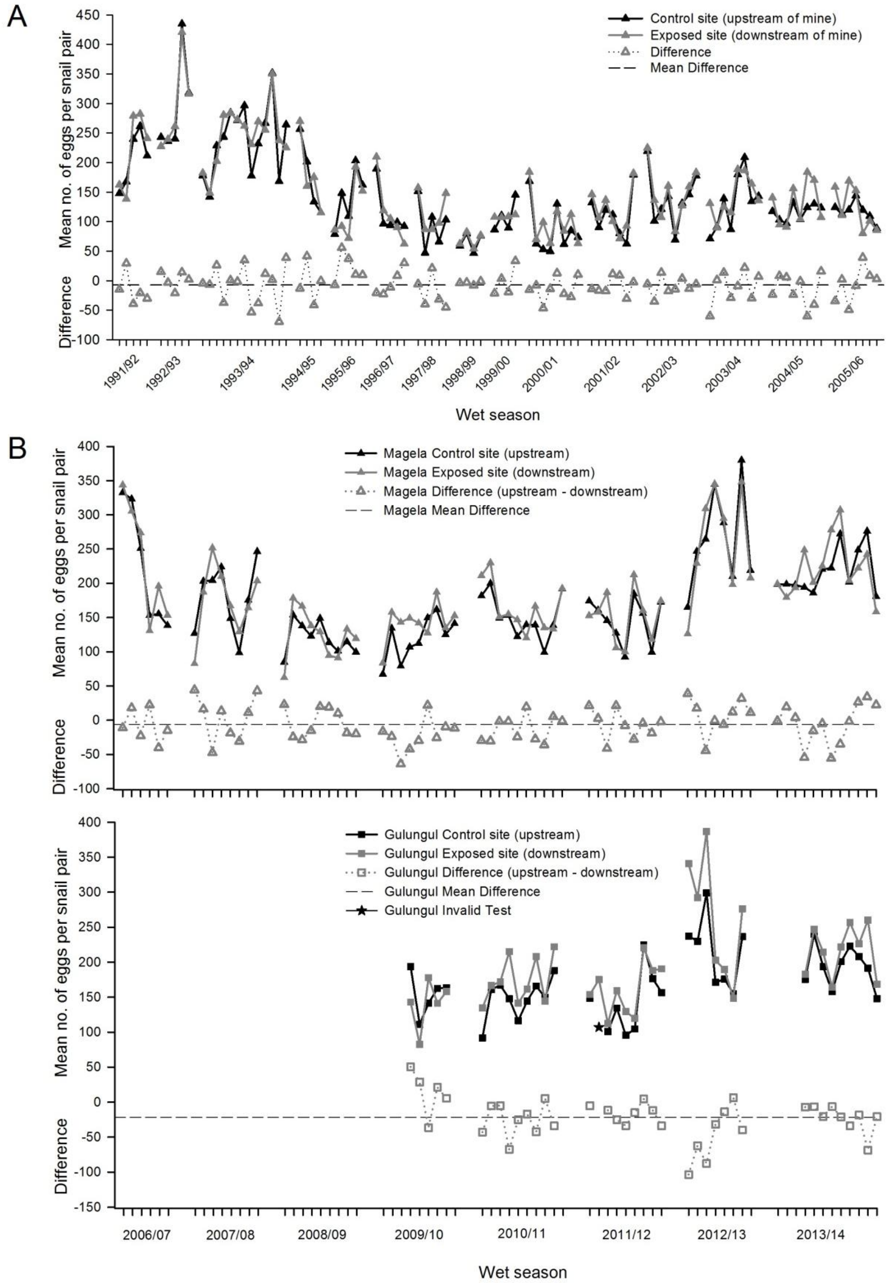

Figure 1 Time-series of snail egg production data from toxicity monitoring tests conducted in Magela Creek using A: creekside tests, and B: *in situ* tests with Gulungul tests commencing in 2009–10.

After each wet season, toxicity monitoring results for the tests are analysed, with differences in egg numbers (the 'response' variable) between the upstream (control) and downstream (exposed) sites tested for statistical change between the wet season just completed and previous wet seasons. This Before-After Control-Impact Paired (BACIP) design, with analysis of variance (ANOVA) testing, is described further in the Supervising Scientist's Annual Report for 2007–2008 (section 2.2.3).

Magela Creek

The historical trend of greater downstream egg production was evident for the 2013–14 wet season, with a mean difference value of -5.21 (mean difference value across all wet seasons of -7.17). ANOVA results for the 2013–14 wet season, together with results from the past several wet seasons, are displayed in Table 1. No significant difference was observed between the difference values derived from the 2013–14 wet season and those from previous wet seasons ($p = 0.865$). The (near-)significant differences observed in previous years, associated with particularly high egg production at the downstream site relative to the upstream site in the 2009–10 wet season, and lower egg production at the downstream site relative to the upstream site in the 2012–13 wet season (Table 1), are discussed in the respective Supervising Scientist annual reports.

Table 1 Results of ANOVA testing comparing Magela upstream-downstream difference values for mean snail egg number for different 'Before versus After' wet season scenarios.

Before	After	Probability value (P)	Significance
All previous seasons	2009–10	0.043	at 5% level
All previous seasons	2010–11	0.436	NS
All previous seasons	2011–12	0.916	NS
All previous seasons	2012–13	0.076	NS
All previous seasons	2013–14	0.865	NS

NS = Not significant

Gulungul Creek

The mean difference value across all Gulungul Creek tests for 2013–14, of -22.5, continued the trend of greater egg production downstream reported in previous years. ANOVA testing found no significant difference between the 2013–14 difference values and those recorded in previous wet seasons ($p = 0.898$).

Apart from the primary Before/After factor and associated hypothesis, the particular two-factor ANOVA model used for toxicity monitoring also allows variation amongst years (or wet seasons) and among tests within a wet season to be estimated separately. The second 'Season' factor can be used to determine whether, within the Before and After periods, any set of difference values for a wet season is significantly different. For Gulungul Creek, the Season factor has been significant since the 2011–12 wet season (inclusive), with a significant value of 0.003 for the 2013–14 wet season (cf Magela Creek where this factor has never been significant). A significant Season factor does not in itself imply potential mine-related impact; in this (Gulungul) case, it highlights the high inter-annual variation observed in seasonal difference values, as shown in Figure 1B and as reported previously by Humphrey et al. (2014).

Assessment and conclusions for both creek systems

As reported previously by Humphrey & Ellis (2014), water temperature and EC influence snail egg laying response in Magela and Gulungul creeks. (Values of water temperature and EC representing each test are the median of 10 minute continuous readings taken across the four-day exposure period at each of the creek sites.) An interacting effect between water temperature and EC has been observed (section 4.4.2). As EC increases (generally across the range ~7-30 $\mu S\ cm^{-1}$), egg production:

- increases at lower water temperature ranges (27–30°C), and
- decreases (i.e. negative effect) at higher water temperatures (>30°C) (see also Humphrey & Ellis (2015), Figure 2, current Summary).

These findings may usefully be applied to interpreting annual toxicity monitoring results in Magela and Gulungul creeks. While the water quality relationships have limited use in explaining the magnitude of egg production at a site, they can generally explain the difference, + or –, in egg production between the paired upstream and downstream sites. Unlike the below-average rainfall in the 2012–13 wet season, rainfall during the 2013–14 wet season was above average (Figure 3.1, Supervising Scientist, Annual Report 2013–14), which resulted in both greater minesite runoff (and thus generally higher downstream EC in both creeks) and generally cooler water temperatures (median values <30°C). This explains why, for most tests in the creeks, there was a return to the typically higher downstream egg production. In six Magela Creek tests (#1–4, 10–11), however, median water temperatures at (usually) both sites exceeded 30°C and for all but the fourth test, lower downstream (compared to upstream) egg production was observed.

A (four-day) downstream median EC in Magela and Gulungul creeks greater than 20 µS cm^{-1}, represents a value typically associated with mine waste water discharges (Humphrey et al. 2013). Five of the 12 Magela tests and eight of the nine Gulungul tests were conducted under these minewater exposure regimes. Such elevated EC in two Magela tests conducted at median water temperatures >30°C appear to have contributed to reduced downstream (compared to upstream) egg production while elevated EC in another Magela and eight Gulungul tests conducted at median water temperatures <30°C appear to have contributed to greater downstream (compared to upstream) egg production. At this stage these mine-related 'effects', representing enhancement in snail reproductive activity in most cases, are not regarded as constituting environmental concern.

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Bioaccumulation in freshwater mussels and fish

A Bollhöfer, C Doering, P Medley & T Fox

Introduction

Metals and radionuclides bioaccumulate in aquatic biota, in particular, freshwater mussels of Magela Creek and tributaries. Thus, it is essential to check that food items collected downstream of Ranger uranium mine are fit for human consumption and that concentrations of metals and radionuclides in organism tissues attributable to Ranger remain within acceptable levels. Mudginberri Billabong, a permanent waterbody located 12 km downstream of the Ranger uranium mine along Magela Creek, has been the focus of the monitoring programme initiated by the Supervising Scientist in 2000 to (a) assure that mussels (annually) and fish (biennially) collected are fit for human consumption and tissue concentrations of radionuclides (and metals) remain within acceptable levels, and (b) provide an early warning of bioavailability of these constituents. As monitoring had not shown any issues of potential concern with regards to bioaccumulation in fish, the focus of the bioaccumulation monitoring programme has been directed at mussel tissue analysis, while the two-yearly fish sampling programme was discontinued in 2007.

Up until 2008, mussels were collected annually from Mudginberri Billabong (the potentially impacted site) and Sandy Billabong (the control site in a different catchment, sampled from 2002 onwards). The results showed that radionuclide burdens in mussels from Mudginberri Billabong were generally about twice that observed in the reference Sandy Billabong. Two research projects reported in previous ***eriss*** Research Summaries concluded that this difference was due to natural catchment influences and differences in water chemistry, rather than mining-related inputs to Magela Creek (Bollhöfer et al. 2011). Thus, the scope of the bioaccumulation monitoring programme was reduced from 2009 onwards. It now involves the annual collection and analysis of a bulk mussel sample from Mudginberri Billabong, rather than analysing separate age-classed mussels from both Mudginberri and Sandy Billabongs, primarily to provide re-assurance that the consumption of mussels does not present a radiological risk to the public. Every three years, starting with the October 2011 collection, a detailed study (analysis of aged mussels from Mudginberri and Sandy Billabongs) is conducted and results compared with those from previous years. Monitoring focuses on radium-226 (^{226}Ra) as it has been shown that ^{226}Ra in mussels is the biggest potential contributor to mine-related ingestion dose from a hypothetical release of pond waters from the minesite (Martin 2000).

In 2012, an Independent Surface Water Working Group (ISWWG) was convened by ERA and GAC, with findings of this review released in March 2013. One of the ISWWG recommendations was for SSD to review existing metals in bush tucker data (including fish) and provide advice on a potential re-introduction of a metals in bushtucker monitoring programme.

Methods

Mussel, sediment and water samples were collected in October 2013 from Mudginberri Billabong. After collection, mussels were placed into acid-washed containers holding billabong water. Surface water samples were collected at the same time in acid washed containers and sediments associated with the mussels were collected and stored in zip-locked plastic bags. The mussels, water and sediment samples were taken to the Darwin laboratories for processing, where they were purged in host billabong water for three days, before being measured for length, breadth and width, and dissected to remove the mussel flesh. The wet weight of the mussel flesh was recorded and samples were freeze dried then re-weighed to determine the dry weight. The age of each mussel was determined by counting the number of annual growth bands preserved in the mussel shell (Humphrey & Simpson 1985). While the routine sampling schedule required just the analysis of a bulk sample for this year, mussels were aged and individual age groups analysed for ^{226}Ra and lead-210 (^{210}Pb) via gamma spectrometry, uranium and other metals via ICPMS, and polonium-210 (^{210}Po) via alpha spectrometry.

The BRUCE tool (**B**ioaccumulation of **R**adioactive **U**-series **C**onstituents from the **E**nvironment), a database that has been developed by SSD to collate biota and media radionuclide data and calculate concentration ratios for various biota-radionuclide combinations (Doering 2013, Doering & Bollhöfer 2014), has been amended in 2013 to include metal concentration data and is continually being populated. This tool has been used to extract geometric mean, or typical, concentrations on a wet tissue weight basis for As, Cd and Pb, measured in mussel and fish as part of SSD's bioaccumulation monitoring over the past decades. Values are given in this report and are compared with the Australian/New Zealand food standards for these metals in molluscs and fish.

Uranium in freshwater mussels

Uranium concentrations in mussels (mg kg^{-1} dry weight) and water collected concurrently from Mudginberri and Sandy Billabongs over the past 14 years are shown in Figure 1.

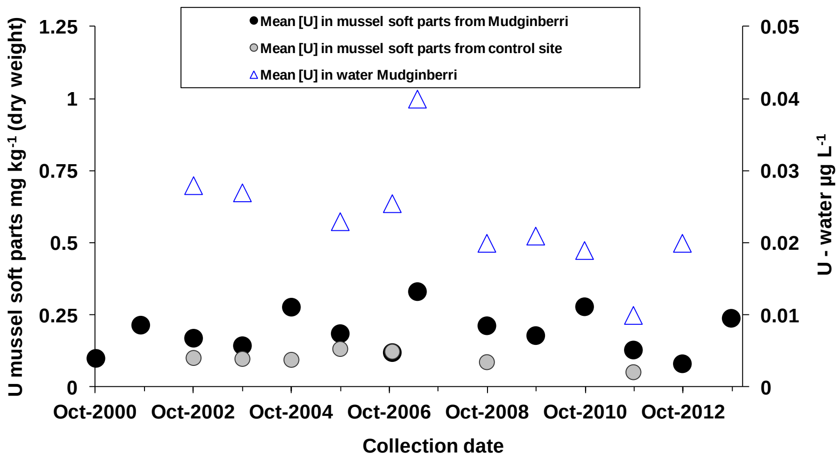

Figure 1 Mean concentrations of U measured in mussel soft-parts and water samples collected from Mudginberri and Sandy Billabongs since 2000.

The average concentrations of uranium in mussels from Mudginberri Billabong are very similar from 2000 onwards, with no evidence of an increasing trend in concentration over time. Essentially constant and low levels were also observed between 1989 and 1995 (previous reports). Notwithstanding some bioaccumulation with age, uranium in mussels is reported to have a short biological half-life, a conclusion that is supported by our data.

The low and constant uranium concentrations including the last sample taken in October 2013 indicate absence of any mining influence.

Radium-226,lead-210 and polonium-210 in mussels

^{226}Ra, ^{210}Pb and ^{210}Po activity concentrations (Bq kg^{-1} dry weight) in mussels collected from Mudginberri Billabong in 2013 are compared with the average activity concentrations measured in previous years in Figure 2.

^{226}Ra activity concentrations appear higher in mussels collected in 2013 compared to the average from previous collections. ^{210}Po is higher than the ^{210}Pb activity concentration, indicating higher accumulation of ^{210}Po from the water column compared to ^{210}Pb, in agreement with previous observations in the Alligator Rivers Region and elsewhere. There is no increase in ^{210}Po activity concentration with age in 1–10 year old mussels, consistent with its short physical halflife of 138 days. Average ^{210}Po activity concentration in mussels collected in 2013 (open symbols: 420 ± 60 Bq kg^{-1}) is higher than in 2012 (solid symbols: 350 ± 30 Bq kg^{-1}), in particular in three, four, five and six year old mussels.

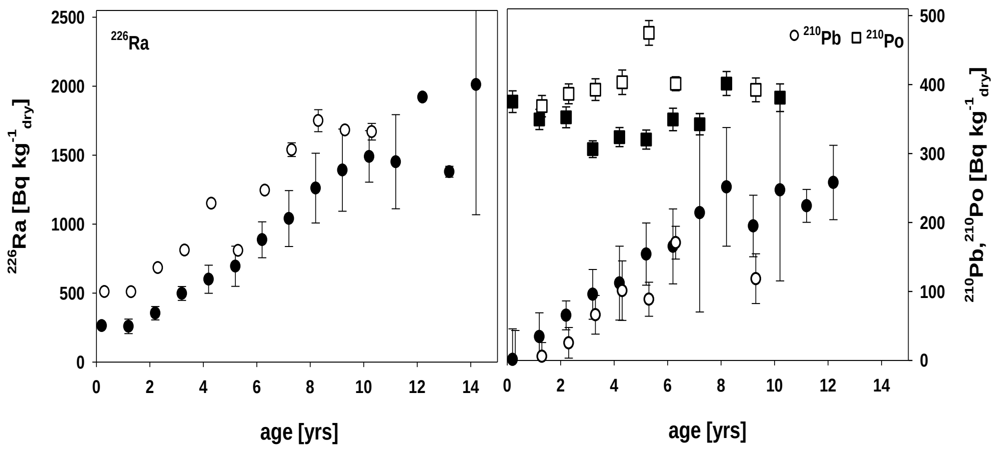

Figure 2 ^{226}Ra, ^{210}Pb and ^{210}Po activity concentrations measured in dry mussel flesh from Mudginberri Billabong plotted against mussel age. Averages of previous end of dry season collections (2000–12) are shown as solid symbols, open symbols show the results from the 2013 collection. Errors shown are 1 standard deviation of the mean.

The higher ^{226}Ra and ^{210}Po activity concentrations are unlikely to be mine-related. Figure 3 shows that ^{228}Ra activity concentrations in mussels collected in 2013 were higher on average as well, although the ^{228}Ra isotope is a member of the thorium decay series. In addition, Figure 3 shows that, compared to previous collections, the ^{228}Ra /^{226}Ra activity ratios in aged mussels have not changed in 2013, indicating that the increase in ^{226}Ra activity concentrations are not due to any mine related increases in water ^{226}Ra activity concentrations. This is confirmed by the low water ^{226}Ra activity concentration in Mudginberri Billabong in October 2013 of only 1.46 ± 0.14 mBq L^{-1}.

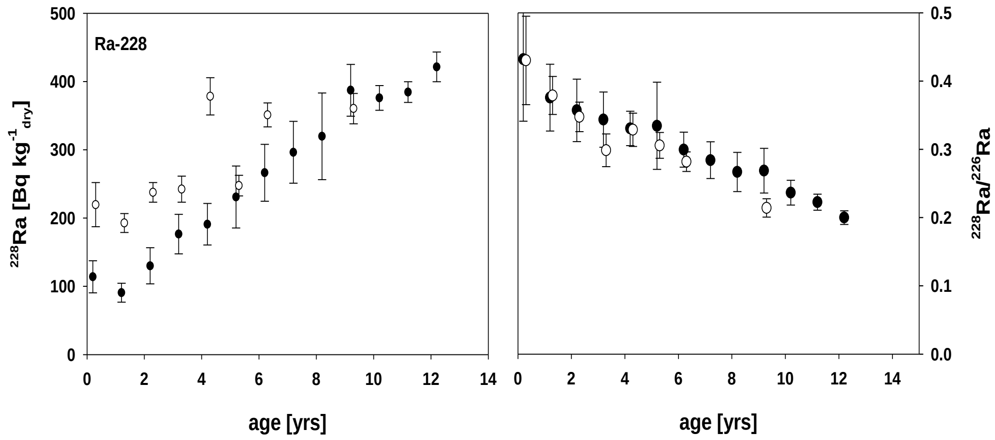

Figure 3 ^{228}Ra activity concentrations and ^{228}Ra/^{226}Ra activity ratios in dry mussel flesh from Mudginberri Billabong plotted against mussel age. Averages of previous end of dry season collections (2000–12) are shown as solid symbols, open symbols show the results from the 2013 collection.

Arsenic, cadmium and lead in mussel and fish tissue from Mudginberri Billabong

Table 1 shows the typical concentrations of arsenic (As), cadmium (Cd) and lead (Pb) measured in mussel and fish tissue collected during past monitoring in Mudginberri Billabong. Concentrations, in particular in fish tissue, are low and 5 - 100 times lower than the maximum levels in fish and molluscs given in the Australian/New Zealand Food Standards (FSANZ 2010). Population of the database with metal concentration data for various bush food items is ongoing and more results will be presented in future reports.

Table 1 Maximum levels for metal contaminants in fish and molluscs given in the Australian/New Zealand food standard (FSANZ 2010) and typical concentrations (mg kg^{-1} wet weight) measured in fish and mussel tissue collected from Mudginberri and Sandy Billabongs.

Biota		Aus/NZ Food Standard	Mudginberri Billabong[1]	Sandy Billabong[1]
	As	2.0	0.02; 2.6 (32)	0.01; 1.7 (20)
Fish[2]	Cd	0.2[3]	0.01; 4.1 (44)	0.002; 1.9 (20)
	Pb	0.5	0.03; 2.8 (44)	0.01; 1.8 (20)
	As	1.0	0.19; 1.4 (73)	0.26; 1.5 (36)
Molluscs	Cd	2.0	0.06; 1.9 (102)	0.07; 1.6 (47)
	Pb	2.0	0.29; 2.3 (160)	0.09; 1.7 (61)

1 Values shown are: geometric mean; geometric mean standard deviation (*number of samples*)

2 Concentrations in fish given as less than the detection level (DL) were included as 0.5·DL

3 In the absence of a cadmium standard for fish, one 10th of the standard for molluscs was assumed

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Monitoring using macroinvertebrate community structure

L Chandler, J Hanley & C Humphrey

Background

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 refined over this period (changes are described in the 2003–04 Supervising Scientist annual report, section 2.2.3). The present design is 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 are 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 '100%' indicates totally dissimilar communities, sharing no common taxa. Disturbed sites may be associated with significantly higher dissimilarity values compared with undisturbed sites (Supervising Scientist Division 2013).

In the following report, data from late wet season sampling in 2014 at sites in Magela and Gulungul creeks are compared with historical data and data collected at the same time from control sites upstream, or in streams unaffected by contemporary mining, to assess potential cumulative water quality impacts arising from Ranger during the 2013–14 wet season.

Results

Compilation of the full macroinvertebrate dataset from 1988 to 2014 has been completed with results shown in Figure 1. This figure plots the paired-site dissimilarity values using family-level (log-transformed) data, for the two 'exposed' streams and the two 'control' streams.

For statistical analysis, dissimilarity values for each of the five possible, randomly-paired, upstream and downstream replicates within each stream are derived. These replicate dissimilarity values may then be used to test whether or not macroinvertebrate community structure has altered significantly at the exposed sites for the wet season of interest. For this multi-factor 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).

A four-factor ANOVA model based on replicate, paired-site dissimilarity values, was performed using the factors Before/After (BA: fixed), Control/Impact (CI or 'Exposure'; fixed), Year (nested within BA; random) and Stream (nested within CI;

random) to determine if any change has occurred. The ANOVA showed no significant change from the before (pre-2014) to the after (2014) periods in the magnitude of upstream-downstream dissimilarity between the control and exposed streams ($p = 0.189$ and $p = 0.838$ for BA and BA*Exposure interaction respectively).

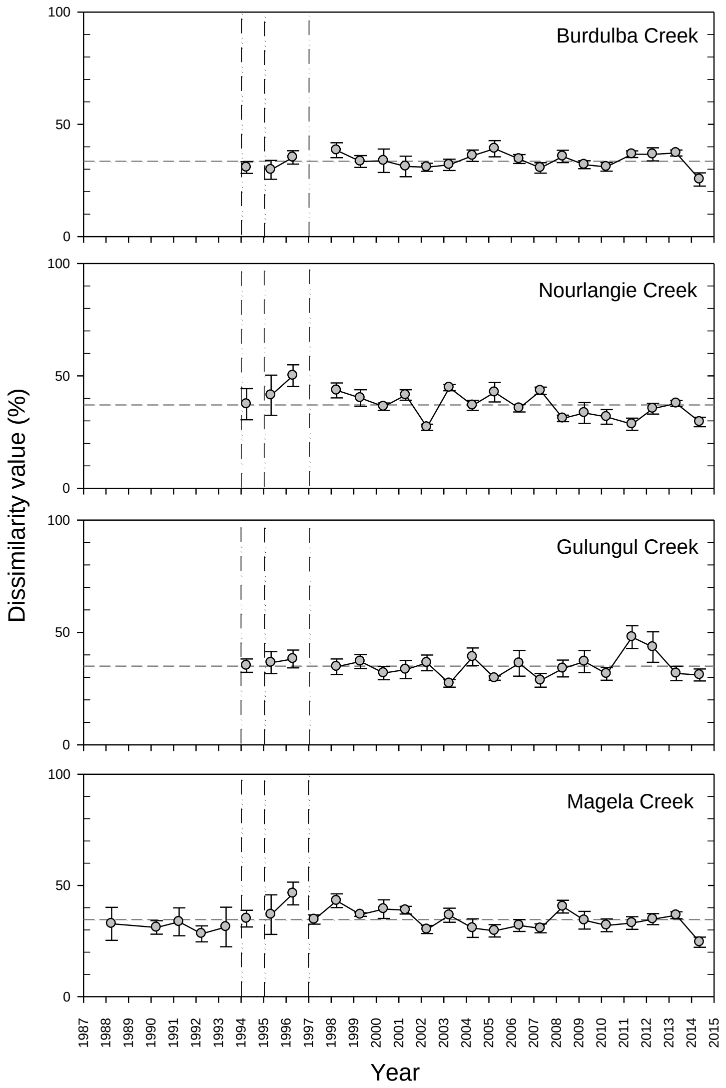

Figure 1 Paired upstream-downstream dissimilarity values (using the Bray-Curtis measure) calculated for community structure of macroinvertebrate families in several streams in the vicinity of the Ranger mine for the period 1988 to 2014. The dashed vertical lines delineate periods for which a different 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.

A lack of significance in the BA and BA*Exposure interaction was observed despite a drop in dissimilarity values for all creeks, especially for Magela, Burdulba and Nourlangie creeks in 2014 (Figure 1), indicating that the upstream and corresponding downstream sites were more similar at the time of sample collection. A marked change in dissimilarity was last reported in 2011 for Gulungul Creek, but in that instance the dissimilarity *increased* relative to historical values (Figure 1). In 2011, the BA*Stream factor was significant ($p = 0.014$). Further examination of the data indicated this result was associated with an increase of flow dependant taxa at the upstream control site in Gulungul Creek and not associated with any mine-related change (Humphrey et al. 2012).

The lower paired upstream-downstream dissimilarity values observed in Magela, Nourlangie and Burdulba creeks may be associated with a combination of the generally lower water levels in the creeks at the time of sampling compared to previous years and a change in sampling locations required because of unsuitable habitat in the usual sampling location (Magela Creek upstream and both Nourlangie Creek sites). Lower creek flows can confer more similar macroinvertebrate communities between creek sites (i.e. lower dissimilarity; see Supervising Scientist Division 2013, Figure 4, and George & Humphrey 2014, Figure 3).

The paired-site dissimilarity approach, used in the ANOVA and depicted in Figure 1, results in loss of information: (i) the influence of individual sites that comprise the paired-site dissimilarity cannot be determined; and (ii) *direction* of change in multivariate space can also not be determined (Supervising Scientist Division 2013). Thus it is not possible to assign any change to either the upstream or downstream sites, while change in macroinvertebrate community structure at the exposed downstream site could occur in different directions in multivariate space. Therefore, a non-significant BA (before vs after dissimilarity) could still mask real change that is occurring at the exposed site.

To ensure change is not passing undetected, two additional multivariate techniques are employed. Firstly, Multivariate Dimensional Scaling (MDS) ordination is used to discern possible shifts in multivariate space at the downstream exposed sites away from control sites. Secondly, PERMANOVA (PERmutational Multivariate ANalysis Of Variance) (Anderson 2001, McArdle & Anderson 2001, Anderson et al. 2008) is used to perform hypothesis tests on the multivariate data from individual sites (as distinct from the paired-site dissimilarity for the ANOVA above). Use of data from the individual sites offers increased partitioning of data variation (and hence increased factors). The ability to use the complete multivariate dissimilarity matrix enables PERMANOVA to better detect changes in direction in multivariate space that might otherwise be missed when using the simple site-pair dissimilarity metric data.

Figure 2 depicts the multivariate ordination derived using replicate within-site macroinvertebrate data. Data points are displayed in terms of the sites sampled in Magela and Gulungul creeks downstream of Ranger mine for each year of study (to 2014), relative to Magela and Gulungul Creek upstream (control) sites for 2014, and all other control sites sampled up to 2014 (Magela and Gulungul upstream sites, all sites in Burdulba and Nourlangie). Samples situated close to one another in the ordination indicate a similar community structure.

Data points associated with the 2014 Gulungul and Magela downstream sites are generally interspersed among the points representing the control sites, indicating that these 'exposed' sites have macroinvertebrate communities that are similar to those occurring at control sites. PERMANOVA conducted on all replicate data from all available years and sites showed a significant differences for BA ($p = 0.0086$). However, the two interactions important for impact detection showed no significant difference (i.e. BA * Stream (Exposure) * Upstream/Downstream $p = 0.299$; and BA * Exposure * Upstream/Downstream, $p = 0.5348$).

Whilst not indicative of a potential mine impact, further investigation was undertaken as to why there was such a significant difference in the Before and After factor, across all sites and exposures in 2014. A pair-wise comparison test indicated that both the upstream Nourlangie and downstream Burdulba sites were significantly different from the before to after periods ($p = 0.0082$ & $p = 0.0029$, respectively). This difference can be attributed to a dramatic increase in the total taxa abundance at both locations (Figures 3 & 4) and in both cases a single family (Chironomidae) was the main contributor to this trend.

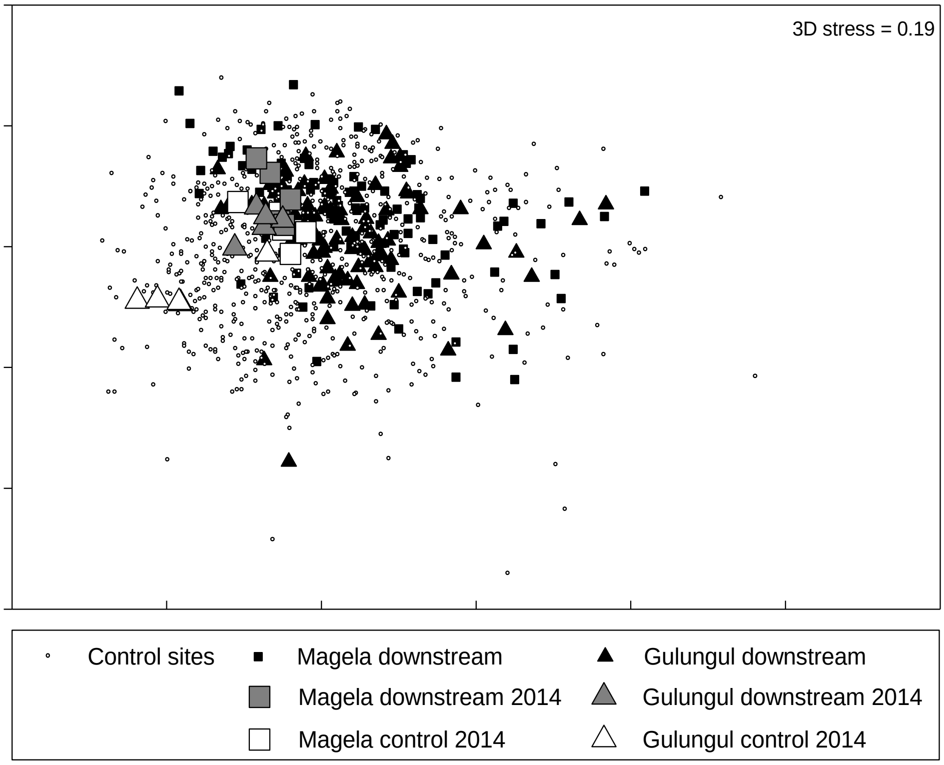

Figure 2 Ordination plot (axis 1 and 2) of macroinvertebrate community structure data from sites sampled in several streams in the vicinity of Ranger mine for the period 1988 to 2014. Data from Magela

Nourlangie Creek water levels were lower than usual, and the regular sampling sites were exposed. As a consequence, the sites were relocated to a nearby area with suitable habitat. The lower water levels were a result of the delay in sampling due to a late wet season rainfall event that was isolated to the East Alligator catchment (Magela and Gulungul creek sites only). The 'new' upstream site in Nourlangie Creek, and downstream Burdulba Creek, were characterised by higher than usual flow conditions and macrophyte beds that were dominated by submerged grass-like native annuals (*Blyxa* sp. and *Xyris* sp.) These aquatic macrophytes are typically utilised by larvae of the Chironomidae genus *Rheotanytarsus*, as a surface to which they attach their cases from which they feed. These cases were observed in unusually high numbers in the field. Species-level identification of these samples is currently being carried out to confirm the identity of these chironomids..

These collective results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining during the period 1994 to 2014 have not adversely affected macroinvertebrate communities.

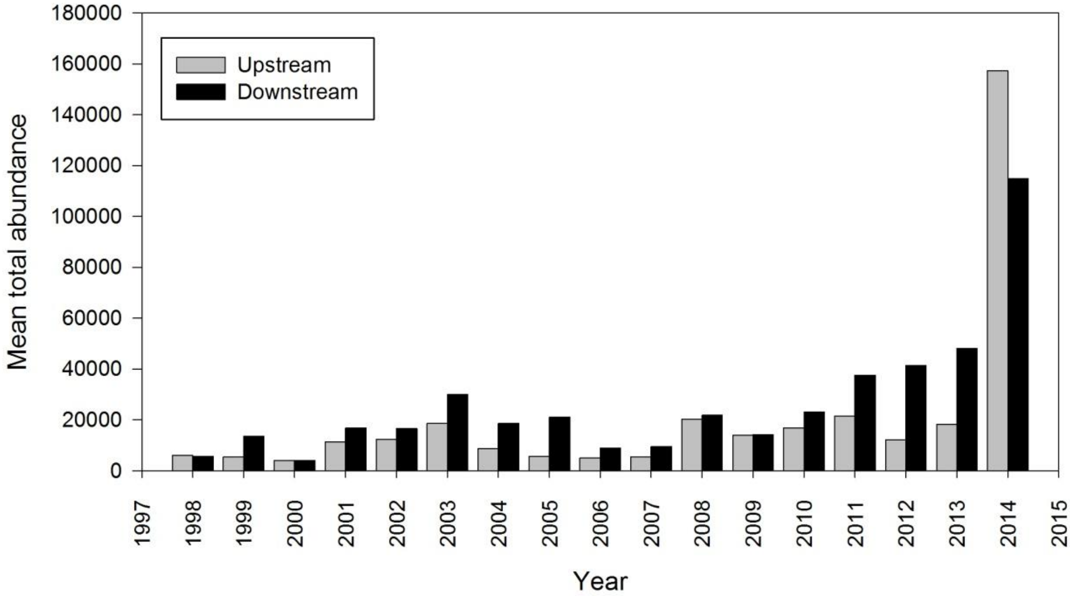

Figure 3 Total abundance of taxa collected from upstream and downstream Nourlangie Creek sites over time.

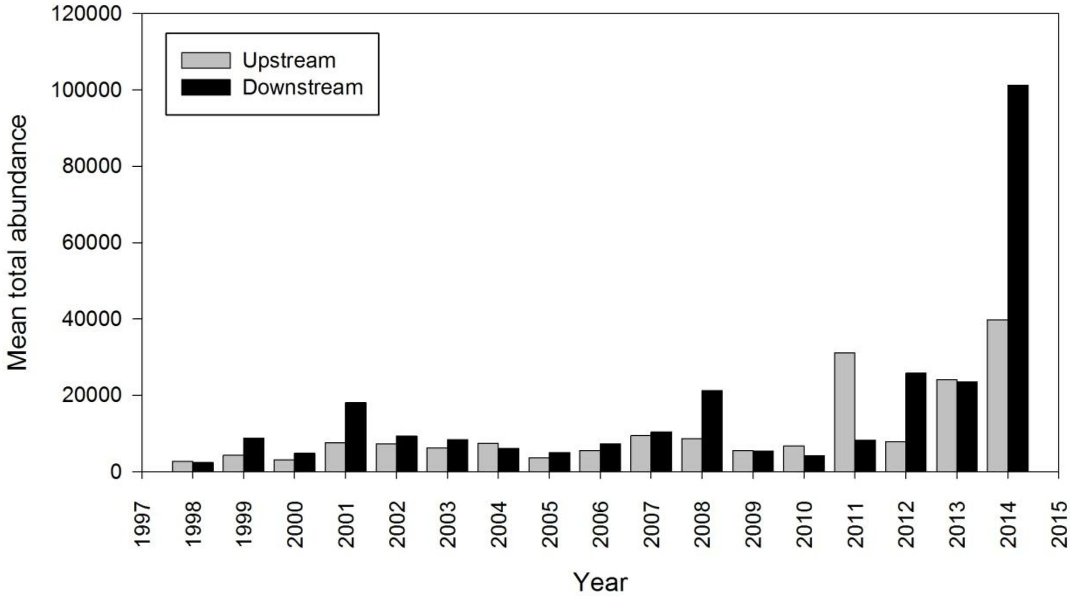

Figure 4 Total abundance of taxa collected from upstream and downstream Burdulba Creek sites over time.

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Monitoring using fish community structure

M Ellis & C Humphrey

Background

Assessment of fish communities in billabongs is conducted between late April and July each sampling year, the precise time of the monitoring being dependent on flow regime, using non-destructive sampling methods at 'exposed' and 'control' locations. Two billabong types are sampled: deep channel billabongs every year and shallow lowland (mostly backflow) billabongs dominated by aquatic plants every two years. Details of the sampling methods and sites were provided in Supervising Scientist (2004; section 2.2.3). These programmes were reviewed in October 2006 and the refinements to their design are detailed in Supervising Scientist (2007, 2008; section 2.2.3). Full protocols for the channel and lowland billabong fish monitoring techniques are provided in Supervising Scientist (2011a, b respectively).

For both deep channel and shallow lowland billabongs, comparisons are made between a directly-exposed billabong in the Magela Creek catchment downstream of the Ranger mine versus control billabongs from an independent catchment (Nourlangie Creek and Wirnmuyurr Creek). The extent of similarity of fish communities in exposed sites to those in control sites is determined using multivariate dissimilarity indices, calculated for each sampling occasion. The use of dissimilarity indices for both of these monitoring approaches has been described and defined in Supervising Scientist Division (2011a, b). A significant change or trend in the dissimilarity values over time could imply mining impact.

Deep channel billabongs

The exposed location for the channel billabong study is Mudginberri Billabong. For the 2014 monitoring period, access to Mudginberri Billabong was not possible due to an important time of mourning for the local Indigenous community. The next assessment will be undertaken in 2015 and will be first reported in the Supervising Scientist annual report for 2014–15.

Shallow lowland billabongs

Monitoring of fish communities in shallow lowland billabongs has previously been conducted every two years, with the exception of a break in sampling between 2009 and 2012. The last assessment of fish communities in these billabongs occurred in June 2012 with results reported in Buckle & Humphrey (2013). Results from sampling conducted in June 2014 are described below.

The monitoring programme for fish communities in shallow billabongs is conducted in six billabongs, comprising three 'control' versus 'exposed' billabong pairs. In a similar manner to fish communities in channel billabongs (see Buckle & Humphrey 2013), the similarity of fish communities in the directly exposed sites downstream of Ranger on Magela Creek (Georgetown, Coonjimba and Gulungul billabongs) to those of the control sites (Sandy Shallow and Buba billabongs on Nourlangie Creek and Wirnmuyurr Billabong – a Magela floodplain tributary) (see Map 3) is determined using multivariate dissimilarity indices calculated for each sampling occasion. A plot of the dissimilarity values of the control-exposed site pairings – Coonjimba-Buba, Georgetown-Sandy Shallow and Gulungul-Wirnmuyurr billabongs – from 1994 to 2014, is shown in Figure 1.

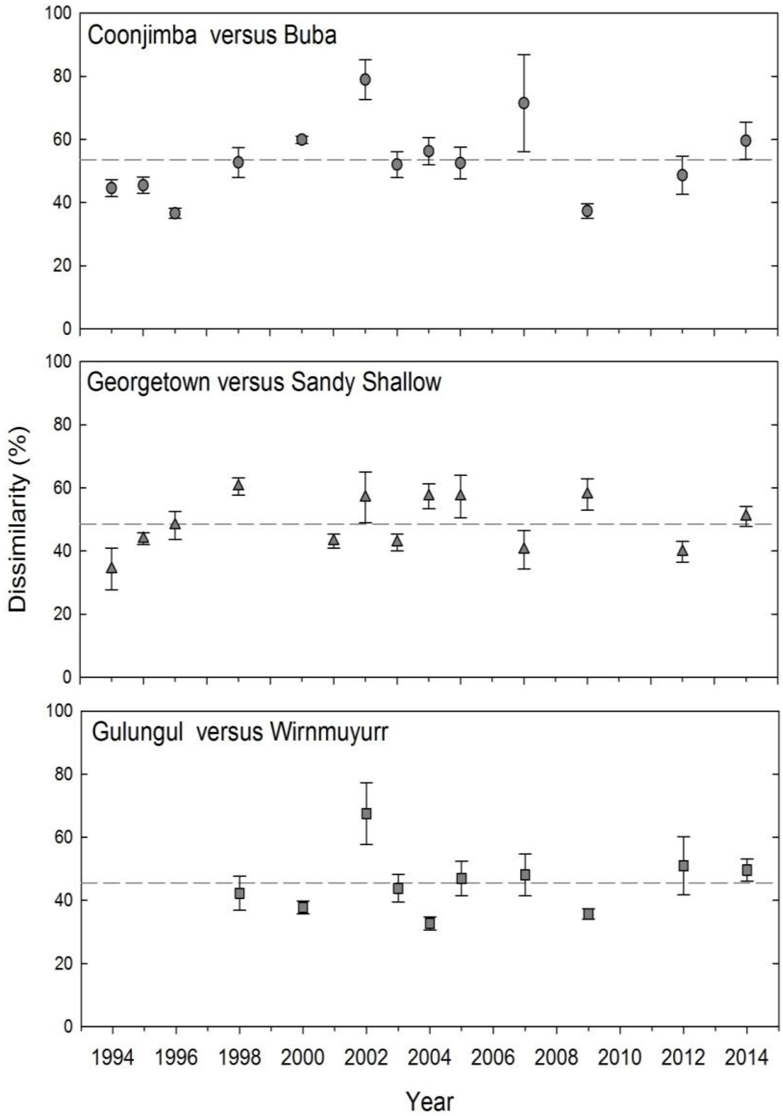

Figure 1 Paired control-exposed site dissimilarity values (using the Bray-Curtis measure) calculated for community structure of fish in 'exposed' Magela and 'control' Nourlangie and Magela Billabongs in the vicinity of Ranger mine over time. Values are means (± standard error) of the 5 possible (randomly-selected)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

The three sets of paired-billabong dissimilarity values measured since 1998 (when sampling of all three site-pairs commenced) have been analysed using a three-factor ANOVA with Before/After (BA; fixed), Year (nested within BA; random) and Site-pair (Fixed) as factors. In this analysis, the BA factor tests whether values for the year of interest (2014) are consistent with the range of values reported in previous years (1998 to 2012), the factor 'Year' tests for differences amongst years within the before or after periods and the 'Site-pair' factor tests for differences amongst the three paired-billabong dissimilarities.

The ANOVA results showed that across all three site-pairs there was no significant change from 2014 to other years (BA factor, $p = 0.763$) and that the change between 2014 and previous years within the individual site-pairs was consistent (BA*Site-pair interaction, $p = 0.953$). These results confirm that dissimilarity values for 2014 for all three site-pairs do not differ from those values from previous years. Significant differences do occur over time within site-pairs (Year*Site-pair interaction, $p = 0.000$), which reflects (natural) changes through time. This variation over time is evident in Figure 1 and is further considered below. The paired-site dissimilarities shown in Figure 1 average between 40 and 60%, indicating fish communities in each of the billabongs comprising a site-pair are quite different from one another. The dissimilarity values appear to reflect differences in aquatic plant communities of the site-pair billabongs, with particularly high dissimilarity values (i.e. Coonjimba-Buba pairing for 2002 and 2007, Gulungul-Wirnmuyurr site pairing for 2002, Figure 1) attributable to high densities of particular aquatic plant types in one or both billabongs in a billabong pair (see Buckle & Humphrey 2008).

Excessive plant densities are unfavourable for fish communities as fish movement, and hence residency, is physically prevented. Collectively, the graphical and statistical results provide good evidence that changes to water quality downstream of Ranger as a consequence of mining during the period 1994 to 2014 have not adversely affected fish communities in shallow billabongs.

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Monitoring of radionuclides and metals in groundwater at Ranger

A Bollhöfer, P Medley & S Marshall

Introduction

Groundwater samples from bores around Ranger uranium mine have been collected by ERA, the Northern Territory Government Department of Mines and Energy (DME) and ***eriss*** for more than 30 years. These groundwater samples were initially collected to monitor seepage of contaminants from the tailings storage facility (TSF) and the land application areas (LAA) into the surrounding environment. More attention has since been given to the mined-out pits, with pit 1 being used as a tailings repository and pit 3 being prepared to accept tailings commencing in 2015.

Groundwater quality parameters routinely measured include major ions (ERA/DME), heavy metals (ERA/DME/***eriss***) and radionuclides (ERA/***eriss***) (Martin & Akber 1996, 1999, Klessa 2001, ERA 2012). Whereas radium isotopes and activity ratios are used as a process tracer that can provide information about adsorbtion and desorption mechanisms of radium in groundwater, uranium and uranium activity ratios are often used as a source tracer to investigate the potential sources of uranium in a groundwater aquifer (Ivanovich & Harmon 1994, Zielinski et al. 1997). Groundwater samples have been analysed by ***eriss*** from the early 1980s for various research projects (Martin & Akber 1996, 1999). From 1996, a number of bores were routinely sampled and analysed by ***eriss***, but sampling discontinued in 2003. However, longer time series are needed to detect any changes in groundwater quality (Johnston & Milnes 2007), and it was decided that ***eriss*** continue its groundwater monitoring, focussing on the measurement of ^{226}Ra, ^{234}U and ^{238}U activity concentrations in aliquots of bore waters sampled by DME. ARRTC also identified at their 12th meeting in April 2003 that the groundwater pathway required further research, and that groundwater dispersion was one of the Key Knowledge Needs that needed to be addressed from a monitoring perspective, extending well into the closure phase of Ranger mine.

Due to the disconnection and discontinuation of the various programmes involved in groundwater monitoring at Ranger, the information gathered is not used to its full potential, with groundwater quality data still stored in several databases and in various formats. As part of an effort to improve the Ranger groundwater knowledge base, as well as facilitate a more coordinated approach to the acquisition and storage of groundwater data (with the goal to progress the development of closure criteria for Ranger) an internal ***eriss*** IR (Bollhöfer et al. 2015) has now been finalised summarising all ***eriss*** radionuclide and metal data from the late 1980s to 2013.

Methodology

All ***eriss*** groundwater data, including ^{226}Ra, uranium isotopes and metals, have been quality checked and imported into a single *Excel* groundwater spreadsheet. The data have also been migrated into the *EnviroSys* database. UTM WGS84 eastings and northings, bore screen depths, and information about the status of all investigated bores (active, buried etc) have been included in the database. This information has mainly been taken from Ranger uranium mine's Bore Audit Project.

Bores have then been grouped according to catchments, and radionuclide and metal data plotted against sampling date. In addition, a Principal Component Analysis (PCA) has been performed using Minitab 16 for analytes in bores from various catchments, to determine factors governing groundwater chemistry in the bores.

Results

A review of the data available to ***eriss*** showed that time series data are available for some bores going back almost three decades (some to the late 1980s). Data from a total of 346 individual bore water samples have been quality checked and imported into the *Excel* groundwater workbook. Figure 1 shows an aerial photograph of the Ranger mine and the location of bores with time series that exceed 7 years, up to 2012–13. A total of 110 bore water samples were analysed from these ten bores. Analyses included uranium and ^{226}Ra, and metals in most of the bores. Additional information for these bores is given in Table 1.

Figure 1 Map of bores at Ranger with water quality time series exceeding 7 years.

Ahmad and Green (1986) have divided the aquifers in the Ranger region into three main types (rather than into a shallow and deep aquifer only): type A is groundwater in the loose sand and gravels with high permeabilities, type B is groundwater in the weathered profile with relatively low permeabilities and type C groundwater occurs in the relatively fresh fractured rocks in the deep aquifer. This has been modified by Salama & Foley (1997) who proposed a system which recognises zonal influences and host lithologies. The aquifers are classified in their work as alluvial (shallow) (aquifer 1a), weathered rock (upper: aquifer 1b; intermediate: aquifer 2) and deep fractured rock (aquifer 3) (Klessa 2001). The aquifers have been characterised chemically by Salama and Foley (1997) and the classification is reproduced in Table 2. Figure 2 shows a ternary diagram of the major cation concentrations (in percent) in the ten bores shown in Figure 1.

Table 1 Information for bores shown in Figure 1.

BoreID	RN number	Data from-to	Easting	Northing	Screen depth	Site name
OB23	RN022937	11/89 – 9/12	271844	8597241	36 – 51	North of TSF
OB1A	RN022902	11/89 – 9/12	271561	8596865	16 – 31	North of TSF
RN9329	RN009329	9/03 – 9/12	270863	8596371	17 – 19	East of TSF
OB20	RN022934	11/89 – 9/12	271822	8595161	21 – 36	South of TSF
OB21A	RN022935	11/89 – 9/12	272484	8595180	31 – 43	South of TSF
RN23551	RN023551	9/04 – 9/12	272435	8597838	4 – 4.5	RP1/Coonjimba
B11	RN007243	9/06 – 9/09	272126	8599051	0 – 13	RP1/Coonjimba
83_1	RN023010	5/03 – 9/12	274414	8598255	0 – 90	Magela LAA
OB30	RN022941	11/89 – 9/12	273923	8595706	20 – 35	Corridor Ck catchment
OB27	RN022930	9/03 – 9/12	275523	8597063	15 – 40	Corridor Ck catchment

Table 2 Cation facies, anion facies, pH and EC ($\mu S \cdot cm^{-1}$) in the three aquifer classes (Klessa 2001).

	Major cations	Major anions	pH	EC
Superficial deposits and alluvium (aquifer1a)	Ca-Na	HCO3-Cl	5.5	<50
Weathered Nanambu complex (aquifers 1b & 2)	Na-Ca-Mg	HCO3-Cl	6-6.5	<200
Fractured Nanambu complex (aquifer 3)	Ca-Na	HCO3 and HCO3-Cl-SO4	6.6-7.1	180-310

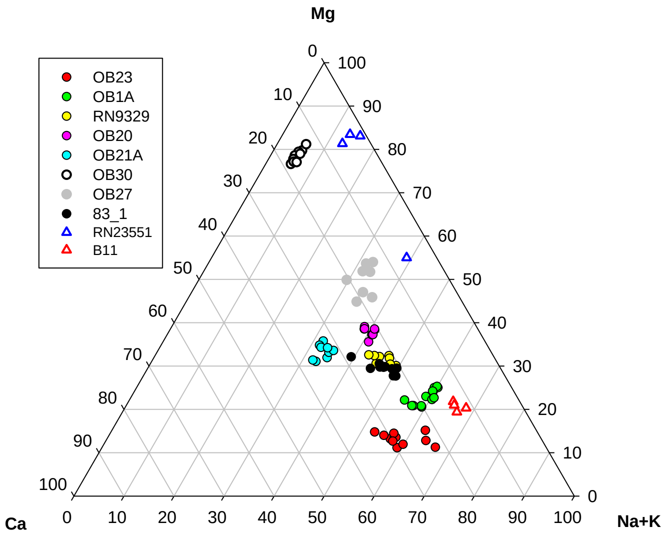
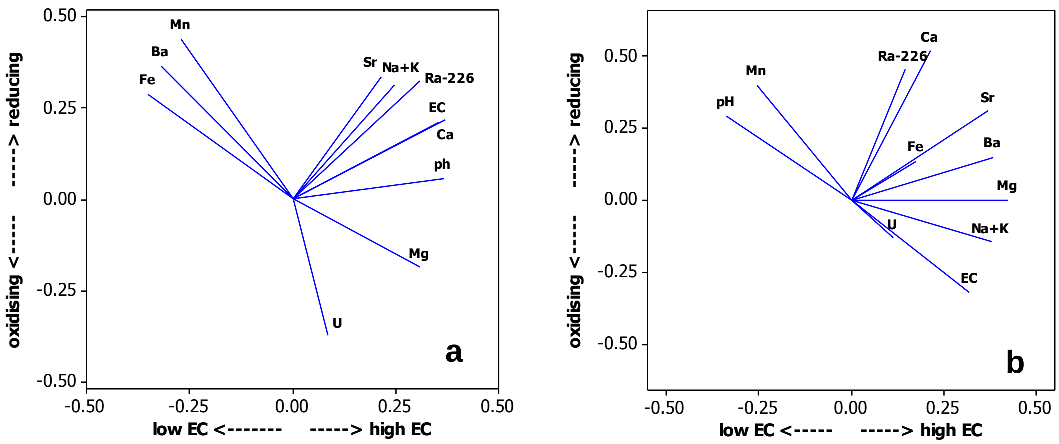


Figure 2 Ternary diagram of major cation concentrations in long time series bores shown in Figure 1.

Bores RN9329, OB20 and OB21A are located south and west of the TSF in the Gulungul groundwater catchment, and they sample aquifers 2 and/or 3. They are characterised by equal Ca, Mg and Na+K. In contrast OB1A and in particular OB23 (sampling deep aquifer 3) are dominated by Na+K and Ca. The PCA (Figures 3a, 4a) shows that groundwater chemistry in the five bores around the TSF is dominated by variations in EC, and bores OB1A and OB23 generally exhibit more reducing conditions compared to the other three bores.



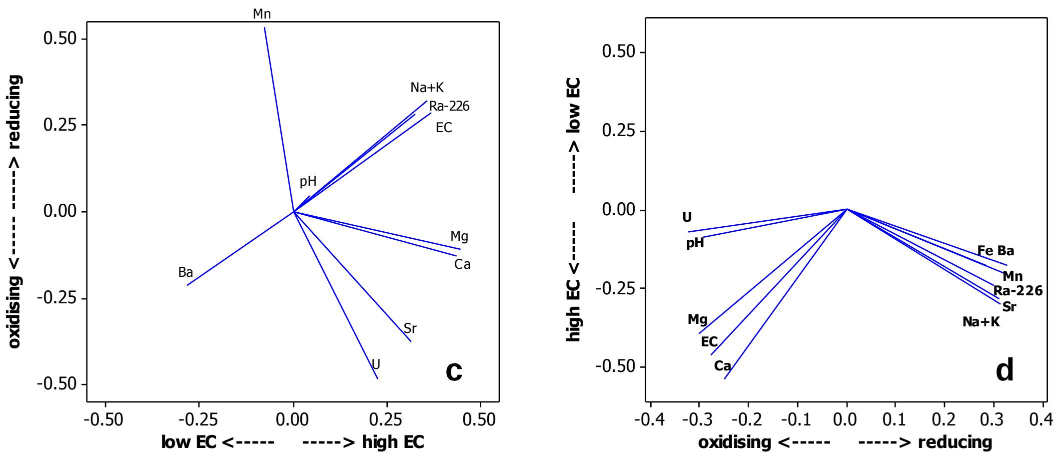


Figure 3 Loading plots of pH, EC, Ba, Ca, Fe, Mg, Mn, Na+K, Sr, U and Ra-226 in bores (a) OB1A, RN9329, OB20, OB21A and OB23 (b) RN23551 and B11 (c) 83_1 and (d) OB30 and OB27.

Bore OB23 is located at the border of the Coonjimba and Gulungul groundwater catchments, close to one of the major fault zones north of the TSF (ERA 2013). Using the uranium activity ratio ($^{234}U/^{238}U$), shown previously (Bollhöfer & Medley 2013), this bore has been identified as being potentially influenced by seepage from the TSF or by other activities on site, such as the construction of the trial landform and several TSF lifts.

Bores RN23551 and B11 are located in the Coonjimba groundwater catchment. RN23551 is located immediately to the North of the TSF along the Coonjimba line (see Figure 1). RN23551 samples aquifer 2 and is dominated by Mg, whereas B11 samples aquifers 1a and 1b and is dominated by Na. Variability in EC appears to be the dominating factor for metal and radionuclide activity concentrations in the two bores (Figure 3b), but there is no discernible trend of increasing metal or radionuclide activity concentrations in the ***eriss*** data.

Bore 83_1 is located in the Djalkmara groundwater catchment close to the Magela LAA, and samples all three aquifers. It is dominated by Na+K (40–50%) and Mg (30%). The bore is located within a sand lens that was previously identified between pit 3 and Magela Creek (ERA 2012). Bore 83_1 is a statutory monitoring bore of ERA. The dominating factor influencing metal and radionuclide activity concentrations in this bore appears to be the EC. ^{226}Ra and EC in this bore are closely correlated but Ba is negatively correlated to EC (Figure 3c). This is potentially due to adsorbtion/desorption mechanisms in the groundwater aquifer and potential barite formation in the groundwater with increasing salinity, as reported by Martin & Akber (1996, 1999) for bores north of the TSF. Redox conditions also have a great influence in particular on U and Mn levels in bore 83_1, and U is generally higher when more oxidising conditions persist.

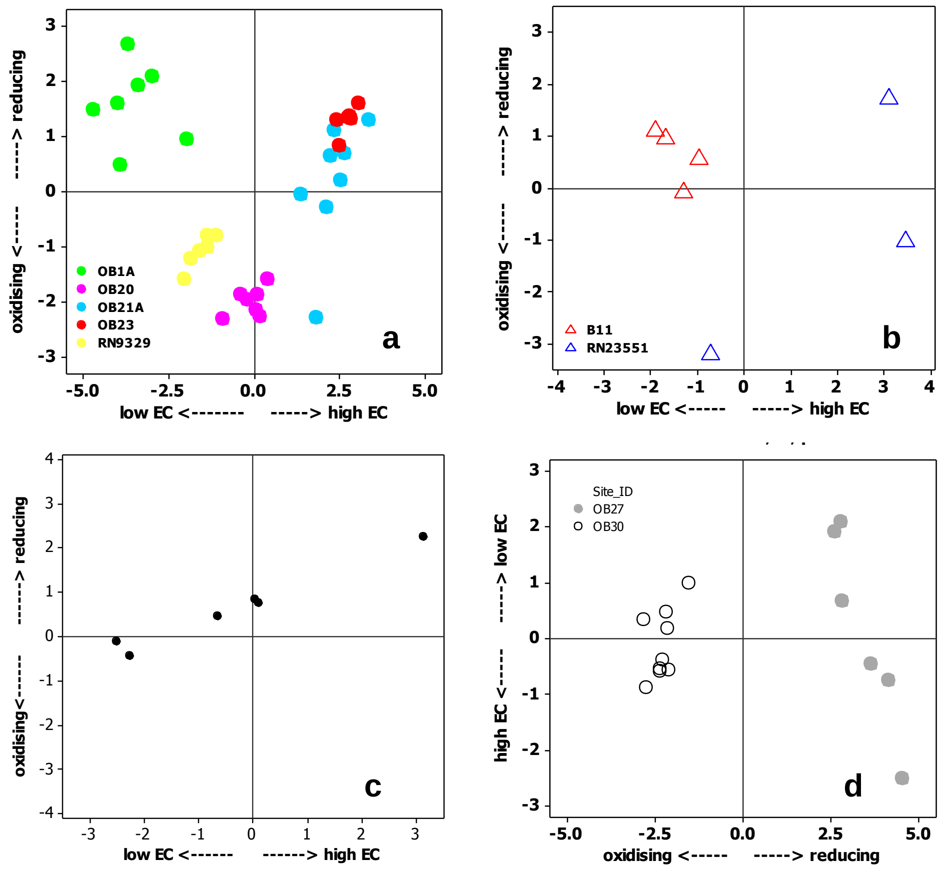


Figure 4 Score plots for bores (a) OB1A, RN9329, OB20, OB21A and OB23 (b) RN23551 and B11 (c) 83_1 and (d) OB30 and OB27.

Groundwater samples from two bores within the Corridor groundwater catchment have been collected over an extended period of time (OB 30: 1989-2012; OB27: 2003-2012). Bore OB30 targets aquifer 3 close to pit 1 and is dominated by Mg, OB27 targets aquifer 2 and 3 on the western side of the confluence of Corridor Creek and Georgetown Creek and is dominated by Mg and Na+K. Metal and radionuclide activity concentrations in these two bores appear to be governed by redox chemistry, which explained almost 80% of the metal variability observed in the two bores, and groundwaters from OB27 are more reducing than those from OB30 (Figures 3d, 4d). No increases of metal or radionuclide activity concentrations over time were observed in these two bores. Uranium concentration however has declined in OB30 over the past decade.

Conclusions

The ***eriss*** internal report (Bollhöfer et al. 2015) provides information about ***eriss*** data availability and type as well as general correlations and water chemistry of bore waters analysed by ***eriss***. It is important to note that it describes only one relatively small existing dataset, much larger databases have been developed by ERA and DME, who sample a large number of bores quarterly and biannually, respectively. The datasets from the three organisations should be combined and investigated in a groundwater GIS, taking into account QA/QC procedures which are in place for populating the various databases.

There is also the need to compare the results from water chemistry investigations for groundwaters in the vicinity of Ranger to the distribution of faults, hydraulic conductivities, the aquifers sampled, lithology, soil type or distance from potential seepage locations. It is recommended to include variables such as sampled aquifer, groundwater catchment and lithology in a PCA of all available data, to strengthen outputs and information that can be gleaned from the groundwater monitoring results. The data can then be used to test hydrodynamic models and assist in forward modelling of long-term closure impacts onto groundwater quality. Understanding the connectivity between pit 1 and Corridor Creek (and the effectiveness of the seepage barrier) and characterising the existing sand lens between pit 3 and Magela Creek will be important in any forward modelling of groundwater quality after closure.

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Atmospheric radioactivity monitoring in the vicinity of the Ranger mine

C Doering, A Bollhöfer & M Fromholtz

Introduction

Uranium mining has the potential to release radon (a radioactive noble gas) and dust-bound radionuclides to the atmosphere at levels above the natural background through ground disturbance and other activities. The inhalation of radon progeny in air and long-lived alpha activity (LLAA) radionuclides contained in or on dust can contribute to the above background radiation dose received by the public in the vicinity of a uranium mine.

National radiation protection standards (ARPANSA 2005) require that the annual radiation dose received by a member of the public from practices such as uranium mining and milling must not exceed 1 millisievert (mSv) above the natural background. This dose limit applies to the sum of doses received via all exposure pathways that are traceable to the practice.

The Ranger mine is currently the only operating uranium mine in the ARR and the main potential source of above background radiation dose to local communities. The main areas of permanent habitation in the vicinity of the mine are Jabiru town and Mudginberri community. ***eriss*** maintains atmospheric monitoring stations to measure radon progeny and dust-bound LLAA radionuclide concentrations at the Jabiru Water Tower (Jabiru town) and Four Gates Road (Mudginberri). The purpose of these measurements is twofold: (i) to provide an independent check of the values measured and reported by ERA; and (ii) to provide assurance that any dose to the public associated with mine-related radioactivity in air is low and does not pose any unacceptable radiation risk.

Method

Environmental radon daughter monitors manufactured by Radiation Detection Systems in Adelaide were used for continuous radon progeny monitoring. The monitors operated at a flow rate of 0.35 l/min drawing air through a Whatman GF/C filter positioned above an alpha counter. Hourly potential alpha energy concentration (PAEC) data were logged in the internal memory of the monitors, which were downloaded at approximately fortnightly intervals.

EcoTech MicroVol-1100 low flow-rate (~3 l/min) air samplers fitted with Whatman GF/C filters were used for dust sampling. Filters were changed at approximately fortnightly intervals and analysed in ***eriss*** laboratories for total alpha activity using Daybreak 582 alpha counters. Count times were typically three to four days to ensure reasonable counting statistics were achieved. Measurement of the background alpha activity of the counting system was made prior to analysis of each filter. The background count rate was subtracted from the filter count rate to determine the net count rate. A callibration factor for counter efficiency was then applied to determine the alpha activity on the filter.

Results

Radon progeny

Figures 1 and 2 show hourly and quarterly average radon progeny PAEC monitoring data from Jabiru town and Mudginberri, respectively, for the 2013 calendar year. Gaps in the data are due to instrument maintenance and data quality issues.

The spikiness in the hourly PAEC data reflects the normal diurnal pattern in radon progeny concentrations in surface air. Higher concentrations typically occur in the early morning around sunrise when atmospheric conditions tend to be most stable. Thereafter the surface air becomes mixed by convection (solar heating) and advection (wind), which disperses the radon progeny into a larger atmospheric volume.

The quarterly average PAEC results show the typical wet-dry seasonal trend, with higher concentrations occurring in the second and third quarter of the year (dry season) and lower concentrations occurring in the first and fourth quarter of the year (wet season). The effect of rainfall is to suppress radon exhalation from the soil surface and thus decrease the radon progeny PAEC in air.

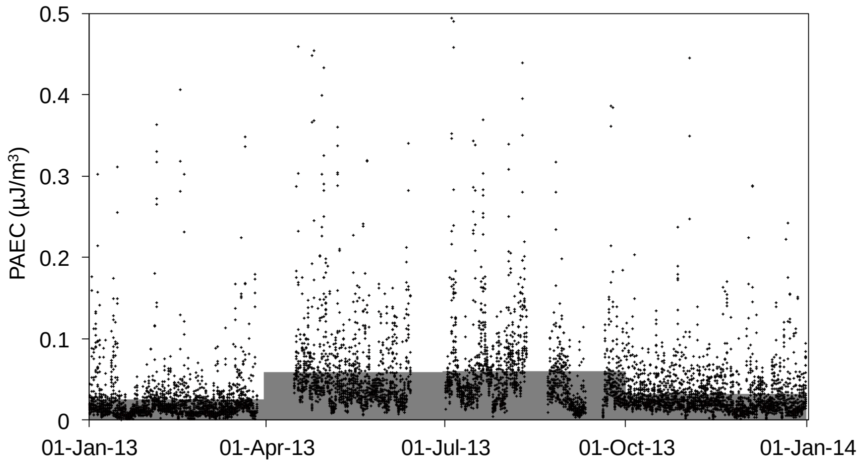

Figure 1 Hourly (black crosses) and quarterly average (grey columns) radon progeny PAEC in air at Jabiru town in 2013.

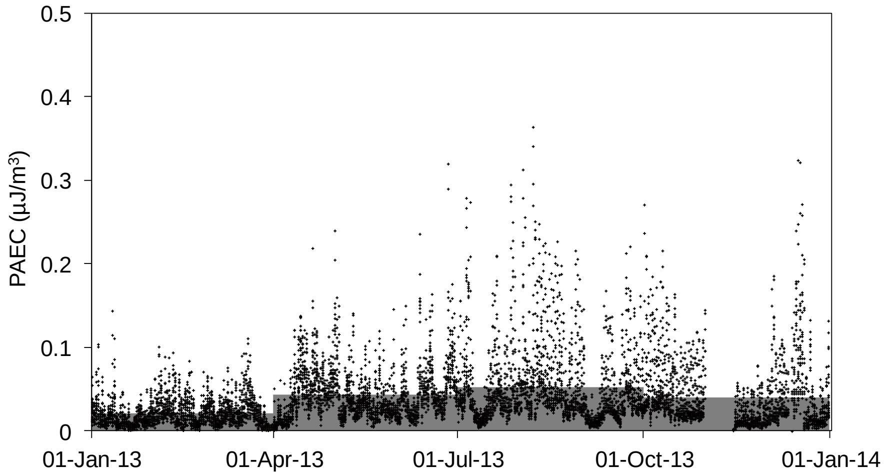

Figure 2 Hourly (black crosses) and quarterly average (grey columns) radon progeny PAEC in air at Four Gates Road near Mudginberri community in 2013.

Table 1 provides a summary of annual average radon progeny PAEC in air and estimated doses to the public, as well as comparison with values reported by ERA for Jabiru town. The total annual effective dose from radon progeny in air, which includes contribution from natural background, has been estimated to be 0.405 mSv at Jabiru town and 0.382 mSv at Mudginberri. This total annual dose has been calculated from the product of the annual average radon progeny PAEC in air, the radon progeny dose conversion factor of 0.0011 mSv per $\mu J \cdot h/m^3$ recommended by the International Commission on Radiological Protection (ICRP) (ICRP 1993) and the assumed full year occupancy of 8760 hours.

Table 1 Radon progeny PAEC in air and estimated doses to the public at Jabiru town and Mudginberri in 2013*

	Jabiru town	Mudginberri
Annual average PAEC ($\mu J/m^3$)	0.042 (0.045)	0.040
Total annual dose (mSv)	0.405 (0.530)	0.382
Mine-related dose** (mSv)	0.055 (0.031)	0.002

* Values in brackets refer to data from the ERA Radiation Protection and Atmospheric Monitoring Programme Report for the Year Ending 31 December 2013.

** The radon progeny PAEC difference used in the SSD mine-derived dose calculation was 0.024 $\mu J/m^3$ for Jabiru town and 0.009 $\mu J/m^3$ for Mudginberri.

The mine-related annual dose from radon progeny in air has been estimated to be 0.055 mSv at Jabiru town and 0.002 mSv at Mudginberri. This dose is dependent on wind direction and has been calculated from the difference in average radon progeny PAEC in air when the wind was from the direction of the mine and when the wind was from directions other than the mine, then multiplying this difference with the ICRP radon progeny dose conversion factor and the number of hours that the wind was from the direction of the mine. Hourly wind direction data for 2013 were obtained from the Bureau of Meteorology weather station at Jabiru Airport. Analysis of these data (Figure 3) suggests that the wind was from the direction of the mine for 2029 hours during the year at Jabiru town (90–110 degree sector) and 210 hours during the year at Mudginberri (140–160 degree sector).

Differences between the ***eriss*** and ERA radon progeny PAEC results and public dose estimates for Jabiru town are most likely due to differences in monitoring regime. Whereas ***eriss*** aims to monitor continuous hourly radon progeny PAEC in air over the full year, the ERA regime is based on a minimum requirement of one week per month continuous monitoring.

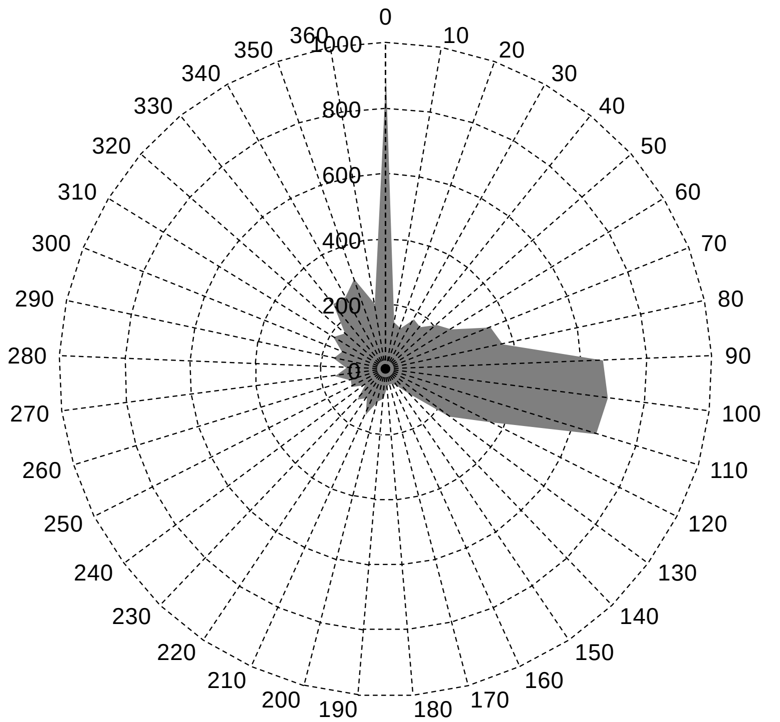

Figure 3 Histogram of hourly wind direction data at Jabiru airport for 2013. A wind direction of 0 indicates still conditions (i.e. no wind).

Dust-bound LLAA concentrations

Figures 4 and 5 show measured and quarterly average concentrations of dust-bound LLAA radionuclides in air at Jabiru town and near Mudginberri community, respectively, for 2013. Gaps in the data are due to instrument maintenance and data quality issues.

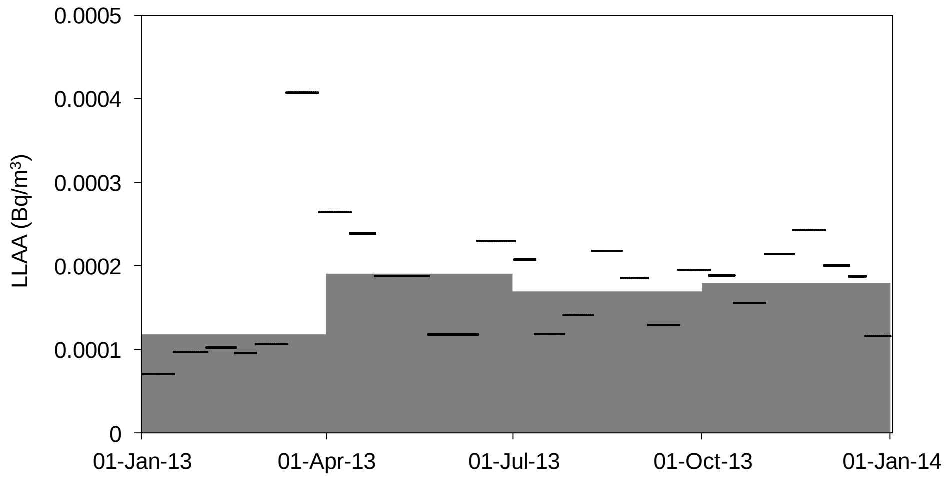

Figure 4 Measured (black lines) and quarterly average (grey columns) concentrations of dust-bound LLAA radionuclides in air at Jabiru town in 2013.

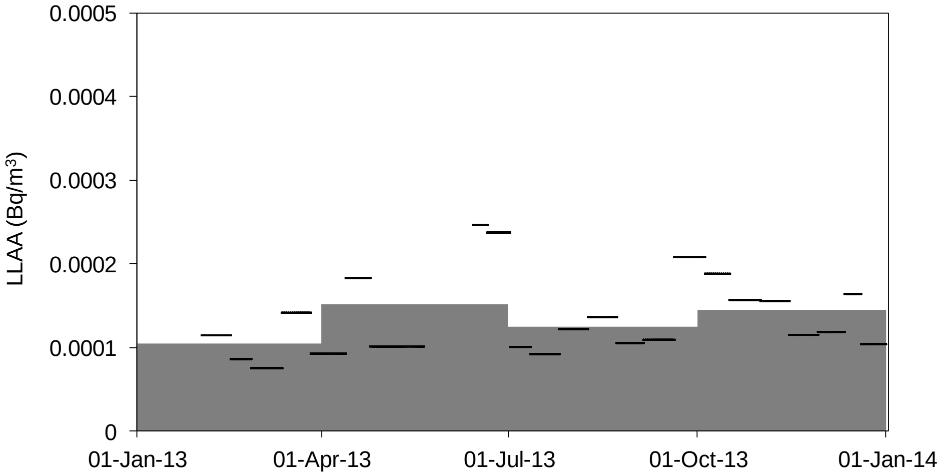

Figure 5 Measured (black lines) and quarterly average (grey columns) concentrations of dust-bound LLAA radionuclides in air at Four Gates Road radon station near the Mudginberri community in 2013.

Table 2 provides a summary of the annual average LLAA radionuclide concentration and estimated total and mine-related doses to the public at Jabiru town and Mudginberri. The total annual effective dose from dust-bound LLAA radionuclides, which includes contribution from natural background, has been estimated to be 0.007 mSv at Jabiru town and 0.005 mSv at Mudginberri. This total annual dose has been estimated by calculating the time weighted annual average LLAA concentration from the individual samples and then multiplying with a dose conversion factor of 0.0061 mSv/Bq_{α}, breathing rate of 0.75 m^3/h and assumed full year occupancy of 8760 hours.

Table 2 LLAA radionuclide concentrations in air and estimated doses to the public at Jabiru town and Mudginberri in 2013.

	Jabiru town	**Mudginberri**
Annual average LLAA (Bqα/m^3)	1.7×10^{-4}	1.3×10^{-4}
Total annual dose (mSv)	0.007	0.005
Mine-related dose* (mSv)	1×10^{-3}	3×10^{-5}

* Calculated from the assumption that the ratio of mine-related to total annual dose from dust is the same as that for radon progeny.

The mine-related dose from dust-bound LLAA radionuclides has been estimated to be 0.001 mSv or less at both Jabiru town and Mudginberri. This dose has been calculated by assuming that the ratio of mine-related to total annual dose from dust is the same as that for radon progeny. This assumption is likely to result in an overestimate of the mine-related dose via the dust inhalation pathway. This is because dust in air should settle out much quicker as a function of distance from the mine compared with gaseous radon, meaning that the mine-related to total dose ratio for dust should be less than that for radon progeny.

Conclusions

The ***eriss*** atmospheric radioactivity monitoring results for 2013 indicate that potential inhalation doses to the public from mine-related sources of radon progeny and dust-bound radionuclides in air are low and do not pose any unacceptable radiation risk.

The total annual effective dose from radon progeny in air, which includes contribution from natural background, was estimated to be 0.405 mSv at Jabiru town and 0.382 mSv at Mudginberri. The mine-related component of this dose was only 0.055 mSv at Jabiru town and 0.002 mSv at Mudginberri.

The total annual effective dose from dust-bound LLAA radionuclides in air, which includes contribution from natural background, was estimated to be 0.007 mSv at Jabiru town and 0.005 mSv at Mudginberri. The mine-related component of this dose was 0.001 mSv or less at both Jabiru town and Mudginberri.

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Environmental factors associated with toxicity monitoring in Magela and Gulungul Creeks

C Humphrey & M Ellis

Influences of different snail culturing conditions on the snail egg production response

Background

The influence of culturing conditions under which snails are reared for toxicity monitoring (Humphrey et al. 2015), is a potential source of variability in egg production during wet season test exposures in the creeks. This was described previously by Humphrey & Ellis (2014). Since the 2011–12 wet season, snail stocks for test exposures have been sourced from both shallow static shallow water containers and deep (~1.1 m) containers with a non-static Recirculating Aquaculture System (RAS). For the 2012–13 wet season testing, egg production was compared under routine creek testing conditions between snails cultured under the two culture-water regimes. 'Age' of the snail cohort, as measured by length of time from initial container seeding with egg masses to use of the snails in a toxicity monitoring test, was also examined. However continuous growth and recruitment of snails from the progeny of ensuing generations of snails held in each container was not strictly controlled and so this aspect could not be properly assessed.

Container type (shallow/deep) was found to be a highly significant source of variation in snail egg counts in the tests: snails sourced from shallow static containers produced more eggs than those from the deep (RAS) containers ($P = 0.0001$). No ready explanation for this difference was provided at the time. However the more frequent dilution of RAS waters with reverse osmosis-filtered and tap (bore) waters may be removing nutrients and potential for algal food production in these containers.

It was not always possible to attribute an egg count from each snail pair to a specific treatment type in the 2012–13 data analyses. This is because snails were not necessarily sourced from the same culture-water regime when setting up replicate pairs of snails placed in each egg-laying chamber.

Methods

With improvements to the design used in the previous wet season, the effect of snail culturing treatments was further examined in the 2013–14 wet. Two broad treatment classes were adopted:

(i) Deep recirculating aquaculture systems (RAS) (2500 L) versus shallow static containers (1000 L), as used in previous years; and
(ii) Snail stocks established at the start of the wet season and self-replenishing throughout the ensuing wet season ('established') versus stocks re-established as new 'cohorts' at regular intervals. Established stocks developed mixed age classes over time, the oldest snails of which could span the entire wet season (230 days), while 'cohort' stocks contained snails between 53 and 112 days of age at the time of testing (Figure 1). Thus cohort snails were of known, and on average younger, age compared

to established snails. Four consecutive cohorts were established for wet season testing. Upper numeric values in the 2nd horizontal axis of Figure 1 delineate, and indicate age of, cohorts used for testing.

It was possible to further compare egg production for snails held in shallow containers by positioning these either under a broad open awning ('outside') or inside a walled-in portion of the same facility ('inside'). Shallow containers held 'inside' were subject to darker day-time conditions (albeit unquantified) while the water temperature outside reached slightly higher day-time but slightly lower night-time water temperatures (± 0.5°C) (data not shown). Differences in egg production between these treatments could thereby indicate light and/or water temperature effects upon egg production.

A total of six treatments were tested over the 2013–14 wet season:

1.	RAS - Deep	Outside	Established at start
2.	RAS - Deep	Outside	Cohorts
3.	Static shallow	Outside	Established at start
4.	Static shallow	Outside	Cohorts
5.	Static shallow	Inside	Established at start
6.	Static shallow	Inside	Cohorts

For each toxicity monitoring test and each creek and site (upstream/downstream), replicate snail pairs were selected for testing in equal numbers from across the six treatments described above (i.e. 3 snail pairs per treatment, per site). Thus, and unlike the 2012–13 wet season, each snail pair was sourced from the same treatment. The total egg count arising from each replicate snail pair was determined, with mean values for all replicate pairs from each of the six treatments plotted for each test in Figure 1.

Analysis of variance (ANOVA) testing was used to examine egg number differences amongst the six treatments (see above) for creeks combined and for separate creeks (from Figure 1). Container age varied with each test and so this factor was unevenly distributed amongst container types and could only be nested within this factor. In the previous year's analysis (Humphrey & Ellis 2014), snail size, snail weight, site (up/downstream) and test order were additional covariates or factors included in the analysis. For reasons outlined by Humphrey & Ellis (2014) these covariates and factors were not considered in the current ANOVA.

Results

ANOVA testing showed:

1. Snail egg number differed significantly amongst the three container-type treatments for creeks combined and creeks separate (P = 0.001). All (three) pairwise comparisons (including shallow inside versus shallow outside) showed highly significant differences for creeks combined and creeks separate (P = 0.001). Greater egg production followed the order shallow outside > shallow inside > deep (RAS), evident in the plotted data shown in Figure 1. This result supported the 2012–13 wet season analysis (Humphrey & Ellis 2014) which showed, similarly, greater egg production for snail pairs reared in shallow static containers compared to deep RAS containers.

2. Egg number differed significantly between established and cohort treatments for creeks combined and creeks separate ($P < 0.01$). Cohort snails produced significantly greater numbers of eggs compared to established snails and this was most evident in the shallow outside treatment (Figure 1).
3. Actual container age (or maximum age for 'established' treatment), nested within the container age factor, was significant for each creek examined separately ($P = 0.001$). When the 'established' treatment was removed from analysis, with container age for just the 'cohort' treatment now nested within container type, the factor held a similar significance for each creek ($P = 0.001$). Examining Figure 1, it is evident that the egg production for cohort snail age 77 days in both the 2nd and 3rd cohorts, was particularly high, probably explaining the significance of this factor. While fecundity increases generally with snail size (Supervising Scientist Annual Report for 2012–2013; Section 4.4), it is also possible that fecundity has an interaction with snail age with greatest egg production observed at an intermediate age of snails (~77 days). This requires further investigation.

The results of the 2013–14 husbandry investigation have practical implications that guide future snail culturing for routine toxicity monitoring in Magela and Gulungul creeks. Firstly, culturing snails in regular cohorts of known snail age appears to result in greater reproductive vigour. Moreover, the more resource-intensive recirculating aquaculture systems is not providing obvious benefit in terms of snail egg production and could be re-considered in terms of future application. Finally, the results highlight the importance of allocating snails from a particular culturing treatment evenly to each of the paired sites in a toxicity monitoring test. If this is not done, the upstream-downstream difference value for the test may reflect an artefact of culture type, as opposed to its proper representation of detecting changes in water quality.

Influences of ambient water quality on the snail egg production response

As reported elsewhere in this Summary (Humphrey et al. 2015) , water temperature and electrical conductivity (EC) influence the snail egg laying response in Magela and Gulungul creeks. Continuous water quality data gathered from sondes at each of the monitoring sites since the 2006–07 wet season has enabled accurate and effective integration of water quality exposure conditions for each of the four-day toxicity monitoring tests. Measures of water temperature and EC used in data analyses represent the median of 10 minute continuous readings taken across the four-day exposure period at each of the creek sites. The collective water quality and egg production data gathered since the 2006–07 wet season are shown in Figure 2. An interacting effect between water temperature and EC has been observed, as EC increases (generally across the range ~7–30 μScm^{-1}), snail egg production:

- increases at lower water temperature ranges (27–30°C), and
- decreases (i.e. negative effect) at higher water temperatures (> 30°C) (see Figure 2).

The increase in egg production with increasing EC at lower water temperature ranges (<30°C) is highly significant and this relationship has generally strengthened in the past several years (cumulative results not provided). The decrease in egg production with increasing EC at higher water temperature ranges (30°C) is a weaker relationship that has fluctuated between significance (P < 0.05) and non-significance after each wet season of accruing data gathered since 2011 (data also not provided).

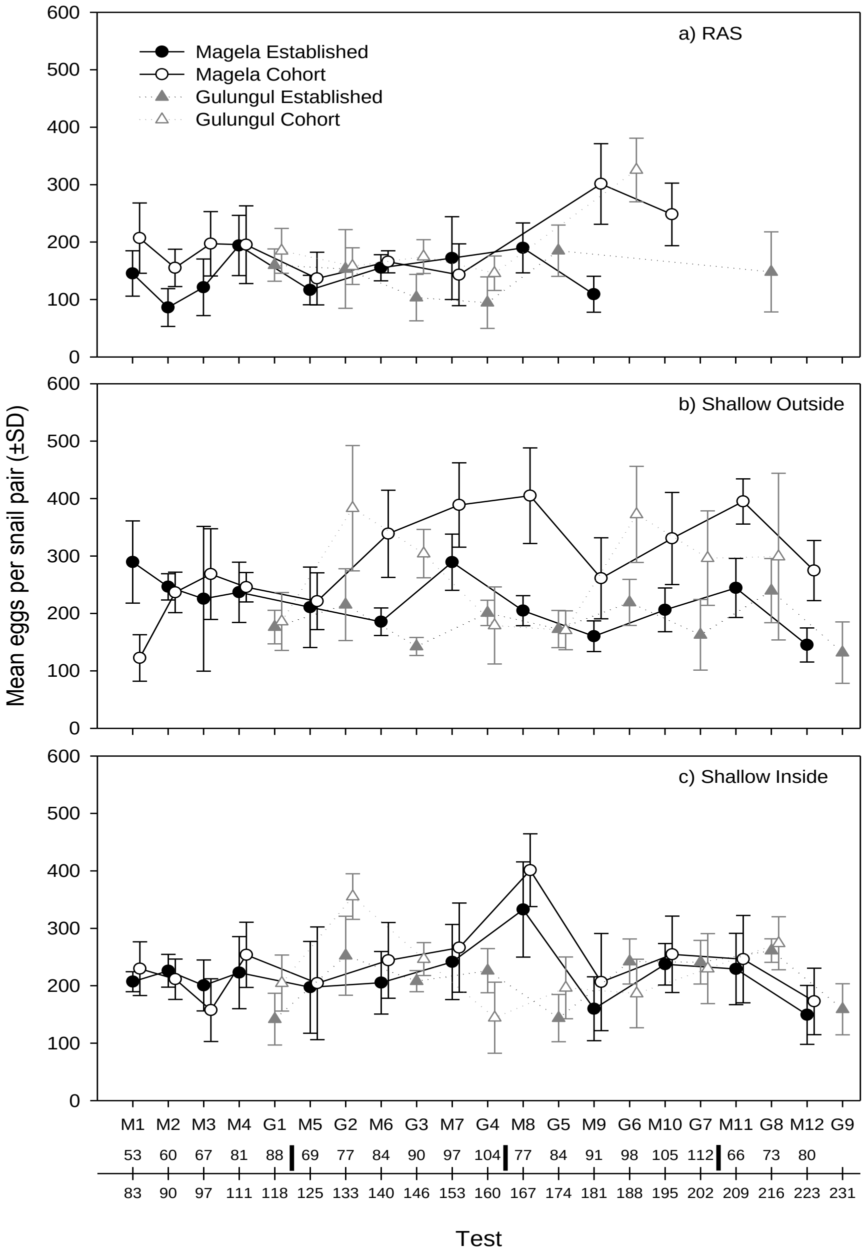

Figure 1 Mean egg count for replicate snail pairs (*n*=3) sourced from shallow static and deep RAS culture container types, and according to culture age (established and cohorts), for Magela and Gulungul toxicity monitoring tests conducted in the 2013–14 wet season. Horizontal axis shows the test order over the wet season, with M and G referring to Magela and Gulungul Creeks, respectively. Upper and lower numeric values in the 2nd horizontal axis indicate age of cohorts and time since establishment, for 'culture age' treatments, respectively.

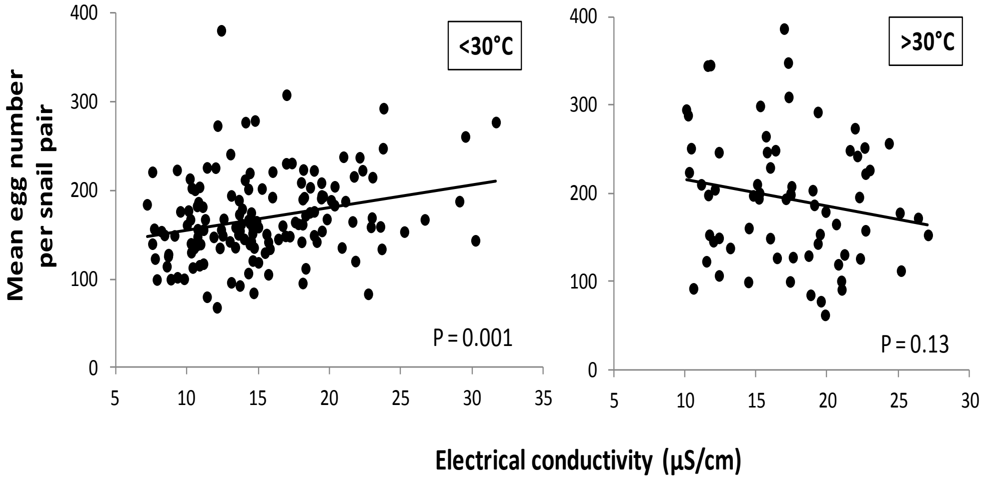

Figure 2 Relationships between mean snail egg number for each site in Magela and Gulungul Creeks, and ambient electrical conductivity and water temperature over the four-day exposure test periods for wet seasons between 2006–07 and 2013–14.

These water quality/egg production relationships as depicted in Figure 2, can usefully be applied to interpreting annual toxicity monitoring results in Magela and Gulungul creeks (see Humphrey et al. (2015) elsewhere in this Summary). The relationships have limited use in explaining the magnitude of egg production at a site, but can generally explain the differences in egg production between the paired upstream and downstream sites.

Of the 100 tests conducted from 2006–07 to the present, the difference in egg number between upstream and downstream sites can be successfully predicted in more than 75% of tests for median water temperature <30°C and in more than 65% of tests for median water temperature >30°C. A (4-day) downstream median EC in Magela and Gulungul creeks greater than 20 µS cm^{-1}, represents a value typically associated with mine waste water discharges. Of the 100 tests conducted since 2006–07 to the present:

- Higher downstream egg production was observed in 15% of the (over 100) tests where median water temperature <30ºC and downstream EC values >20 µS cm^{-1} were observed; and
- Lower downstream egg production was observed in 9% of the (over 100) tests where median water temperature >30ºC and downstream EC values >20 µS cm-1 were observed. Seventy-five percent of these (high water temperature) tests were conducted in Magela Creek.

At this stage these mine-related 'effects', representing enhancement in snail reproductive activity in most cases, are not regarded as constituting environmental concern.

Conclusions

Toxicity monitoring results for the 2013–14 wet season supported the findings from the previous (2012–13) wet season that snail culturing conditions have a significant influence upon the egg laying response in ensuing toxicity monitoring tests. For toxicity monitoring tests conducted in 2012–2013, Humphrey & Ellis (2014) suggested that variation in snail egg counts was mostly ambient water quality-related compared to culture-related. This was based upon the proportion of the variation associated with treatments/factors from the ANOVA testing (described above). For the 2012–13 wet season, variation in snail egg counts accounted for by culture conditions was attributed to the factor container type and ambient creek water quality to test order. For these, variation associated with ambient water quality was about 2.5 times greater than culture-type variation.

For the 2013–14 wet season, factors associated with culture conditions included container type and container age. When variation within the ANOVA for these factors is combined and this value is compared to the variation associated with test order (data not provided here), variation associated with 'ambient water quality' is about two times greater than culture-type variation for Gulungul Creek, but is the same order of variation and contribution as ambient water quality for Magela Creek.

Such comparative assessments (culturing versus water quality) need to be considered carefully, because egg production measured and associated with the different culturing treatments is itself, a reflection of creek water exposures. Hence, culturing responses are not independent of the effects of exposure to creek water quality. The best evidence that creek water quality is the more important contributor to the egg laying response is the fact that the pattern of response in Magela and Gulungul creeks is each relatively independent of one another over the wet season for each of the culturing treatments (Figure 1). An improved understanding of the relative influences of culturing versus water quality upon egg production would arise from a laboratory control, where water quality is held constant amongst consecutive tests, run in parallel with the toxicity monitoring tests.

If culturing conditions are contributing to the overall magnitude of egg production in the creeks over the wet season, then snails sourced from the 'outside' static shallow containers in both the 2012–13 wet season (Humphrey & Ellis 2014, Fig 1) and 2013–14 wet season (Figure 1) may be responsible for the high egg production observed in both seasons (see Humphrey et al. 2015 elsewhere in this Summary). Lower overall egg production in the 2013–14 wet season compared to the previous wet season may be a consequence of the comparatively lower egg production in snails sourced from the deep RAS containers in this wet season (Figure 1) compared to the previous wet season (Figure 1, Humphrey & Ellis 2014).

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Part 2: Ranger – Rehabilitation

Key Knowledge Need 2.2: Landform design

Sediment losses from the trial landform

M Saynor, W Erskine[1,2] J Boyden, R Houghton & K Turner

Introduction

With the current mine lease at Energy Resources Australia Pty Ltd's (ERA) Ranger mine due to expire in 2021, there is an increasing focus on key aspects of progressive rehabilitation on the mine-site. ERA is establishing a mine closure plan with a key focus on reshaping and revegetating the final mine landform so that it may be handed back to the traditional owners for incorporation into Kakadu National Park. In order to meet legislated Environmental Requirements for closure, the final landform must resemble the surrounding landscape, be radiologically stable, exhibit erosion characteristics similar to the surrounding environment, and act as a functional containment structure for the mine tailings, which must be physically isolated from the environment for 10,000 years post-closure.

A further challenge for rehabilitation will be to ensure that the Ranger Project Area does not become a significant future source of elevated sediments and solutes to surrounding areas. Pits 1 and 3 are designated containment areas for mine tailings as well as the brine residue from the brine concentrator. While the final landform will include various engineered solutions to prevent expression of contaminated waters from Pit 3, the potential for mobilisation of sediments and solutes via surface water runoff and seepage through the landform remains. Further work is required to understand the processes associated with sediment and solute generation and transport.

eriss and ERA are collaboratively undertaking research to determine the optimal design of the final, rehabilitated landform for the Ranger mine. This work includes measurement of solute and sediment loads generated and transported from the landform during rainfall events (Saynor et al. 2011). These data are also used for validation of predictive computer modelling of the long-term geomorphic behaviour of the proposed landform designs for the Ranger mine, also carried out by ***eriss*** (Lowry et al. 2013).

A trial landform (TLF) of approximately 8 ha was constructed in late 2008 and early 2009 adjacent to the north-western wall of the tailings storage facility (TSF) at Ranger mine (Figure 1). The trial landform was designed to assess:

1. two types of potential capping material: (i) waste rock, and (ii) waste rock blended with approximately 30% fine-grained weathered horizon material (lateritic material)
2. two types of potential planting methods: i) direct seeding and ii) tube-stock.

The TLF was segmented into four main treatment areas and the surface was ripped on the contour before planting was carried out (Saynor et al. 2009).

Since construction, measurements have been carried out to assess the generation and transport of sediments and solutes, and ability to achieve and sustain growth of plant

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species native to the region. Updates of the generation and transport of solutes, plus hydrology and bedload yields, have been reported annually (Saynor et al. 2011, Saynor et al. 2012, Saynor & Erskine 2013, Saynor et al. 2013). This report provides the update of the 2013–2014 wet season.

Methods

Erosion plots were installed during the 2009 dry season (Saynor et al. 2009) on each of the four main treatment areas by physically isolating approximately 30 x 30 m area (Figure 1) from the surrounding landform surface by raised damp course and concrete borders on three sides and an open PVC drain on the down-slope side. Plots 1 and 4 were planted with tube-stock in March 2009 with infill planting to replace dead specimens in January 2010 (Daws & Gellert 2011). Plots 2 and 3 were direct seeded in July 2009. However, because of poor germination, these plots were infill planted with tube stock in January 2011 (Gellert 2012). For 2013 vegetation results, tubestock has a superior performance over direct seeding for plant height, density, and diversity, with the survival of the tubestock higher on the waste rock than on the laterite mix (Gellert 2014).

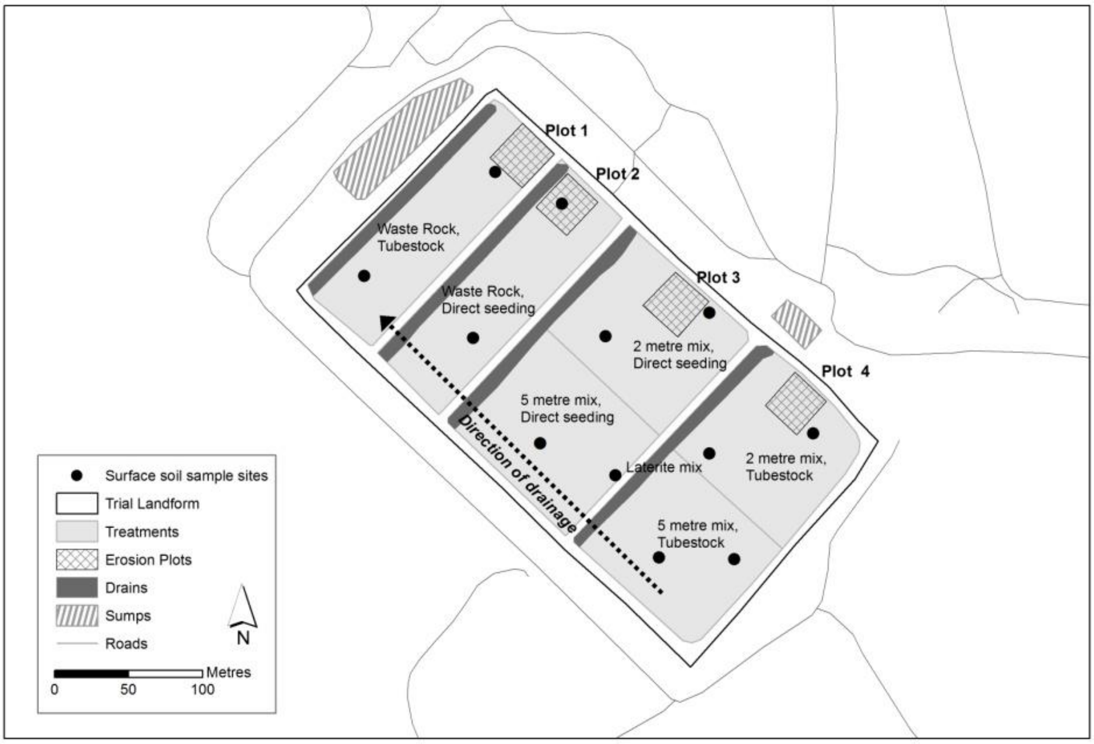

Figure 1 Layout of the erosion plots on the trial landform.

Each plot was instrumented with a range of sensors that were described in detail in Saynor et al. (2012). In summary, these included: rectangular broad-crested (RBC) flume to accurately determine discharge; a tipping bucket rain gauge, a primary shaft encoder with a secondary pressure transducer to measure stage height; a turbidity probe from which suspended sediment concentration could be determined; electrical conductivity (EC) probes located at the inlet to the stilling basin and at the entry to the flume to provide a measure of the concentration of dissolved solutes in the runoff; an automatic pump sampler to collect event based water samples triggered by predetermined changes in EC and turbidity readings; and a data logger with mobile phone telemetry connection (Figure 2).

The samples triggered by EC changes were analysed in the laboratory for general water quality (EC and pH) as well as concentrations of dissolved (<0.45 μm) trace metals (uranium, manganese, aluminium, iron, zinc, copper, barium, nickel, silica and lead) and major ions (magnesium, sodium, potassium, calcium, chloride, sulfate). These analyses were undertaken for the wet seasons 2009–10 to 2012–13. Samples were collected during the 2013–14 wet season but were not sent for analysis. The suspended sediment concentration was measured by filtering a standard volume of sample through a 0.45 μm filter of known weight and then determining the oven dry weight of the sediment.

Bedload samples were collected at weekly to monthly intervals during each wet season, depending on the magnitude of runoff events and staff availability. The samples were processed in the laboratory by weighing (after oven drying) as well as measuring the particle size distribution using the Wentworth size fractions of gravel (> 2 mm), sand (63 μm to 2 mm) and silt and clay (< 63 μm). Each sample was sieved to determine the sediment fractions.

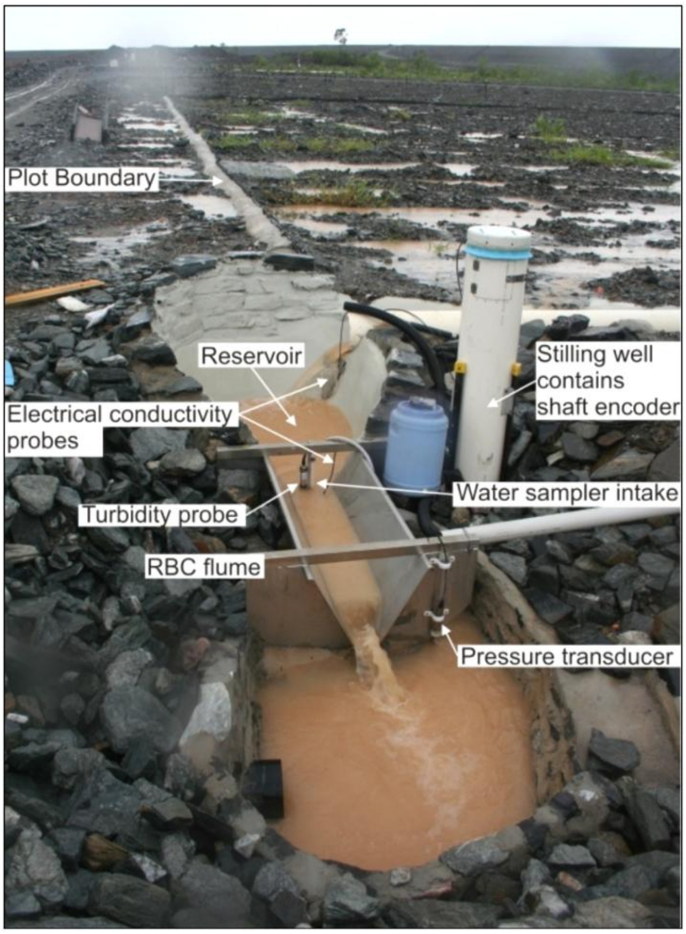

Figure 2 Runoff through the flume on the trial landform erosion plot 3 during a storm event.

Rainfall and runoff results and discussion

Overview

Data are presented for a 'water-year', from September to August. Preliminary sediment and solute losses from the four erosion plots were presented for the first wet season of monitoring, in Saynor et al. (2011). Rainfall for all four plots and runoff from erosion Plots 1 & 2 are reported here for five wet seasons. We are currently investigating some issues with the runoff data from the plots (that mainly affect the larger runoff events), but have reported the current results here. Resolution of the issues will be reported later and will result in a slight change in the volume of water for each event but will not impact on the number of events.

Rainfall and runoff results and discussion

Mean annual rainfall at Jabiru Airport (Station No. 014198, located 2.3 km from the trial landform) is 1584 mm (Bureau of Meteorology 2014). The 2013–14 wet season was above average, having the second highest rainfall for the five years. The annual rainfall for the 2012–13 water year on the trial landform was the lowest for the five years of study with 1274 mm (Table 1) and was 19.6% lower than the mean annual rainfall at Jabiru airport.

Table 1 Rainfall data for the four erosion plots on the trial landform for the five years of measurement.

Water year	Erosion Plot 1 Rainfall (mm)	Erosion Plot 2 Rainfall (mm)	Erosion Plot 3 Rainfall (mm)	Erosion Plot 4 Rainfall (mm)	Mean Annual Rainfall ± Standard Error (mm)
2009–10	1533	1531	1480	1528	1518 ± 13
2010–11	2227	2290	2205	2296	2255 ± 23
2011–12	1508	1531	1456	1489	1496 ± 16
2012–13	1283	1274	1260	1264	1274 ± 5
2013-14	1961	1962	1950	1991	1966 ± 5

Surface runoff from the erosion plots occurs as a number of discrete events. Some of the smaller rainfall events result in water flow off the plots into the reservoir upstream of the flume (Figure 2), without water flow over the flume. The runoff data for plots 1 and 2 are shown in different tables because there are subtle differences in plot area, and the topography created by the rips lines is different, creating slightly different runoff characteristics.

For the discussion below only those flows over the flume have been included

The lowest number of discrete runoff events for both plots for the five wet seasons was for 2012–13. The number of runoff events that produced discharge over the crest of the flume was lowest in the driest water year (2012–13), with 92 events on plot 1 and 114 events on plot 2, and was greatest in the wettest water year (2010–11), with 213 events on plot 1 and 221 events on plot 2. Unusually, annual runoff was lowest in the 2009–10 water-year (rainfall 5.2% below average) than in both the drier 2011–12 water-year (rainfall 5.6% below average) and the much drier 2012–13 water-year (rainfall 19.6% below average) (Table 2). The lowest runoff occurred in the 2009–2010 water-year on both plots 1 & 2, and is likely the result of the infilling with water of the initially empty pore space in the waste rock and laterite from which the trial landform was constructed.

Annual runoff on plot 1 was greatest in the wettest year (2010–11) when 13.5% rainfall was converted to runoff, and was least in 2009–10 when the trial landform was 'wetting up' (Table 2). On plot 2 annual runoff was always higher than for plot 1, with the wettest year (2010–11) having 14.8% of rainfall converted to runoff. Interestingly the 2012–13 wet season had the greatest runoff coefficient of 15.9%, a result that requires further investigation.

Table 2 Rainfall and runoff data for erosion plot 1 on the trial landform for the five years of measurement.

Water year	Maximum event rainfall (mm)	Total Number of runoff events	Number of runoff events over the flume	Annual Runoff (L)	Runoff (mm)	Runoff coefficient (%)
2009–10 – plot 1	76.6	165	131	74886	81	5.3
2010–11 – plot 1	189.4	249	213	275650	300	13.5
2011–12 – plot 1	85.0	174	129	97366	106	7.0
2012–13 – plot1	72.8	129	92	111603	121	9.5
2013–14 – plot 1	72.6	183	156	138228	150	7.7

Table 3 Rainfall and runoff data for erosion plot 2 on the trial landform for the five years of measurement.

Water year	Maximum event rainfall (mm)	Total Number of runoff events	Number of runoff events over the flume	Annual Runoff (L)	Runoff (mm)	Runoff coefficient (%)
2009–10 – plot 2	77.4	166	125	121794	139	9.1
2010–11 – plot 2	180.0	258	221	298294	341	14.8
2011–12 – plot 2	85.0	174	150	151853	173	11.3
2012–13 – plot 2	56.4	143	114	177534	203	15.9
2013–14 – plot 2	73	171	151	236416	246	12.5

There looks to be an exponential relationship between event rainfall and event runoff over the full range of rainfall for all five years for plot 1 (Figure 3) although due to the issues with large events this has not yet been tested statistically. It is hypothesised that when event rainfall exceeds a value (in this case 30 mm) there is proportionally greater runoff than for smaller events (Figure 3). These smaller events do not totally infill the rip lines with water and so runoff is only produced from a small part of the plot near the down slope border. Event rainfall greater than 30 mm can totally infill the surface storage, hence generates runoff from the whole plot surface.

Sediment results and discussion

Suspended sediment

Since monitoring of the trial landform commenced, a large number of water samples have been collected based on turbidity reading increases measured in the flume of each plot. These samples have been analysed for turbidity and suspended sediment concentrations. The associated data have not yet been analysed and are not reported here.

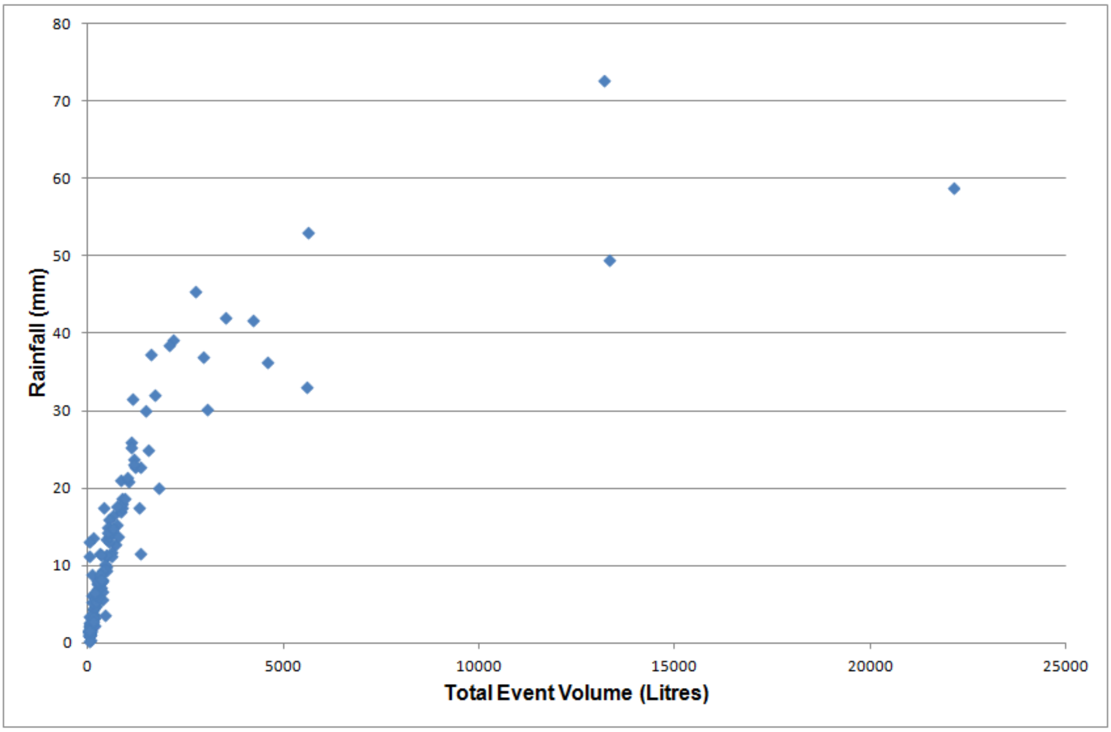

Figure 3 Relationship between total event rainfall and runoff for erosion plot 1 for 156 runoff events in the 2013–14 water-year.

Bedload

The annual bedload yield for each plot has clearly declined progressively since construction (Table 3). Time since construction of the TLF and increasing vegetation have impacted on annual bedload yields as yields have declined progressively over time on all plots, except for plot 2 for 2012–13 and 2013–14, (observations and photographs of plot 2 suggest that this plot has the least amount of vegetation growth). Sediment yields for major land disturbances, such as construction or landslides, are usually characterised by an initial pulse followed by a rapid decline (Duggan 1994). This is true for the TLF annual bedload yield, which is characterised by an exponential decline in annual bedload yield over the five years since construction (Figure 4).

Table 3 Annual bedload yields (t/km^2.yr) for each plot for each year of measurement.

Water-year	Plot 1	Plot 2	Plot 3	Plot 4	Mean Annual Bedload Yield ± Standard Error
2009–10	106	147	111	143	127 ± 11
2010–11	59	113	54	56	71 ± 14
2011–12	34	48	38	15	34 ± 7
2012–13	28	50	14	14	26 ± 9
2013-14	24	53	10	14	25 ± 10

Previous research in the Alligator Rivers Region has shown that sediment yields decline progressively over at least the first three years following a major surface disturbance, as a result of initial washout of fine sediment and the subsequent formation of a gravel-armoured surface (Duggan 1994). Clearly, time since construction, rather than rainfall, is the dominant driver of bedload yield as the greatest rainfall occurred in the second year

(Table 3). Using the average rainfall per rain day as an index of rainfall intensity, the values for the four years were 13, 15, 11, 10 and 15 mm d^{-1} for the 2009–10, 2010–11, 2011–12, 2012–13 and 2013–14 water years, respectively. The 2010–11 and the 2013–14 wet seasons were the wettest seasons and also had the most intense rainfall, further supporting the fact that rainfall is not a key driver for annual bedload yield on the trial landform.

The highest annual bedload yields were always generated from plot 2 (Table 3). While it is still not clear why this happens, shallow rip lines dominate the lower part of plot 2, resulting in direct connection of diffuse overland flow with the down slope plot drain and poorer vegetation establishment.

In the fourth year after construction, a clear signature of the effect of vegetation establishment is evident in the annual bedload yields which continue in the fifth year. The two plots (1 and 4), originally planted with tube stock, now have the greatest average tube stock heights (3.14 m for plot 1 and 2.74 m for plot 4) and both show lower bedload yields than the plots (2 and 3) initially planted by direct seeding followed a year later by infill planting with tube stock (1.90 m for Plot 2 and 1.53 m for plot 3)(Gellert 2014). Plots 3 and 4 have the lowest yield because they have also been invaded by weeds, which densely cover about half of the plot and mitigate against erosion.

Particle size analysis

For plots 1, 2 and 3, analysis showed that the sand fraction has the highest percentage (all over 50%, Table 4). On plot 4, the sand fraction was higher for years 2009–10 and 2012–13 and was essentially the same for the other three years. This indicates the importance of the sand fraction and shows that it is the main erosion product in the early years after construction of a rehabilitated landform. The gravel fraction constitutes the next highest percentage with little silt and clay contained in the bedload. Surface armouring of coarse gravel has occurred by the washing out of silt and clay from the ground surface.

Solute generation and transport

The waste rock used to construct the trial landform originates from Ranger Pit 3 (Map 2). Waste rock is the most abundant substrate for capping the final landform. Previous work on the chemical characterisation of Ranger mine waste rock has shown its potential to generate soluble manganese, magnesium, sulfate, uranium, calcium, aluminium, iron and potassium (East et al. 1994, Taylor et al. 1996 Hollingsworth et al. 2003). These solutes are mobilised during rainfall events and are transported via both the surface runoff and seepage pathways.

Typically, the rate of solute generation from waste rock declines over a period of years as the source becomes exhausted, and eventually stable, with inert end-products remaining (Hollingsworth et al. 2003). An intra-seasonal decline is also observed, with high solute concentrations causing a ‘first flush’ effect at the commencement of the wet season followed by a subsequent decline in concentration towards the end of the wet season. The first flush effect is attributed partly to the fact that rainwater in the region has a lower pH during the early wet season (compared to the late wet season) and is more effective at remobilising solutes and also to the fact that epsomite ($MgSO_4.7H_2O$), which is the dominant soluble salt formed as a result of weathering of Ranger waste rock, accumulates at the surface during dry periods and is flushed away during the first rains (Noller et al. 1990, East et al. 1994). This behaviour would be true for other major salts present.

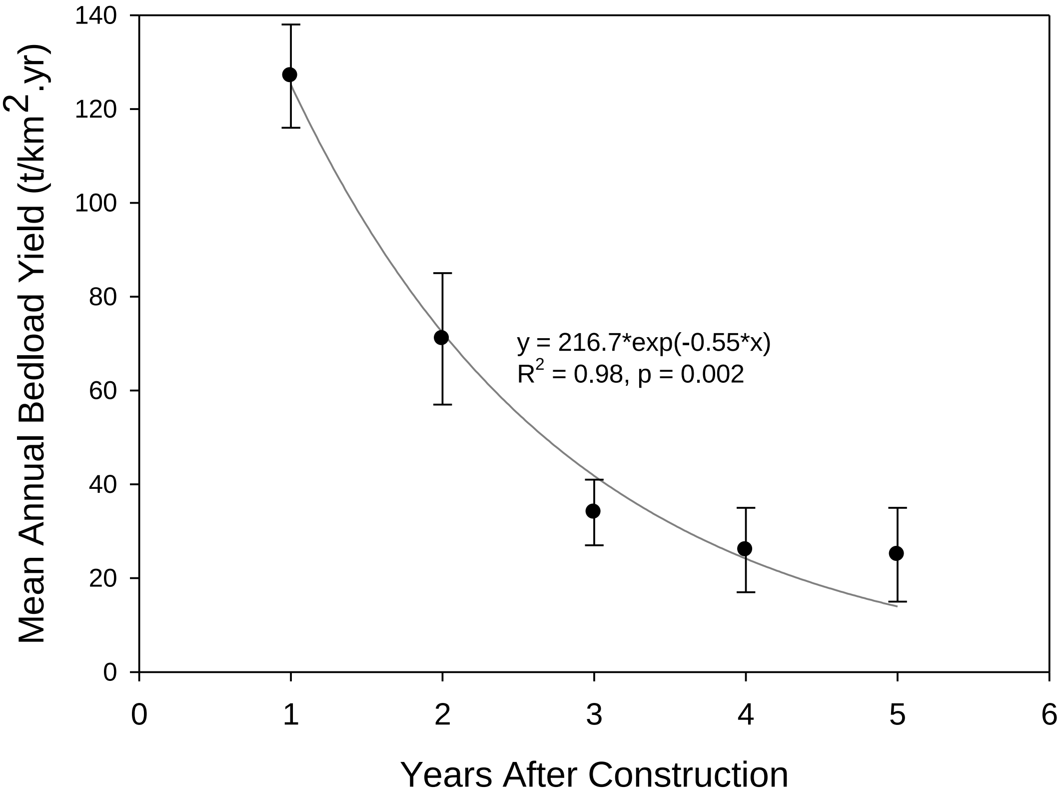

Figure 4 Exponential decrease in mean annual bedload yield with time since construction for the four plots on the trial landform. Data represent annual mean and standard error of estimate for all plots.

Table 4 Annual yield of bedload fractions as a percentage of total bedload yield for each plot for each year of measurement.

	Plot 1			**Plot 2**			**Plot 3**			**Plot 4**		
Water year	**% gravel**	**% sand**	**% silt and clay**	**% gravel**	**% sand**	**% silt and clay**	**% gravel**	**% sand**	**% silt and clay**	**% gravel**	**% sand**	**% silt and clay**
2009/10	34	60	6	34	55	11	37	59	4	35	61	4
2010/11	33	64	3	40	55	5	46	53	1	50	49	1
2011/12	44	53	3	42	55	3	47	52	1	50	49	1
2012/13	40	57	3	31	65	4	45	54	1	45	54	1
2013/14	42	55	3	36	60	3	45	53	2	50	48	1

The findings for the first three years of monitoring were presented in Saynor et al. (2013), focussing on chemical characterisation of the surface runoff and the inter- and intra-seasonal variation in runoff solute concentrations and loads. Data for plot 1 only were presented, as the quality of the EC and solute data cannot be verified until the discharge data have been validated. To date no additional work has been completed, but the results will be contained in the final Supervising Scientist Report.

Conclusions and future work

The priority for further work is to complete the calculation of runoff data from all plots, since the runoff must be determined before suspended sediment and solute loads can be derived. Correction of the runoff from all plots is progressing as expected. After five years the TLF project was to be reviewed with the results published as a Supervising Scientist Report. The data analysis for this report is currently under way.

Monitoring the trial landform during the 2014–15 is planned to continue in a reduced capacity. Rainfall, runoff and bedload will be collected whilst all efforts are concentrated in processing and reporting the first five years of the project. Additional objectives include quantifying the effect of developing vegetation on erosion rates, such that a higher level of confidence can be placed in the predictions from the landform evolution models (see Lowry et al. 2011) that are being used to predict long-term erosion performance. The runoff, sediment and solute loads that are being measured will also inform the design of sediment traps and wetland water quality polishing systems that will need to be incorporated into the rehabilitated mine footprint to manage the export of erosion products.

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A multi-year assessment of landform evolution model predictions for a trial rehabilitated landform

J Lowry, M Saynor & W Erskine

Introduction

The Environmental Research Institute of the Supervising Scientist, in collaboration with research partners at the University of Hull (Professor Tom Coulthard) and the University of Newcastle (Associate Professor Greg Hancock), has carried out a multi-year assessment of the geomorphic stability of the trial rehabilitated landform of the Ranger mine using the *CAESAR-Lisflood* landscape evolution model (LEM) (Lowry et al. 2014). *CAESAR-Lisflood* (Coulthard et al. 2013) is an enhanced version of the CAESAR LEM (Coulthard 2000) that had previously been used to assess the geomorphic stability of the Ranger trial landform (Lowry et al. 2011, Saynor et al. 2012). The LEMs can provide information on landform stability at decadal or centennial scales over large spatial extents, and evaluate the sensitivity of these processes to environmental changes. An important issue associated with the use of models is the ability to assess the reliability and accuracy of the model. In this study, the *CAESAR-Lisflood* LEM is tested for its ability to predict bedload and suspended sediment loads from specially constructed erosion plots on the Ranger trial landform (Figure 1).

These were compared with field measured observations collected over four wet seasons from 2009. Once calibrated for the specific site hydrological conditions, the predicted bedload demonstrated an excellent correspondence with the field data. However, longer-term simulations of 10 years identified an exhaustion effect in sediment yield from the landform. This latter result indicated that the incorporation of a weathering function into the *CAESAR-Lisflood* LEM will improve the model's ability to correctly predict the long term evolution of a rehabilitated landform once it has been constructed.

Methodology

The application of the *CAESAR-Lisflood* LEM to the trial landform required the collation and integration of data from a range of different sources. The key data inputs used by the model were a digital elevation model (DEM) of each erosion plot; rainfall data and surface particle size data.

A DEM of the trial landform was produced from data collected by a Terrestrial Laser Scanner in June 2010. For the purposes of this study, the data for the erosion plots were interpolated to produce a surface grid with a horizontal resolution of 20 cm. The DEMs were processed using ArcGIS software to ensure that the DEMs were pit-filled and hydrologically corrected. This pit filling was important in order to remove data artefacts, which included remnants of vegetation (peaks) as well as artificial depressions, or sinks that existed in the data but were not on the ground.

For the purposes of this study, the *CAESAR-Lisflood* model utilised rainfall data collected on the trial landform at a 1-minute interval. The data were aggregated into 10-minute intervals for use in the LEM. Rainfall data collected during the 2009–10, 2010–11, 2011–12 and 2012–13 rainfall years from the trial landform surface are used in the simulations reported here.

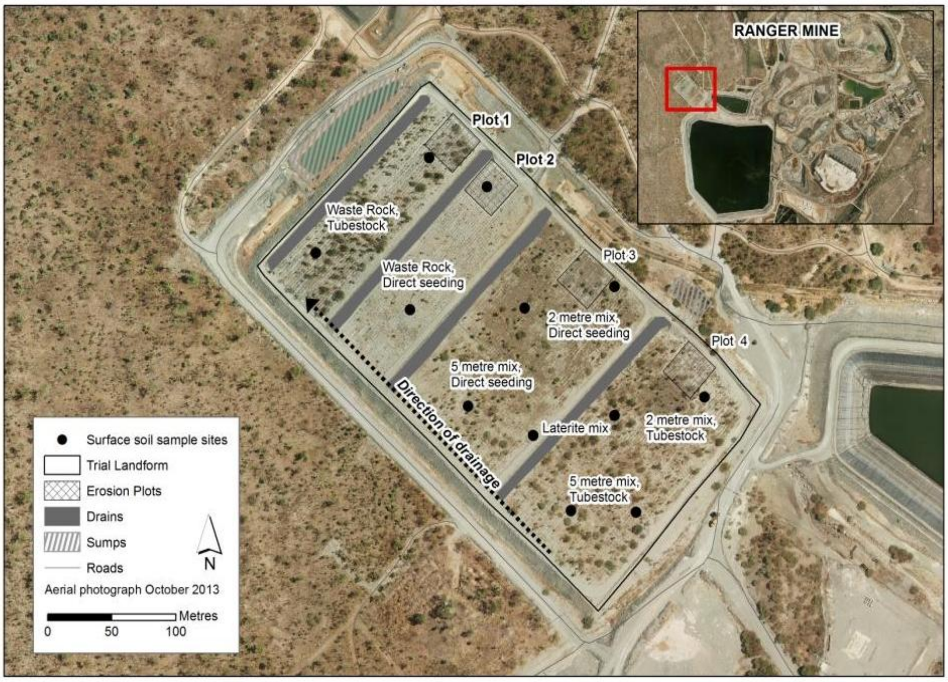

Figure 1 Aerial photograph of the trial landform at Ranger mine showing the size and location of plots 1 and 2.

Using the process described in Saynor and Houghton (2011), the grain size data for *CAESAR-Lisflood* were obtained from size fractionated bulk samples of surface material collected at eight points on the waste rock surface of the Ranger trial landform. Grain size analysis was completed on these samples and the results averaged into nine grain size classes ranging from 63 μm to 64 mm.

The *CAESAR-Lisflood* model currently does not have a weathering function. Consequently, when running simulations for periods of four years, it was necessary to manually simulate a weathering effect. This was done by stopping the simulation after two years, modifying the proportions of the grain size classes used (reducing the proportion of the largest grain size) and restarting the simulation to run for the remaining two years of the simulation period.

The comparison of *CAESAR-Lisflood* modelled results and field measurements focused on Plots 1 and 2 as they had the most complete sets of validated hydrological and measured bedload data, and corrected DEMs at the time of writing. Consequently, only these two plots have been used in this study.

Five sets of simulations have been conducted for each plot:

1. A 4-year simulation using rainfall data collected on the landform for the period 2009–13 at intervals of 10 minutes.
2. A 10-year simulation using the 2009–13 rainfall data looped 2.5 times.
3. A 10-year simulation using the 2009–13 rainfall data looped 2.5 times with the inclusion of measured data from an extreme rain event, in which 785 mm fell over 72 h between 17:00 h on 27 February and 17:00 h on 2 March 2007 at Jabiru Airport, inserted in the first year of the simulation.
4. A 10-year simulation using the 2009–13 rainfall data looped 2.5 times with the inclusion of the 2007 extreme event inserted in the third year of the simulation.
5. A 10-year simulation using the 2009–13 rainfall data looped 2.5 times with the inclusion of the 2007 extreme event inserted in the eighth year of the simulation.

Results

The four-year simulation results for measured and modelled bedload yields are shown in Figures 2 (Plot 1) and 3 (Plot 2). These indicate that after a period of four years, the modelled and measured bedload figures for both plots are within a range of 10% of each other and thus very similar. Longer term 10 year simulations of Plots 1 and 2 were run utilising the rainfall scenarios described earlier in the methodology section. Both plots returned the same trends in denudation and sediment yield under the different scenarios (Table 1). These show that the addition of an extreme rainfall event after three years produces the greatest increase in sediment yield, whilst the addition of an extreme rainfall event after eight years does not appear to have an impact on the sediment yield or denudation rate.

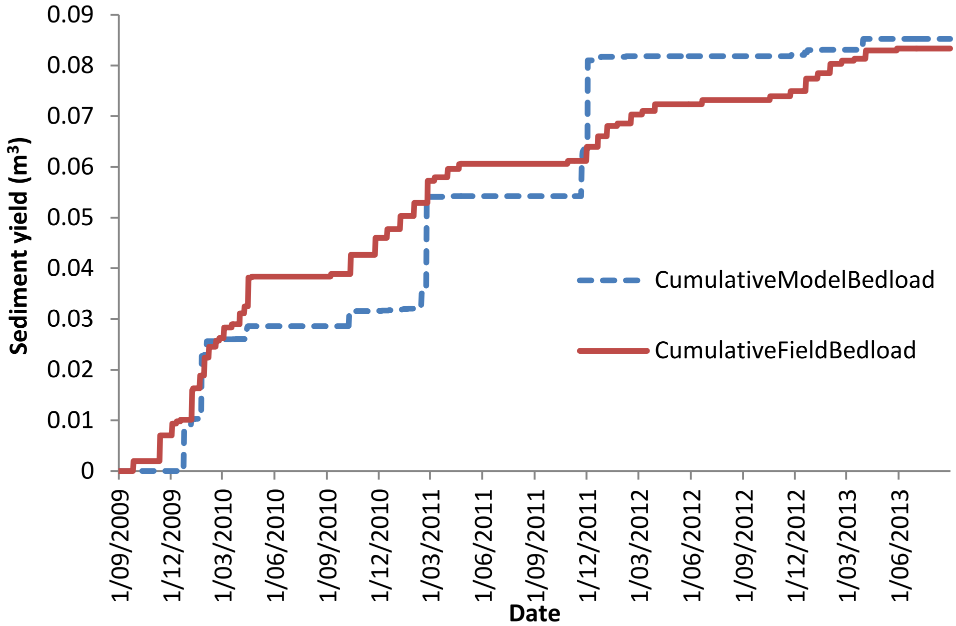

Figure 2 Comparison of modelled cumulative bedload (blue broken line) and field-measured cumulative bedload (red solid line) yield for Plot 1.

Table 1 Predicted total loads and denudation rates after 10 years for plots 1 and 2.

	Plot 1		**Plot 2**	
	Total Load (m^3)	**Denudation rate ($mm\ yr^{-1}$)**	**Total Load (m^3)**	**Denudation rate ($mm\ yr^{-1}$)**
10 years	0.38	0.04	0.24	0.02
10 years – extreme event in year 1	0.44	0.05	0.37	0.04
10 years – extreme event in year 3	0.54	0.06	0.41	0.05
10 years – extreme event in year 8	0.38	0.04	0.24	0.02

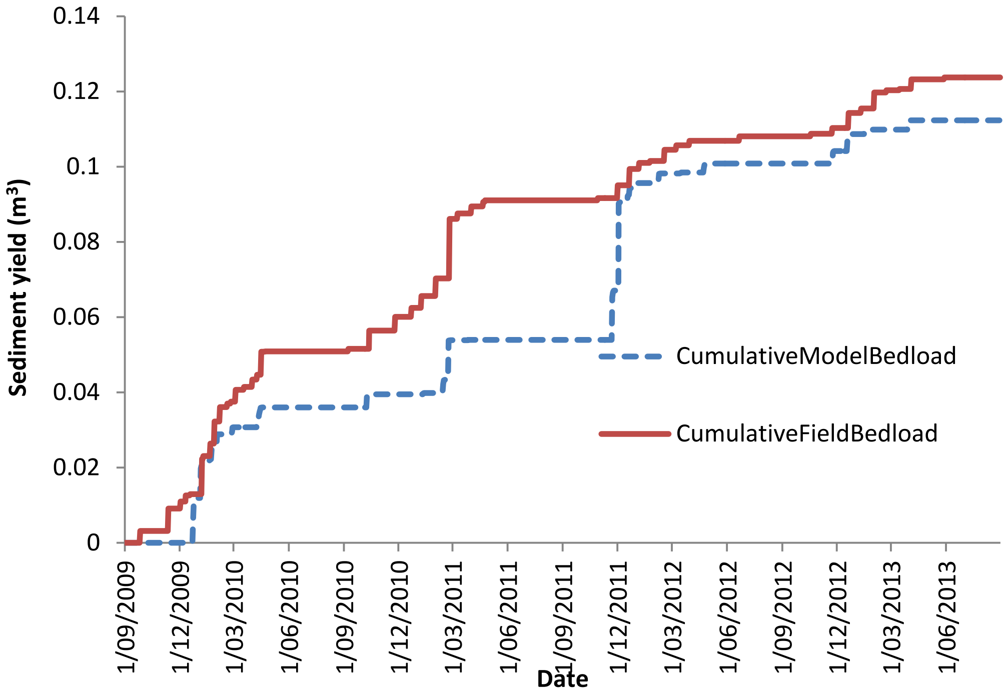

Figure 3 Comparison of modelled cumulative bedload (blue broken line) and field-measured cumulative bedload (red solid line) yield for Plot 2.

The introduction of an extreme event (utilising the rainfall from February 2007) at the beginning of year 3 and toward the end of the simulation period is shown in Figures 4 and 5 respectively. All simulations for both plots show a sediment exhaustion effect well before the end of the 10 years, regardless of the presence or timing of the extreme event.

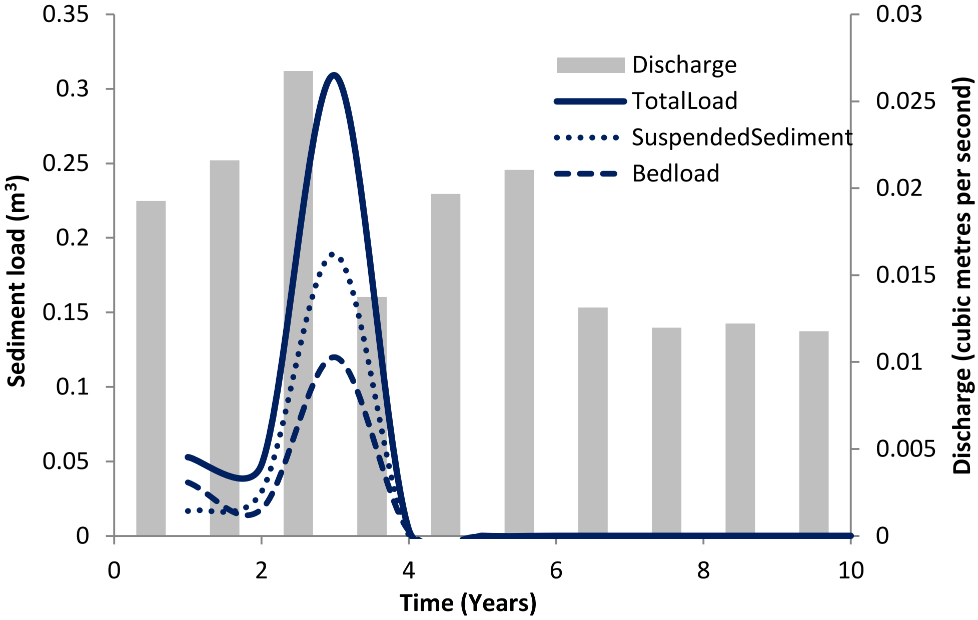

Figure 4 10 year simulation for Plot 2 with the extreme rain event after 3 years.

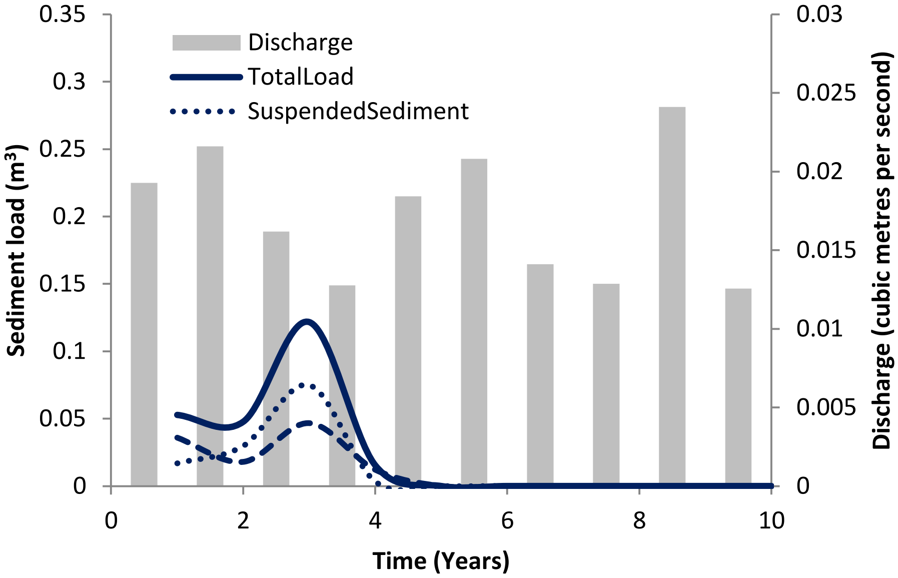

Figure 5 10 year simulation for Plot 2 with the extreme rain event after 8 years.

Conclusions and future work

The predicted bedload yields from the *CAESAR-Lisflood* simulation for the period 2009–13 demonstrate an excellent correspondence with the field measurements for the same period for both Plots 1 and 2. For the first two years (2009–11), measured cumulative bedload from Plot 1 (Figure 2) was slightly higher than the predicted cumulative bedload. However, at the start of the 2011–12 water year, a spike in predicted bedload yield occurred, which exceeds the measured bedload. This is attributed to a manual (as opposed to automatic) modification in the proportion of the larger particle sizes used in the simulation. This represented an attempt to manually introduce a weathering function into the *CAESAR-Lisflood* model. The effect of this was to produce a final predicted bedload yield which is approximately 3% greater than the final measured bedload yield. The similarity between the final predicted and measured bedload yields provides encouragement that the model is able to predict bedload from a rehabilitated surface. For Plot 2 (Figure 3), the predicted bedload generally compares well with the measured bedload. Compared to Plot 1, the predicted bedload is less than the measured throughout the simulated period. However, at the end of the simulation period, the total bedload yield of both the predicted and measured datasets are very similar – within 7% of each other.

The denudation rates for Plots 1 and 2 are 0.07 mm yr^{-1} and 0.06 mm yr^{-1}, respectively, over a simulated period of four years. These are higher than the published rates (0.01 – 0.04 mm yr^{-1}) of natural denudation for the region (Cull et al. 1992, Erskine & Saynor 2000). However, it must be noted that the latter were determined from a range of catchments of different size. In this study, each plot represented a closed catchment of approximately 900 m^2 with little initial vegetation cover and freshly exposed unweathered waste rock. When extended to a simulated period of 10 years, the predicted denudation rates for both plots matched the published rates.

Several caveats need to be placed on the results produced to date. Foremost is recognition that the simulations have modelled an 'idealised' environment. Specifically, the erosion plots are located on a uniformly gently sloping (2%) surface that represents only a component, albeit a substantial fraction, of the total area of the proposed rehabilitated landform. In addition, the roles of vegetation or fire were not considered in the simulations. Similarly, the study plots are closed catchments, with no capacity to recharge or replenish the material within the plot.

Previous studies have focused on collecting field data to enable the parameterisation of specific LEM applications (i.e. SIBERIA). Currently, field data are being collected on a stand-alone basis and can be used to support a range of model applications. In this case, field measurements closely match predicted outputs of the CAESAR model, thereby validating model results over the period of field collection. The development of a weathering module to incorporate into the *CAESAR-Lisflood* model will provide increased confidence in the ability of *CAESAR-Lisflood* to predict the long-term stability of a rehabilitated landform.

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Use of slackwater deposits and other forms of geologic evidence to determine the number, magnitude and frequency of palaeofloods in the Alligator Rivers Region

M Saynor & W Erskine[1,2]

Background

The stability of the proposed rehabilitated Ranger landform over a period of 10,000 years (Refer to KKN 2.2.4 – Geomorphic stability of the Ranger landform) will be assessed using the *CAESAR-Lisflood* and SIBERIA landscape evolution models. There are various inputs into the models including extreme rainfall and floods. To undertake the landform evolution modelling, it is essential that the number, magnitude and frequency of palaeofloods in the Alligator Rivers Region (ARR) larger than the largest historical events that have occurred during the last 7000 years since sea level has been relatively stable (Woodroffe et al. 1987) are known. Climate change has also been minor during this period.

Palaeoflood hydrology was defined by Baker (1987a, page 79) as "the study of ancient floods which occurred prior to the time of human observation or direct measurements by modern hydrologic procedures". This technique allows the estimates of flood magnitude and frequency to be made for actual events that have occurred during a period of essentially constant climate, provided that these palaeofloods have left a record of flood magnitude by erosion or deposition. The sites of erosion and deposition, usually bedrock channels, must not have changed in channel geometry during the flood to enable calculations of flood peak discharge and peak stream power. Therefore, resistant bedrock gorges are often used for such investigations. Erosional features include flood trim lines, cataracts or scour holes, spillover areas and scars on trees. Depositional features are usually slackwater deposits (SWD) of typically fine-grained sand and silt, which accumulate rapidly from suspension during major floods in protected areas where flow velocities are locally reduced (Baker et al. 1983). SWDs are found in four main locations:

- tributary mouths
- shallow caves along bedrock walls of a gorge
- separation envelopes with reverse currents in areas upstream of constrictions, and
- overbank deposits (Kochel & Baker 1988, Pickup et al. 1988).

These locations provide favourable conditions for the deposition and subsequent preservation of SWD. Large floods usually deposit SWD, in a vertically layered sequence, however, except for inset deposits, new deposits cannot be emplaced unless the subsequent flood exceeds the height of the earlier one. SWD record the minimum peak height of a flood because the flood waters must exceed the SWD to enable deposition of the sediment (Baker 1987b). The best recorded

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accuracy for SWD in terms of flood peak discharge estimation is an underestimate of 9% (Erskine & Peacock 2002). Therefore, it is best practice to increase peak discharges estimated by SWD by 10% to allow for a zone of non-deposition immediately below the flood peak.

The upper reaches of the East Alligator River and its tributaries contain anabranching bedrock channels and long straight gorge reaches, often with abrupt angled bends that are potential areas where flood erosional features and SWD could exist. Recent investigation using imagery within Google Earth Pro has indicated areas of channel scabland on parts of the East Alligator River and Cooper and Magela Creeks. Channel scabland consists of extensive bedrock anabranching channels and scour holes or cataracts eroded by catastrophic floods on bedrock uplands where the formative floods could not be contained in a single channel.

The largest recorded flood on the East Alligator River occurred following extreme rainfall in February–March 2007 and caused extensive bank erosion and stripping of point bars and floodplain resulting in large amounts of sand transport (Saynor et al. 2009, 2012). Deep sand deposits were laid down on top of inset features in some areas of the East Alligator River (Figure 1). The large flood of 2007 provided an historical benchmark of erosion and deposition on the East Alligator River and information on potential sites of SWD laid down by even larger palaeofloods. If SWD can be identified, it should be possible to date them either by optically stimulated luminescence (OSL) (suitable for quartz sand) and/or post-bomb accelerator mass spectroscopy radiocarbon dating (suitable for leaves). Surveys of the channel/gorge cross-sections combined with the use of hydrodynamic models will provide information on the magnitude of these events and their hydraulics.

Figure 1 Sand deposits laid down on top of a point bar in the Gorge section of the East Alligator River after the 2007 flood, shown in the May 07 image. The photos were taken from different elevations with the red line indicating the same locations on the photos.

To try and identify potential SWD, river reach analysis was undertaken on the East Alligator River (7000 km^2 catchment) and its three named tributaries, Magela, Tin Camp and Cooper Creeks (Saynor & Erskine 2013, 2014).

Methodology

The approach adopted to identify, name and describe river reaches in the study area follows Erskine (2005) and Erskine et al. (2014). River reaches are homogeneous lengths of channel within which hydrologic, geologic and adjacent catchment conditions are either sufficiently constant so that a uniform river morphology is produced (Kellerhals et al. 1976) or a consistent pattern of alternating river morphologies is produced (Erskine et al. 2001). Alternatively, river reaches could be defined as relatively homogeneous associations of channel units that distinguish them from adjoining reaches (Bisson & Montgomery 1996, Brierley & Fryirs 2005). However, the latter approach requires too much field work for application in remote areas, such as the East Alligator River. River reaches are typically 2 km to more than 100 km long although can be shorter and longer. It is relatively easy to identify the core length of a reach, but it is more difficult to define precisely the boundaries of a reach because of their transitional nature (Erskine 1996).

Formal names for reaches comprise of at least three terms. The first term is a geographic name for a location or feature within or next to the reach. The second term is a geomorphological descriptor for one of the dominant characteristics of the reach. The third term, when needed, is the word, reach or zone. River reaches on the East Alligator River and its named tributaries were identified using topographic maps (1:50,000), aerial reconnaissance and imagery viewed in Google Earth Pro, and supplemented with very limited ground truthing. Once identified, the lengths of the reaches were measured and long profiles constructed using the topographic maps.

The classification of the named rivers in the East Alligator catchment provides information that has been used to locate potential palaeoflood erosional features and SWD sites. Potential sites will be visited, to determine if they contain SWD and/or erosional features. Where SWD and erosional features are identified, samples of the sediments will be collected for dating and particle size analysis. Topographic surveys or possibly the UAV will be used to generate cross sections used to reconstruct flood heights.

Results and Progress to date

The East Alligator River and its named tributaries was classified into different river reaches (Table 1). Dominant river types on the East Alligator River in terms of channel length were anabranching rivers, sandstone gorges and cuspate tidal meanders. On Magela Creek, dominant river types were wetlands and channel billabongs, island anabranching and sandstone gorges. On Cooper and Tin Camp creeks, anabranching rivers dominated. Table 1 shows that anabranching reaches dominated on 3 of the 4 characterised rivers and five different types of anabranching were identified and named (Saynor & Erskine 2013, 2014).

Table 1 Number of reaches classified on the East Alligator River and its named tributaries.

River	Length (km)	Number of reaches	Dominant reach type	Percentage of length
East Alligator River	241.4	16	Anabranching reaches	30.1
Magela Creek	118.8	10	Wetlands and channel billabongs	35.9
Cooper Creek	147.3	6	Anabranching reaches	60.9
Tin Camp Creek	79.7	8	Anabranching reaches	79.8

A number of sites in the upper sandstone gorge reaches are ideal for the preservation of SWD and could be potential sites for SWD research. The reach classification process also identified areas of channel scabland in the East Alligator catchment. They have been previously identified on the neighbouring Katherine River (Baker & Pickup 1987). The sites that will be investigated for erosional and depositional evidence of palaeofloods on the East Alligator River and its named tributaries are outlined in Table 2.

Table 2 River reaches to be investigated in the East Alligator River catchment for palaeoflood analyses. See Figure 3 for location of river reaches.

River	River Reach with possible evidence of Palaeofloods	Palaeoflood Evidence to be Investigated
East Alligator River	Upper East Alligator River Gorge* (Reach 6)(56.8 km long)	SWD, channel scabland
Magela Creek	Mamadawerre Upland Gorge* (Reach 5)(4.7 km long)	SWD, channel scabland, flood bars
	Magela Falls Gorge (Reach 7)(10.5 km)	Possible SWD
Cooper Creek	Coopers Lagoons Bedrock Anabranching Reach* (Reach 4) (15.5 km long)	Channel scabland, SWD, flood bars
Tin Camp Creek	Myra Falls Gorge* (Reach 5)(0.8 km long)	Possible SWD

*Preferred site

Figure 2 Oblique aerial photos of Coopers Lagoons on the sandstone plateau surface near Mt Borradaile with essentially no feeder channels. They have formed by waterfall scour and retreat leaving a large plunge pool during palaeofloods flowing across the sandstone plateau.

An excellent SWD site has already been found on the East Alligator River (Figure 1) and cataracts have been found on Coopers Creek (Figure 2) that indicate that palaeofloods have inundated the whole sandstone plateau surface there to sufficient depth to generate stream powers that have eroded waterfalls with large plunge pools in quartzose sandstone.

Steps for completion

Previous palaeoflood hydrology on the East Alligator River (Pickup et al. 1987, Murray et al. 1992) was conducted in reaches downstream of the true sandstone gorges and found that the existing slackwater deposits were very young, only occurring at low elevations. Because these authors had not identified the true gorges on the East Alligator River, their study site was subject to frequent scour and fill, a fact that they explicitly recognized. Following the river reach analysis, it is now clear that such research should have been completed in the Upper East Alligator Gorge and other sandstone gorges. On Magela Creek, Cooper Creek and upper East Alligator River, channel scabland has been identified (Table 2). We are unaware of any of the previous geomorphic investigations of the East Alligator River recognising the Upper East Alligator Gorge and its associated channel scabland (Pickup et al. 1987). Flood erosional evidence and slackwater deposit investigations will be undertaken to determine palaeofloods. We plan to concentrate our research efforts on sandstone gorge areas, and are currently planning palaeoflood investigations with Professors R. J. Wasson of the National University of Singapore and W.D. Erskine of the University of Newcastle and Charles Darwin University. During the

2015 dry season, sites in the Upper East Alligator River gorge will be inspected for SWD and sampled where they are located. Other sites will be inspected aerially and sampled in subsequent dry seasons.

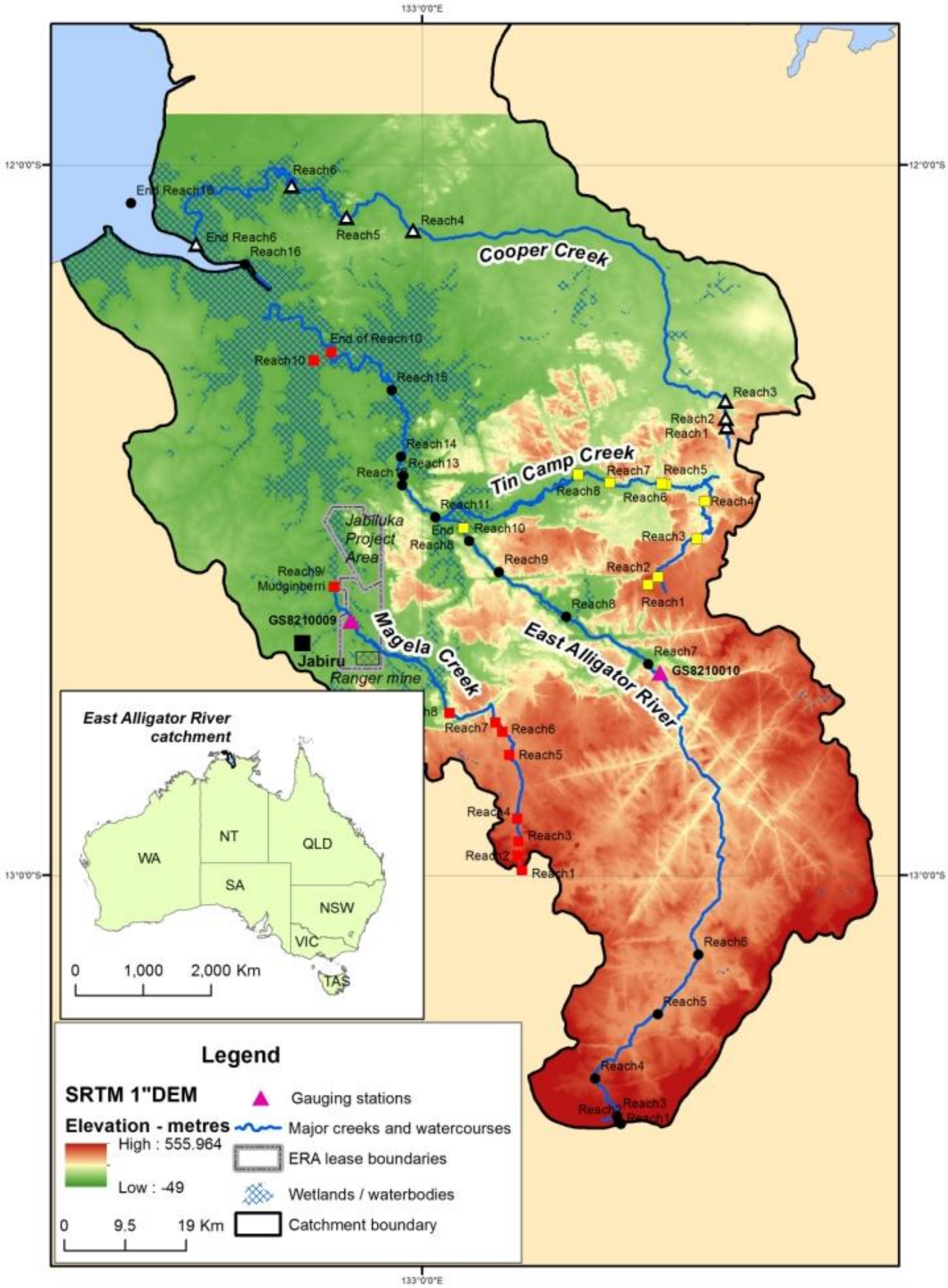

Figure 3 Location of the river reaches listed in Table 2 on the East Alligator River and named tributaries.

Discussions with the Northern Land Council are underway to obtain permission from the traditional land owners to allow us access to the identified areas to confirm the existence of SWD and other palaeoflood indicators. Any SWD that are identified will be surveyed for palaeostage information, sampled for sedimentological and dating purposes, and a channel digital elevation model obtained to reconstruct flood magnitudes by hydrodynamic modelling. At least the HEC-RAS one-dimensional backwater model (Brunner 2010) will be used, although more sophisticated hydraulic models will also be investigated. The channel scablands will be investigated as part of this work because they also provide important palaeostage information of flood erosion which can be combined with the depositional evidence.

The key benefits of this project are that the number, magnitude and frequency of palaeofloods that are larger than the largest historical event in the ARR will be determined for the current period of relatively stable climate. The peak discharge information will be scaled for the Magela and Gulungul creek catchments at the Ranger Mine by catchment area. The palaeoflood data will also be used to estimate the formative rainfall by extension of historical data so as to construct input data for landform evolution modelling.

Knowledge of the number, magnitude, frequency and time of occurrence of palaeofloods and their formative rainfall will significantly extend our knowledge of extreme events in the ARR. Furthermore, greater confidence will be gained in the results of the landform evolution modelling of the stability of the rehabilitated Ranger Mine for the first 10000 years following construction once actual palaeofloods have been determined. This is important to convey to stakeholders that world's best practice has been applied to the assessment of the stability of the rehabilitated mine site.

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Sediment movement in Magela Creek Catchment

W Erskine[1,2], M Saynor, K Turner, T Whiteside, J Boyden & K Evans

Background

Studies of soil erosion rates and sediment yields in the Alligator Rivers Region have been investigated since before the construction of the Ranger mine. These studies have been undertaken to determine where sediments eroded from the rehabilitated mine site will be deposited. Erskine & Saynor (2000) suggested that the most significant storage areas downstream of the mine site will be in the mine site tributaries and their associated floodplains and backflow billabongs. This work brings together soil erosion and sediment yield data, presents a sediment budget and shows sediment movement for Magela Creek.

Summary

The work collates sediment loads for Magela creek Catchment and fits into two Ranger Rehabilitation KKN's: Defining the reference state and baseline data; and 2.2.4 Geomorphic behaviour and evolution of the landscape. Soil erosion rates on plots of waste rock at Ranger uranium mine and basin sediment yields have been measured for over 30 years in Magela Creek in northern Australia. Soil erosion rates on chlorite schist waste rock are higher than for mica schist and weathering is also much faster. Sediment yields are low but are further reduced by sediment trapping effects of flood plains, floodouts, billabongs and extensive wetlands. As a result, suspended sediment and bedload do not move progressively from the summit to the sea along Magela Creek and lower Magela Creek wetlands trap about 90.5% (Figure 1) of the total sediment load input.

Generally most of the sediment is deposited in the natural sediment traps on the many tributaries of Magela Creek. Suspended sediment yields exceed bedload yields in this deeply weathered, tropical landscape but the amount of sand transported greatly exceeds that for silt and clay. The material used in the rehabilitation of the Ranger mine may greatly increase sediment loads. With implications that there should be sediment traps around the rehabilitated landform that mimic the natural sediment traps and reduce the sediment movement similar to that of the natural Magela Catchment.

In summary, the title sets the scene "Do suspended sediment and bedload move progressively from the summit to the sea along Magela Creek, northern Australia" (Erskine et al. 2014) and the simple answer for both suspended sediment and bedload is no.

This work was presented at an international conference on Sediment Dynamics: From the Summit to the Sea was held in New Orleans, USA, 11 – 14 December, 2014. Wayne Erskine attended and gave the following paper, "Do suspended sediment and bedload move progressively from the summit to the sea along Magela Creek, northern Australia?" by Wayne D. Erskine, Mike Saynor, Kate Turner, Tim Whiteside, James Boyden and K Evans.

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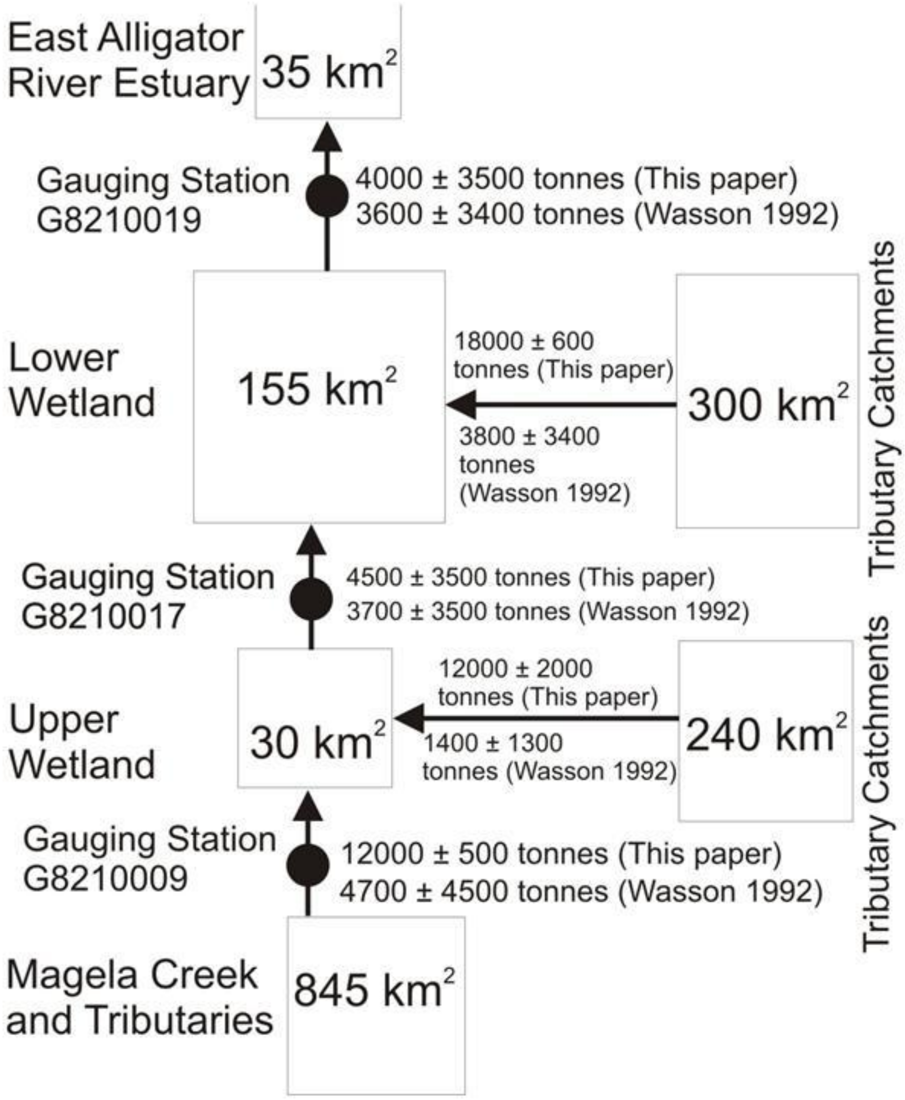

Figure 1 Mean annual sediment fluxes for lower Magela Creek wetlands. Wasson (1992) only cites suspended sediment load and this paper uses total sediment load for the period 2003–04 to 2012–13.

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Radon exhalation from a rehabilitated landform

R Akber, A Bollhöfer & C Doering

Introduction

The inhalation of radon decay products is to be included as an exposure pathway in the development of a radiological dose model to estimate above background doses to the public from the rehabilitated Ranger mine site. At Jabiru or Mudginberri, inhalation doses are unlikely to be of concern given that annual mine-derived doses received via the inhalation pathway for the operating mine site at present are less than 0.1 mSv (Supervising Scientist 2103). However, people roaming the site after rehabilitation for hunting and gathering activities and camping on site, or in areas nearby, may be exposed to higher radon decay product concentrations. Knowledge of the radon exhalation fluxes from the surface of the substrate used to shape the landform is the first step to predict potential post-rehabilitation doses received via the inhalation of radon decay products.

Radon (^{222}Rn) is part of the natural uranium decay chain and is produced by the decay of radium (^{226}Ra) in soil particles. It is a noble gas and some of the radon emanates from the particles, migrates through the soil pore space and eventually exhales from the soil surface. Radon exhalation depends on soil ^{226}Ra activity concentration and the ^{222}Rn diffusion length, which in turn is influenced by soil porosity, moisture and particle size (Porstendörfer 1994). Radon in air decays with a half life of 3.82 days to short-lived isotopes of the metals polonium (^{218}Po, ^{214}Po), lead (^{214}Pb) and bismuth (^{214}Bi). It is these radon decay products, rather than the radon gas, that can deposit in the lungs and deliver a radiation dose upon inhalation.

Previous work has focused on the Ranger trial landform to determine the seasonal and temporal changes in radon exhalation fluxes from different substrates (waste rock only and waste rock-laterite mix) (Bollhöfer et al. 2013). In 2013–14, the focus was on the measurement of radon flux densities from various height radon columns, which were established at the Jabiru Field Station to experimentally determine the radon diffusion length for waste rock used to rehabilitate Ranger uranium mine.

Methods

Two sets of six 240 mm diameter PVC tube columns (Figure 1) were set up at the Jabiru Field Station in April 2013, in collaboration with Energy Resources of Australia Ltd and Safe Radiation, Brisbane (SafeRadiation 2013).

The columns were filled with waste rock from Ranger mine consisting of a 4:1 mix of rocks with a diameter of ~70 mm and rocks and gravel less than 40 mm, believed to reasonably represent the substrate that will be used as a cover material for the site. The six columns in each set covered a rock depth from 0.5 m to 3.0 m in 0.5 m intervals, with a 0.2 m deep head space left at the top of the columns. The open end of each column has a machined flange to attach a lid and connect to the *Durridge Rad7* radon detectors that measure the ^{222}Rn exhaling from the surface of the waste rock in the columns. ^{222}Rn activity flux density measurements were conducted in May 2013 and one year later, in April 2014, to investigate whether there is a change in diffusion length with time due to compaction effects of the material in the columns. In addition, the ^{226}Ra activity concentration of the material used to fill each column was measured using scintillation detectors and gamma spectrometry.

Figure 1 Set up of the radon columns at the Jabiru Field Station (photo: *Safe Radiation* 2013).

The constant replenishment of ^{222}Rn through the decay of ^{226}Ra in the waste rock leads to the establishment of a ^{222}Rn concentration gradient in the pore space of the waste rock columns. Measurements show ^{222}Rn activity concentrations in the pore space in the order ~10^5 Bq·m^{-3} at two metres below the surface and about 200 Bq·m^{-3} at the surface. Radon diffuses along this concentration gradient and the ^{222}Rn activity flux density from the surface of each of the columns is determined from the increase of the ^{222}Rn activity concentration measured in air within the head space, after closing the column with the lid attached to the *Rad7* instrument.

Assuming that diffusion is the main process governing ^{222}Rn migration though the pore space, the following equation can be used to determine the diffusion length L in the material:

$$\text{Equation 1: } E(h) = E_\infty \cdot \tanh(h/L) = E_\infty \cdot [(1-e^{-2h/L})/(1+e^{-2h/L})]$$

where $E(h)$ and E_∞ are the ^{222}Rn activity flux densities in Bq·m^{-2}·s^{-1} from a layer of thickness h (in m) or an infinitely thick layer of a particular material, and L is the diffusion length, or effective relaxation length, in meters.

Results and Discussion

The ^{222}Rn activity concentration measured in the head space of a column is humidity corrected, as the relative humidity was generally above 30% during the measurements and the *Rad7* gives low values for a relative humidity above 10%. Figure 2 shows a plot of the ^{222}Rn activity flux densities from the various height columns, calculated from the increase of the ^{222}Rn activity concentrations measured in the head space of the columns over time, plotted versus the column height. A curve of the form given in Equation 1 has been fitted to the calculated ^{222}Rn activity flux densities from the various height columns. The coefficients L and E_∞ were obtained through a Levenberg-Marquardt non-linear least square curve fit and the covariant matrix generated was used to determine the variance and coefficient of determination, R^2.

From the fit to the experimentally derived data, a ^{222}Rn diffusion length L of 1.8±0.2 m and a ^{222}Rn activity flux density $E\infty$ of 0.87±0.08 Bq·m^{-2}·s^{-1} were obtained for waste rock with a ^{226}Ra activity concentration of 1860±220 Bq·kg^{-1} (average ± 2 stdev of waste rock ^{226}Ra activity concentration in the columns). The diffusion length is a little higher than typical diffusion lengths in natural soils (Porstendörfer 1994) most likely due to the more porous nature of the material compared to aged natural soils.

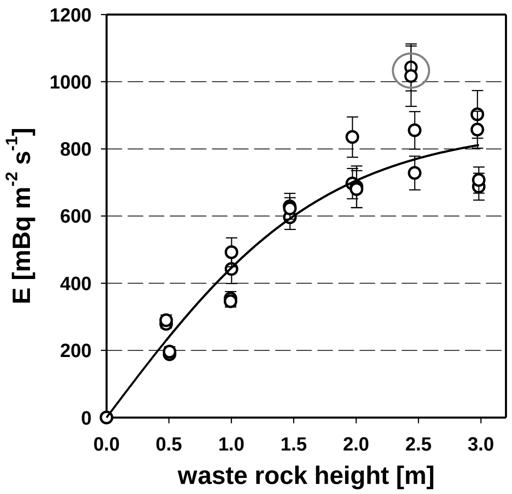

Figure 2 ^{222}Rn activity flux density E versus the height of waste rock in the columns from the April 2014 measurements. The circled data belong to column 2.5M2 and have been treated as outliers for the fit. A possible explanation for the higher ^{222}Rn activity flux density could be a higher than estimated activity concentration of the material used to fill the column, due to heterogeneities in the waste rock. The solid line is a curve fit to the data of the form given in equation 1.

Figure 3 shows the percentage values of E/E_∞ plotted against h/L. This plot has been used to estimate changes in the ^{222}Rn activity flux from the cover material with height of the material. Assuming a diffusion length of 1.8 m, there will be a less than 5% increase in the ^{222}Rn activity flux from the material layers exceeding 3.3 m in thickness. Two metres of the material will exhale^{222}Rn at an activity flux density of ~80% of the maximum value.

Effectiveness of waste rock as a cover material

The obtained diffusion length also allows to determine the effectiveness of the waste rock as a substrate to reduce the ^{222}Rn exhalation from a source underneath. Equation 2 can be used to determine the reduction of the ^{222}Rn activity flux density from a source (such as buried tailings) over which the waste rock is laid for capping. If a material with surface activity flux E_{0s} is covered with a thickness h of a capping material, then the flux will reduce to a value E_{hs} as:

$$\textit{Equation 2: } E_{hs} = E_{0s} \cdot e^{-h/L}$$

where E_{0s} and E_{hs} are the ^{222}Rn activity flux densities from the surface of the uncovered material and after a waste rock capping of height h has been applied, respectively. Figure 3 shows the reduction of the ^{222}Rn exhalation by applying a waste rock cover of height h as cover material, plotted against h/L.

It is obvious from Figure 3 that, while waste rock is suitable as a capping material to lower ^{222}Rn exhalation from buried ^{226}Ra rich material, it will also be a source of ^{222}Rn due to its generally above natural background ^{226}Ra activity concentration. A capping height of about three times the diffusion length will reduce the source ^{222}Rn exhalation to less than 5% but, at the same time, the ^{222}Rn exhalation from the capping material itself will effectively be E_∞. The value E_∞ for the radon activity flux density depends on the ^{226}Ra activity concentration of the capping material and, consequently, the lowest grade waste rock should be used for the surface of the rehabilitated landform.

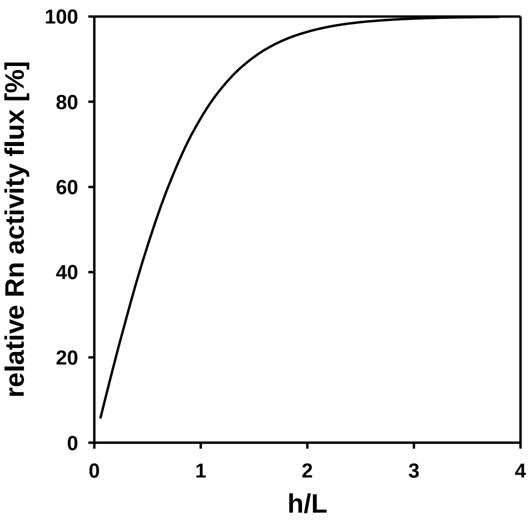

Figure 3 Relative ^{222}Rn activity flux density plotted versus the ratio of waste rock height and diffusion length. The black line shows the increase of the ^{222}Rn activity flux density from the waste rock itself with increasing height of the material (equation 1), the grey line shows the reduction of the ^{222}Rn activity flux density from a capped ^{222}Rn source with increasing height of the capping material (equation 2).

Conclusions

We have determined the radon diffusion length one year after set-up of the radon exhalation columns to be 1.8±0.2 m. The diffusion length has decreased little, if at all, compared to initial measurements conducted in May 2013. The diffusion length has been used to examine the effectiveness of waste rock as a cover material and also the ^{222}Rn exhalation from the waste rock itself. The ratio $R_{E\text{-}R}$ of E_∞ to the average ^{226}Ra activity concentration in the waste rock, determined in this experimental set up, is (0.47±0.07) $Bq \cdot m^{-2} \cdot s^{-1}$ per $Bq \cdot g^{-1}$, similar to the maximum dry season ratios determined through measurements on the trial landform. (Bollhöfer & Doering 2015).

A further set of ^{222}Rn exhalation measurements will be conducted in the dry season of 2015. This is to determine whether there is a statistically significant decrease in the ^{222}Rn diffusion length with time. Future work will include the modeling of above baseline ^{222}Rn exhalation fluxes from the rehabilitated landform, using pre-mining fluxes reported in Bollhöfer et al. (2014) and various rehabilitation scenarios. Ultimately, these above background fluxes will be used to determine above background doses from the inhalation of radon decay products.

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Radionuclide fluxes from the Ranger Trial Landform

F Evans, P Medley & A Bollhöfer

Introduction

Construction of the Trial Landform (TLF) started in late 2008 to investigate the generation and transport of sediments and solutes, and the ability to sustain growth of plant species native to the region (Saynor et al. 2009, Saynor et al. 2013). The TLF has also been used to assess radon exhalation and its temporal variability from different capping materials, (i) waste rock, and (ii) waste rock blended with approximately 30% fine-grained lateritic material (Bollhöfer et al. 2013). Erosion plots were installed on the TLF and instrumented with a range of sensors that are described in detail in Saynor et al. (2012). An important feature of the instrumentation are electrical conductivity (EC) probes located at the inlet to the stilling basin and at the entry to the flume (outlet of the stilling basin). These EC probes measure the concentration of dissolved solutes in the runoff and trigger an automatic pump sampler to collect event based water samples by predetermined changes in EC and turbidity readings (Saynor et al. 2013).

Saynor et al. (2013) and Supervising Scientist (2013) provided the first assessment of the generation and transport of key chemical solutes on erosion plot 1 since monitoring commenced. They showed that a number of analytes, including dissolved U concentrations, decrease with the cumulative runoff volume over a wet season, most likely an effect of exhaustion of available soluble metals from the substrate of the TLF. It has also been reported that annual mean concentrations of U have decreased compared to the first wet season, from 10 µg L^{-1} (124 mBq L^{-1} ^{238}U) in the 2009–10 wet season to 4.7 µg L^{-1} (58 mBq L^{-1} ^{238}U) in 2010-11 and 6.4 µg L^{-1} (79 mBq L^{-1} ^{238}U) in 2011–12.

From the decrease in dissolved U concentrations, it was assumed that a similar temporal decline would be observed for total U concentrations measured in TLF runoff samples. However, the total sample concentration (filtered through < 0.45 µm plus the residual, or > 0.45 µm, fraction) is not routinely analysed. Consequently, total U concentrations associated with TLF surface runoff were not calculated. As the residual sample fractions from the routine in-situ monitoring have been archived, mainly to inform an estimate of total suspended sediment in the runoff (Saynor et al. 2011), these filters have been retrospectively analysed in this project for U and other metals.

Radionuclide analysis was also conducted in 2010 on samples (filtered and residual fractions separately) collected from all four erosion plots during the 2009–10 wet season. These samples showed that activity concentrations of uranium-series radionuclides in the dissolved fraction generally followed the order ^{238}U > ^{226}Ra > ^{210}Po. The data, although limited, also suggested a decline in uranium-series radionuclide activity concentrations over the course of a single rainfall event and across successive rainfall events.

Project objectives:

- To derive total U concentration values for selected surface water runoff samples collected during the 2009–10, 2010–11, 2011–12 and 2012–13 wet seasons.
- To investigate the temporal trend of water U concentrations (total and particulate fractions) over the course of a single wet season and across successive wet seasons.
- To analyse radionuclide data to determine uranium-series radionuclide relationships.

Methods

Hundreds of filters containing the residual fraction of event triggered samples have been archived over the period of routine TLF runoff monitoring. Only filters from erosion plot 1 were selected for this study, as only data from this erosion plot have undergone appropriate validation and reporting (Saynor et al. 2013), and waste rock only is assumed to be the most likely substrate for rehabilitation of the Ranger mine site. Samples have been selected for analysis of the residual fraction on the following basis: (1) filters were chosen from those samples collected during runoff events with a difference in EC between in-flowing and out-flowing water from the stilling basin of less than 20 percent. This yielded an insufficient number of samples for analysis, and the following less restrictive selection criteria were established: (2) sitting times of the EC triggered water sample in the Gamet sampler was less than 6 weeks, and (3) soluble U data were available over the event. Based on the sample selection criteria steps 1-3 above, only a limited number of samples were identified and deemed appropriate for further investigation, resulting in 49 samples for further analyses.

As part of the routine monitoring of TLF runoff, the filtered fractions of EC triggered, event based samples from erosion plot 1 over the three consecutive wet seasons (2009–10, 2010–11, 2011–12) have been analysed for trace elements via inductively coupled plasma mass spectrometry (ICP-MS). For the purposes of deriving total values, the residual fractions for this project were analysed in 2014 via ICP-MS. Radionuclide analyses were conducted in-house following methods published in Martin & Hancock (2004) and Medley et al. (2005).

Results and Discussion

Figure 1 shows the results of the ^{238}U activity concentrations measured in the dissolved and total fractions (mBq L^{-1}), their ratio and the activity concentration (Bq kg^{-1}) in the particulate matter. Table 1 shows the results of uranium-series radionuclide activity concentrations measured in samples collected from the four erosion plots in April 2010.

The analysed residual fractions of the filter papers do not usually cover an entire runoff event, nor do they cover an entire wet season. For example, out of the 815 filtered water samples collected between 2009 and 2012, only 40 samples complied with QA step 1 (Turner, pers comm.) and Supervising Scientist (2013) noted that less than three good quality water samples were collected over the majority of runoff events. However, some observations can be made based on the less confounding sample selection procedure outlined above.

Dissolved ^{238}U activity concentrations shown in Figure 1 over the course of an individual rain event and throughout a wet season are highly variable. Concentrations depend on the intensity of rainfall that triggers an event, but also on total wet season cumulative runoff itself (Saynor et al. 2013). It has been reported that dissolved ^{238}U activity concentrations have decreased between 2009–10 and 2011–12. Our data also indicate a decrease of total ^{238}U activity concentrations, both within a wet season and across successive wet seasons. The ratio of the total to dissolved ^{238}U uranium activity concentrations measured over time decreases, as the amount of suspended sediment in the runoff from the landform decreases compared to the first wet season. However, the particulate matter's ^{238}U activity concentration (in Bq kg^{-1}) increases. This increase most likely reflects that, although the amount of suspended sediment decreases, it consists of relatively more fine particulates which generally show higher activity concentrations of uranium-series elements than the coarser sediments (Bollhöfer et al. 2011). Activity concentrations in the suspended sediment are up to 10 times higher than those measured in the substrate (Bollhöfer & Doering 2013).

Table 1 shows the results of activity concentration measurements of uranium-series radionuclides in the dissolved, residual, total and particulate fractions in surface water runoff from erosion plots 1 to 4. Samples were collected at the end of the wet season 2009–10. In addition, activity ratios of U:Ra:Po are shown.

The average ratio U:Ra:Po for runoff from a waste rock cover in the dissolved fraction was 138(*27*):19(*4*):1 (*numbers in brackets represent the standard error*), illustrating a substantial disequilibrium between uranium-series elements in the dissolved phase with ^{238}U activity concentrations being on average 138 times and ^{226}Ra being 19 times greater than ^{210}Po activity concentrations. This disequilibrium is much smaller in runoff from the waste rock - laterite erosion plots with a U:Ra:Po of 21(*2*):7(*2*):1. The difference of uranium-series element activity concentrations measured in the dissolved phase of runoff from the waste rock treatments shows that uranium is more readily dissolved than radium or polonium. Differences in activity concentrations in the particulates themselves are much smaller with ^{238}U being on average twice as high as ^{226}Ra and ^{210}Po activity concentrations.

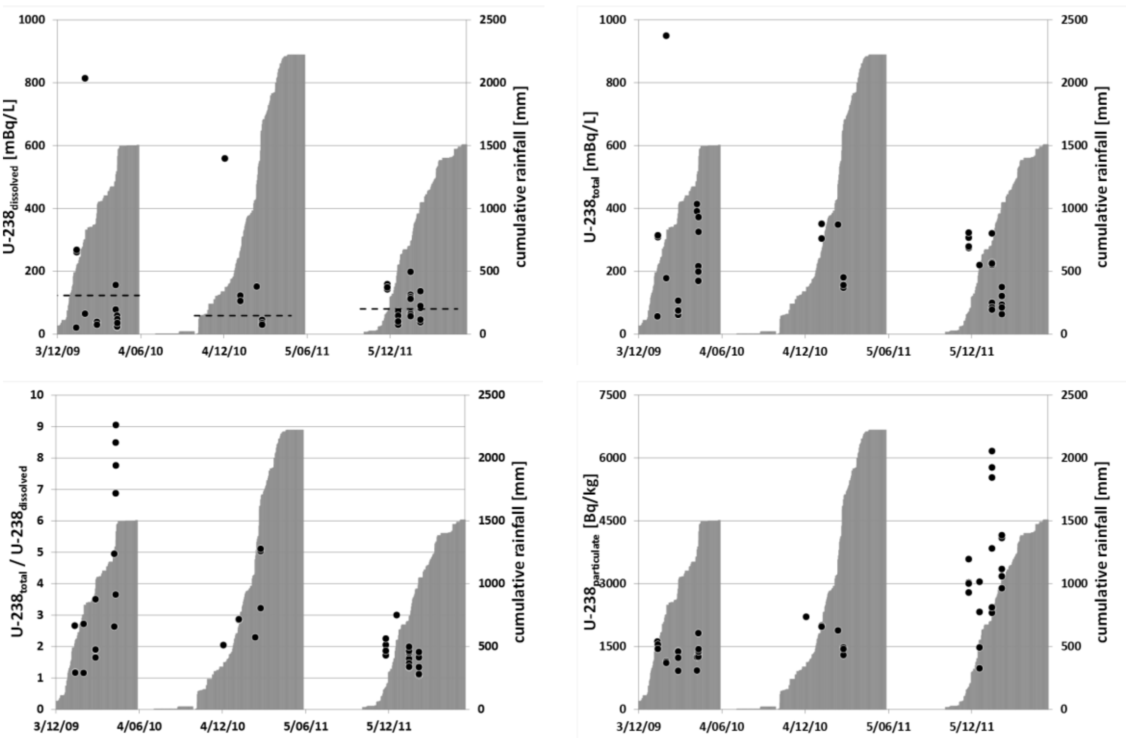

Figure 1 Activity concentration of ^{238}U in the dissolved (mBq $L^{-1}{}_{d}$) and total (mBq $L^{-1}{}_{tot}$) fractions, their ratios, and the sediment activity concentration (Bq $kg^{-1}{}_{p}$) in surface water runoff from erosion plot 1, collected over three successive wet seasons. The dashed lines show the average dissolved ^{238}U activity concentration over the three wet seasons as reported in Saynor et al. (2013). The grey area gives the magnitude of the cumulative rainfall over one wet season.

Table 1 Activity concentrations of uranium-series radionuclides in the dissolved (mBq $L^{-1}{}_{d}$), residual (mBq $L^{-1}{}_{res}$), total (mBq $L^{-1}{}_{tot}$) and particulate (Bq $kg^{-1}{}_{p}$) fractions in surface water runoff from erosion plots (EP) 1 to 4, collected in April 2010, and activity ratios of U:Ra:Po.

Sample ID	Date		mBq/L_d	mBq/L_{res}	mBq/L_{tot}	Bq/kg_p	$(U:Ra:Po)_d$	$(U:Ra:Po)_p$
TL10001 (EP1)	9/04/10	^{238}U	157±16	258±26	414±30	1254±125	126:20:1	2.6:1.4:1
		^{226}Ra	24.6±1	136±5	161±5	662±24		
		^{210}Po	1.24±0.18	100±4	101±4	487±21		
TL10002 (EP1)	9/04/10	^{238}U	79±8	313±31	392±32	931±93	68:9:1	1.8:1.5:1
		^{226}Ra	10.6±0.6	253±7	264±7	754±21		
		^{210}Po	1.16±0.2	170±9	171±9	507±28		
TL10003 (EP1)	13/04/10	^{238}U	48±5	325±32	372±33	1384±138	43:5:1	1.6:1.3:1
		^{226}Ra	5.3±0.5	257±11	262±11	1095±47		
		^{210}Po	1.11±0.21	197±11	198±11	839±46		
TL10004 (EP1)	13/04/10	^{238}U	36±4	290±29	325±29	1440±144	222:26:1	2.0:1.8:1
		^{226}Ra	4.2±0.4	259±8	263±8	1286±40		
		^{210}Po	0.16±0.27	143±6	143±6	711±32		
TL10005 (EP2)	9/04/10	^{238}U	147±15	735±74	882±75	3287±329	130:13:1	2.5:1.7:1
		^{226}Ra	14.6±0.7	480±18	495±18	2146±80		
		^{210}Po	1.13±0.29	291±11	292±11	1299±47		
TL10006 (EP2)	12/04/10	^{238}U	350±35	385±39	735±52	3870±387	170:28:1	3.9:1.1:1
		^{226}Ra	58.7±2.9	111±5	170±6	1115±50		
		^{210}Po	2.06±0.42	99±4	101±4	991±38		
TL10007 (EP2)	12/04/10	^{238}U	220±22	522±52	742±57	4116±412	207:29:1	3.3:1.3:1
		^{226}Ra	31±1.3	200±7	231±7	1578±55		
		^{210}Po	1.06±0.4	158±5	159±5	1245±40		
TL10008 (EP3)	9/04/10	^{238}U	20±2	669±67	689±67	1588±159	22:9:1	1.5:1:1
		^{226}Ra	7.9±0.4	452±18	460±18	1073±43		
		^{210}Po	0.92±0.27	444±35	445±35	1054±84		
TL10009 (EP3)	9/04/10	^{238}U	20±2	728±73	748±73	1628±163	18:7:1	1.7:1.1:1
		^{226}Ra	7.4±0.5	480±19	487±19	1073±42		
		^{210}Po	1.1±0.33	434±29	435±29	970±64		
TL10010 (EP3)	9/04/10	^{238}U	21±2	588±59	609±59	1601±160	29:13:1	1.7:1.1:1
		^{226}Ra	9.2±0.5	392±8	401±8	1067±22		
		^{210}Po	0.72±0.32	343±20	344±20	934±55		
TL10011 (EP4)	12/04/10	^{238}U	29±3	210±21	239±21	1544±154	18:4:1	1.6:1.2:1
		^{226}Ra	5.7±0.6	156±7	162±7	1147±51		
		^{210}Po	1.58±0.37	135±5	136±5	989±40		
TL10012 (EP4)	12/04/10	^{238}U	28±3	231±23	259±23	1777±178	16:3:1	1.6:1:1
		^{226}Ra	5.3±0.7	145±5	150±5	1117±39		
		^{210}Po	1.7±0.46	144±6	146±6	1113±45		

Using the U:Ra:Po presented above for the dissolved phase in runoff from the waste rock erosion plots 1 and 2, allows an estimation of the average ^{226}Ra and ^{210}Po activity concentrations in runoff from erosion plot 1, using the average ^{238}U activity concentrations for the wet seasons 2009–10, 2010–11 and 2011–12 presented in Saynor et al. (2013). These average activity concentrations amount to 0.9±0.2 mBq L^{-1} (2009-10), 0.4±0.1 mBq L^{-1} (2010-11) and 0.6±0.1 mBq L^{-1} (2011–12) for ^{210}Po and 16.0±1.2 mBq L^{-1}, 7.5±0.6 mBq L^{-1} and 10.2±0.8 mBq L^{-1} for ^{226}Ra.

Steps for completion

Six additional samples for the three wet seasons have been analysed via ICP-MS but results were not available at the end of the 2013–14 reporting period. A future report will include these data. Additional measurements will be conducted in event-triggered samples collected from erosion plot 1 in the 2014-15 wet season to determine whether the ratio of U:Ra:Po in the dissolved fraction has changed or can be used to estimate average ^{226}Ra and ^{210}Po activity concentrations from dissolved U in the runoff water samples.

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Key Knowledge Need 2.5: Ecosystem establishment

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

C Humphrey & L Chandler

Background and progress to date

A key component of rehabilitation and closure planning for Ranger is the development of closure criteria. These measurable and quantifiable benchmarks are the targets against which the long term success and sustainability of rehabilitation will be measured. They will serve as the basis for issuing of a close-out certificate (ERA 2014).

For key water quality variables, closure criteria for receiving water environments are being developed using local laboratory toxicity and field biological effects information. A field-effects based approach for developing criteria for magnesium sulfate ($MgSO_4$), electrical conductivity (EC, a reliable surrogate and correlate of $MgSO_4$) and uranium (U), for waterbodies within and outside of the Ranger Project Area (RPA), has been underway since 2005. The responses of macroinvertebrate communities in Georgetown Billabong (GTB) are being used as a basis for closure criteria development. Specifically, there are periods of the water quality record in GTB where the biological condition of the billabong, as measured by macroinvertebrate community diversity and abundance, is similar to reference, yet water quality is mine-influenced. Suitable characterisation of the altered water quality supporting this ecological condition over time can be used to derive closure criteria, in accordance with the principles applied in ANZECC & ARMCANZ (2000).

Sampling of macroinvertebrate communities in GTB in 2011 showed, for the first time, significant adverse changes in communities relative to reference waterbodies[1], associated with a decline in water quality (due to an increase in $MgSO_4$ concentrations) that had been underway since 2006 (Humphrey et al. 2013). This observation, if confirmed, provides a threshold of biological change across a water quality disturbance gradient. Such a threshold provides an additional, potentially complementary, approach for closure criteria setting, i.e. one that is no longer constrained to water quality ranges supporting 'no biological effects'. Information about a contaminant-effect concentration may enhance options for developing closure criteria for waterbodies within and outside of the RPA, where the Environmental Requirements (ERs) are framed differently. Within the RPA, which includes GTB and Coonjimba Billabong (CJB), the ERs state that impacts must remain "as low as reasonably achievable" during and after mining, whereas outside of the RPA, rehabilitation goals are directed at ensuring ecosystem condition is consistent with similar sites in Kakadu National Park (ERA 2014).

In response to 2011 results in GTB, sampling was repeated during the wet-dry season transition period (between April and May) of 2013 in the same 13 waterbodies as sampled in most previous years (i.e. 1995, 1996, 2006, 2009 and 2011). This sampling followed an antecedent period (2012 and early 2013) of high EC values in GTB similar to 2011; the sampling aimed to confirm the 2011 results, or at least determine whether the responses observed in 2013 were consistent with the Mg-response relationship developed to 2011 (Figures 4 & 5 from Humphrey et al. 2013). Opportunity was also taken at this time to more fully assess other possible causes of biological

[1] Reference waterbodies include Gulungul, Baralil, Corndorl billabongs and Jabiru Lake in Magela Creek catchment, Wirnmuyurr, Malabanjbanjdju, Anbangbang, Buba and Sandy billabongs in Nourlangie Creek catchment.

changes in GTB in 2011, including possible habitat changes (viz aquatic plant communities) and differences in sample processing methods since 1995. These collective studies were used to assess the weight of evidence for possible minewater effects upon GTB macroinvertebrates observed in 2011.

Consistency in biological response to contamination from minewaters

Sampling in 2013 was used to assess whether the additional results supported, or were consistent with, observations made to 2011. The sampling of water quality and macroinvertebrate communities in waterbodies in 2013 followed the same methods as used in 2011 – see Humphrey et al. (2013).

Water quality

Seasonal, antecedent (to sampling) median U concentrations for sampling years 1995, 1996, 2006, 2009, 2011 and 2013 are provided in Table 1. The concentrations of mine-water-derived U in GTB between 1995 and 2013 have remained well below the guideline value for protection of local aquatic organisms (van Dam et al. 2012) (Table 1).

Table 1 Unless indicated, median EC, Mg and U values in GTB for recent wet and dry seasons relevant to macroinvertebrate community sampling.

Season or year	EC (µS/cm)	N	Mg (mg/L)	N	U (µg/L)	N
Antecedent wet season						
1995	19.7	5	1.03	1	0.25	5
1996	27.0	5	0.98	1	0.35	7
2006	33.8	31	2.25	31	0.64	31
2009	54.0	30	4.35	30	0.3	30
2011	38.1	31	2.75	31	0.4	31
2013	64.0	32	3.9	31	0.23	31
Antecedent dry season						
1995	88.4	8	2.94	3	0.9	9
1996	59.2	9	2.16	4	0.51	8
2006	69.0	5	1.85	5	0.37	5
2008	91.6	11	5.88	11	0.46	11
2010	128.3	25	8.60	25	0.38	25
2012	95.1	35	4.65	30	0.46	23
Average of wet and dry season median values						
1995	54.1		1.98		0.57	
1996	43.1		1.57		0.43	
2006	51.4		2.05		0.5	
2009	72.8		5.11		0.38	
2011	83.2		5.68		0.39	
2013	79.6		4.28		0.34	
Laboratory derived guideline	42		3		6	

Plots of the median of (generally) weekly EC values measured in GTB since 1991 are shown in Figure 1 for the wet season (January to May inclusive) and dry season (June to December) months. Humphrey et al. (2013) reviewed briefly the EC and associated Mg water quality record from the early 1980s to May 2011, noting that water quality associated with sampling in 2009 and 2011 approached or exceeded EC and Mg guidelines derived from laboratory toxicity testing (van Dam et al. 2010, see Table 1). With the inclusion of 2013 data, it is evident that water quality improved slightly in GTB, with $MgSO_4$ contamination lower than that reported in 2009 and 2011, but (Mg) still exceeding the locally-derived guideline (Table 1).

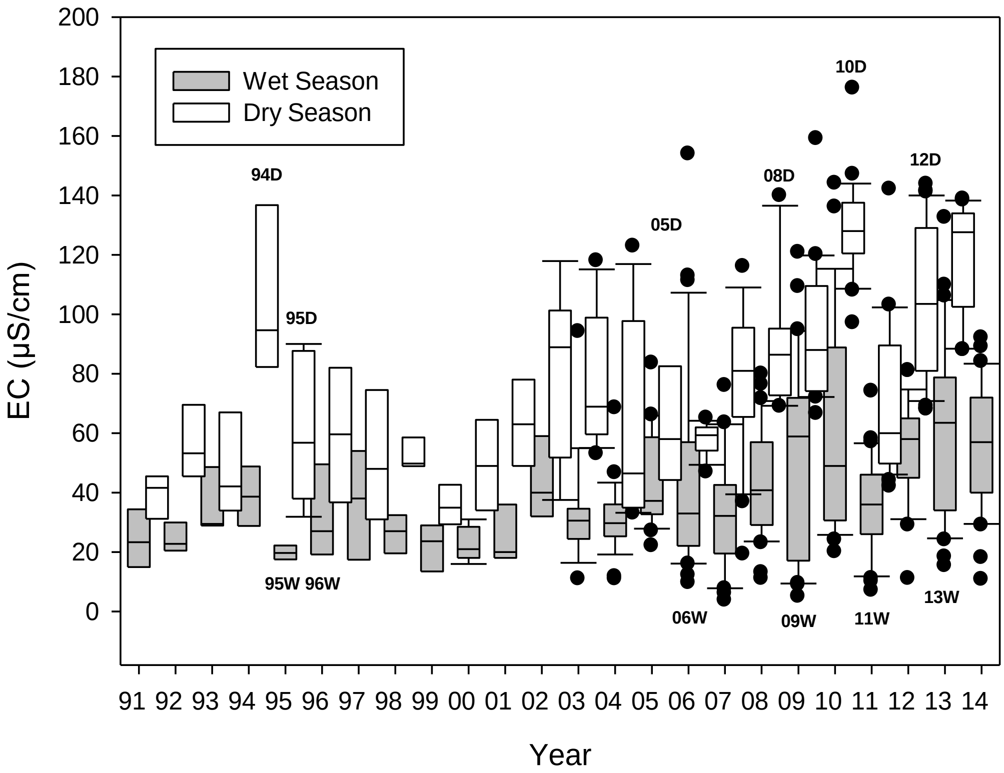

Figure 1 Summary box plots of generally weekly electrical conductivity (EC) values measured in GTB between 1991 and 2014. Box plots show median, range, 25th and 75th percentiles and outliers (points). Periods relevant to macroinvertebrate sampling are indicated by year and season (wet (W) or dry (D). Data from Energy Resources of Australia, NT Dept of Mines & Energy, and Supervising Scientist. Wet season – January to May. Dry season – June to December.

Macroinvertebrate communities

The statistical analysis methods of Humphrey et al. (2013) for macroinvertebrate data were applied, with the addition of 2013 data. Key methods for assessing the response of macroinvertebrate communities in GTB to water quality changes entailed plots of taxa number (relative to reference waterbodies) and Analysis of Similarity (ANOSIM) R values for each year of sampling, against antecedent (to sampling) Mg concentration. (For the years of sampling, these Mg values appear under "Average of wet and dry season median values" from Table 1.)

For each year of sampling and for each GTB replicate, the ratio (as a percent) of taxa number relative to the mean of 9 other randomly-selected replicate taxa number values corresponding to the 9 reference waterbodies, was determined. Successive taxa number ratios were calculated using a random-without-replacement approach. The mean (±SE) of the 5 ratios for each year of sampling was used in subsequent plots.

ANOSIM, effectively an analogue of the univariate ANOVA, compares the observed differences *between* groups – in this case exposure type, GTB (5 replicates per year), versus reference waterbodies (up to 45 replicates per year) – with the differences among replicates *within* the groups. The degree of separation between groups is denoted by the R-statistic, where R-statistic > 0.75 = groups well separated, R-statistic > 0.5 = groups overlapping but clearly different, and R-statistic < 0.25 = groups barely separable (Clarke & Gorley 2006). A significance level of < 5% = significant effect/difference. Possible mine-related effects upon community structure in GTB would be inferred if ANOSIM in any particular year depicted well-separated groups (i.e. GTB vs reference waterbodies).

For 2009 sampling, only one replicate sample was collected from each of the reference waterbodies (compared to five replicate samples in other years) (see Humphrey et al. 2013). While taxa number comparisons would not be expected to be as sensitive to sample size differences, similarity-based comparisons such as ANOSIM are sensitive to both small sample size and discrepancies in sample size (see below) and for this reason the regression relationships and subsequent analyses based on similarity do not include 2009 data.

Mean GTB taxa number, relative to reference waterbodies, for each year of sampling is plotted with antecedent (average of wet and dry season median) Mg concentration in Figure 2. ANOVA testing found no significant difference amongst the mean values for the different years (P = 0.07), though a decline in taxa number is evident in the data at Mg concentrations between 4.3 (2013 sampling) and 5.7 (2011) mg/L. Data from 2013 are consistent with the plotted data from previous years (Figure 2).

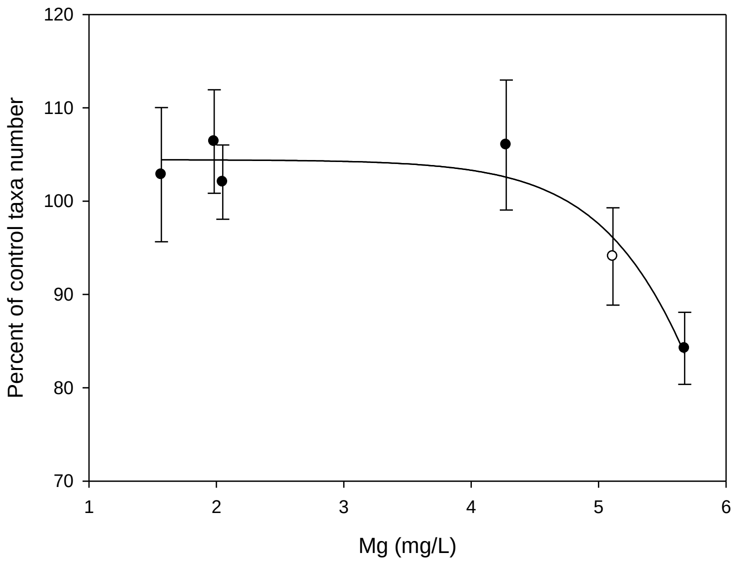

Figure 2 Mean (±SE) macroinvertebrate taxa number (as % of mean reference waterbody taxa number) in GTB in relation to median antecedent wet and dry season Mg concentration. (Year of sampling can be inferred from Table 1.) Open symbol indicates 2009 sampling where sampling effort in reference waterbodies was reduced compared to other years. Line fitted according to sigmoidal model.

The ANOSIM results for GTB relative to reference waterbody community structure are plotted with antecedent Mg concentration in Figure 3A. The ANOSIM R statistics were generally low and near the criterion defined above as 'barely separable' for all years except 2011. For these years, the macroinvertebrate communities of GTB are not regarded as different from those in adjacent reference waterbodies in the region. For 2011, however, ANOSIM results (R >0.5, P <0.05) indicate that GTB samples are clearly and significantly separated from reference waterbody samples (Figure 3A).

ANOSIM R values are sensitive to the pattern of variability amongst samples, including the presence of outliers. Negative R values (1996 and 2013 sampling years, Figure 3A) are characteristic of very patchy assemblages with similar amounts of variability observed within each of the groups (Chapman & Underwood 1999). Thus the scatter of sampling years in Figure 3A, while identifying clear group separation in 2011, is not informative in depicting a possible response-concentration relationship. A different pairwise (GTB-reference) comparison of community structure was examined, using the Bray-Curtis similarity index. An index value of 1 indicates macroinvertebrate communities identical in structure while a value of 0 indicates totally dissimilar communities, sharing no common taxa. For each year of sampling and for each GTB replicate, mean similarity between community structure data for that replicate and those for 9 other randomly-selected replicates corresponding to the 9 reference waterbodies, was determined. Successive similarity values were calculated using a random-without-replacement approach. The mean (±SE) of the 5 similarity values for each year of sampling was used in subsequent plots and analyses.

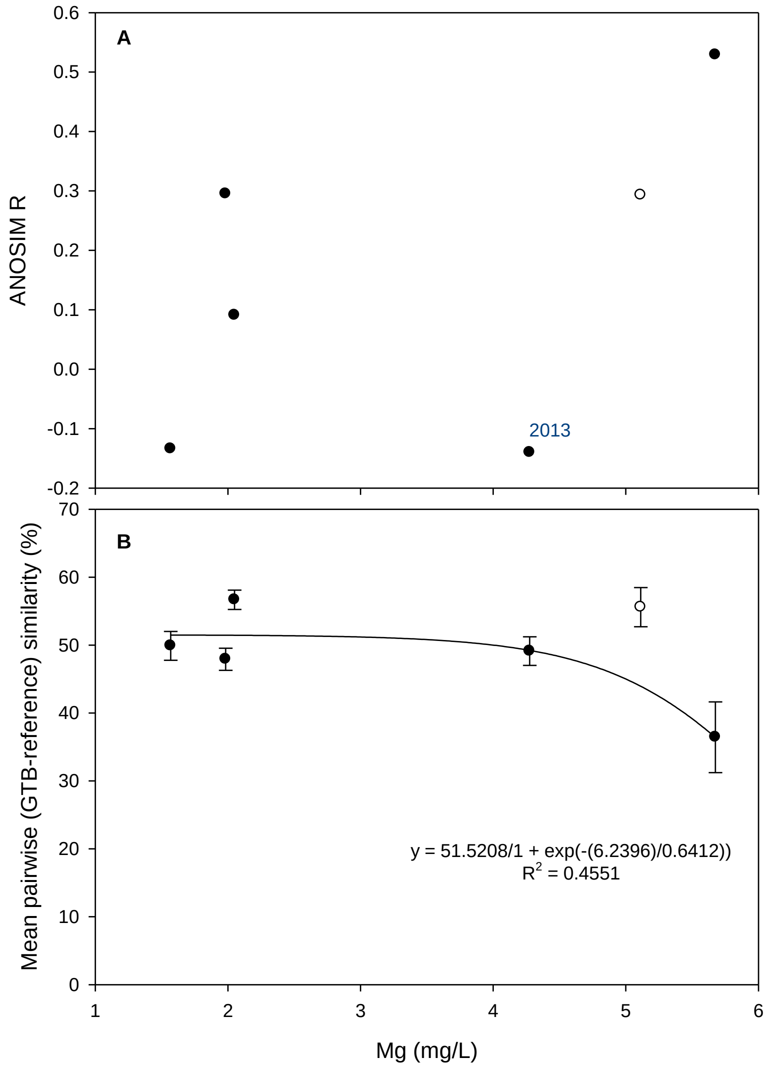

Figure 3 GTB vs reference waterbody a) ANOSIM R values and b) mean (±SE) replicate pairwise Bray-Curtis similarity, in relation to median antecedent wet and dry season Mg concentration. Open symbol represents 2009 sampling year and ANOSIM value for 2013 sampling year also indicated.

Mean similarity for the GTB-reference waterbody community structure comparison for each sampling year is plotted with antecedent Mg concentration in Figure 3B. ANOVA testing found a significant difference amongst the mean similarity values for the different years ($P < 0.001$), with Tukey's multiple comparison test showing significant differences ($P < 0.05$) between 2011 and each other sampling year apart from 1995 (not significant), but not between any other pairwise sampling year comparison. These results are consistent with those derived for taxa number, indicating significantly different community structure in GTB relative to reference waterbodies in 2011. Similarly, a decline in similarity is evident in the data at Mg concentrations between 4.3 (2013 sampling) and 5.7 (2011) mg/L. As shown for taxa number, 2013 data are consistent with the plotted data from previous years (Figure 3B).

The biological response data for GTB are further considered below ('Towards deriving closure criteria for Mg, EC and other solutes for Ranger receiving waters').

Assessment of other possible causes of biological changes in GTB

Other factors have potential to cause the biological changes observed in GTB in 2011, including differences in sample processing methods since 1995 and habitat changes (viz aquatic plant communities). These are assessed in turn below.

Differences in sample processing methods since 1995

For ARR waterbodies sampled in the present study, different sample processing methods have been applied over time. In 1995, 1996 and 2006, live macroinvertebrates were extracted from samples by eye in the field – so-termed live-sorting. In 2009, 2011 and 2013, the processing method differed; samples were preserved in the field and later subsampled and sorted in the laboratory under a stereo microscope, i.e. laboratory-processed samples. This change in methodology has potential to confound results observed to date, with a key question to address: are the changes in GTB macroinvertebrate communities observed after 2006 (particularly for 2011) an artefact of the method change?

The most effective way to assess potential confounding of this type is to compare analyses of community structure datasets derived (i) from the samples first live-sorted in field, then (ii) from the same sample residues preserved and later subsampled and sorted in the laboratory. Some limited local datasets were available for this from specific investigations conducted in GTB and/or Gulungul Billabong (GUL) in 1996 and 2013 (18 samples). A much larger dataset, however, was also available from the results of an extensive, Australia-wide AUSRIVAS bioassessment study, acquired from the mid to late 1990s (80 reference-site samples).

As part of the Commonwealth's National River Health Programme, the Monitoring River Health Initiative (MRHI) was established in the mid 1990s, to develop a bioassessment approach in Australia using riverine macroinvertebrate communities. The MRHI was the precursor to development of regional AUSRIVAS predictive models for bioassessment of Australia's rivers. ***eriss*** and a collaborator conducted a QA/QC study for the MRHI in the late 1990s with results reported in Humphrey et al. (2000). A key objective of the QA/QC was to assess the performance of operators sorting macroinvertebrate samples either live in the field or from preserved subsamples using a microscope in the laboratory. ***eriss*** staff received a large number of preserved macroinvertebrate samples, representing a variety of different riverine habitats, from different State and Territory agencies. Most agencies (QLD, NSW, VIC, TAS & WA) adopted live-sorting as the sample processing protocol, and assessment of operator performance involved:

(i) further processing in the laboratory of the preserved residues from field live-sorted (LS) samples using microscopic examination of residue subsamples; and
(ii) comparison of community structure data from live-sorted samples and their associated laboratory-processed component. The latter component was termed the 'whole sample

estimate' (WSE), i.e. results that would have arisen had the same (live-sorted) sample been sub-sampled and processed in the laboratory.

A key aspect of the comparative assessment in (ii) above, was whether operators were missing taxa when live-sorting, as a consequence of the cryptic and small size of specimens. The most significant findings from the QA/QC were: (a) small and/or cryptic taxa were commonly overlooked during live-sorting (chironomid pupae, ceratopogonids, empidids, elmid larvae, hydroptilids, oligochaetes, Acarina), while conversely (b) some taxa were better represented in live-sort data, including large or mobile, but less abundant taxa (odonates, shrimps and adult beetles) (Humphrey et al. 2000).

While the MRHI QA/QC study determined that sample processing biases resulted in (subtly) different community structure data for the two sample processing methods, the important aspect to address is whether group comparisons (in this instance GTB versus reference waterbodies) give similar or different results when either live-sorted or laboratory processed data are analysed. If GTB versus reference waterbody comparisons, by way of taxa number, ANOSIM R or Bray-Curtis similarity, are very similar for live-sorted or associated laboratory processed data, this indicates sample processing method is *not* a confounding factor in the interpretation of results from the full dataset (1995 to 2013) used in the present study.

To mimic the GTB versus reference waterbody comparison that is conducted separately for each year of the present study, the LS and associated WSE community structure data from local and MRHI datasets were assessed:

(i) ANOSIM R and Bray-Curtis similarity were calculated between different sample groups from the larger sample pool using both the LS and the associated WSE data, i.e. LS-to-LS and WSE-to-WSE comparisons. LS and equivalent WSE results from the various group comparisons were then compared.

(ii) Taxa number was derived for LS and WSE components of each MRHI and local (ARR) sample. A mean LS/WSE taxa number ratio was derived for sample groups of interest.

For ANOSIM R and Bray-Curtis similarity response measures, two types of group to group comparison by processing method (i.e. LS-LS and WSE-WSE) were applied: (a) all possible pairwise combinations of MRHI state to state (e.g. QLD vs VIC, QLD vs WA, NSW vs TAS, etc), and (b) habitat from one state compared to the same habitat from all other states combined (e.g. QLD macrophyte vs macrophyte from all other states, NSW riffle vs riffle from all other states, etc). The habitat comparison described in (b) was most relevant to the present study where replicates from one billabong (GTB) are compared to replicates from all reference billabongs combined because the sample numbers were similar (typically 4-6 'replicates' from one state versus ~20 or more replicates from all others states) and the local (ARR) macrophyte habitat dataset described above (GTB vs GUL) could be directly assessed in the same analysis. State to state and within-habitat relationships between ANOSIM R values calculated on live-sort or whole-sample-estimate data are shown in Figure 4.

A median LS/WSE taxa number ratio was also derived for each state and local dataset. These results are shown in Figure 5.

Despite taxa biases and ensuing differences in macroinvertebrate community structure data represented in the two sample processing methods, pairwise ANOSIMs from the same method (LS-LS and WSE-WSE) yield very similar results (Figure 4). Indeed, the plotted relationships for state-to-state and within-habitat comparison are almost a 1:1 relationship. Macrophyte habitat data for the comparison of five GTB with five GUL replicates processed using both methods are plotted in the within-habitat relationship (Figure 4b). The ANOSIM R values for the GTB-GUL processing method comparisons are also very similar (R values of 0.52 and 0.55 for LS and WSE methods respectively).

Processing method comparisons using the Bray-Curtis similarity measure were also conducted. The results are not provided here but these showed a slightly higher correlation between LS and WSE groupwise comparison, i.e. very high concordance between the results for each sample processing method.

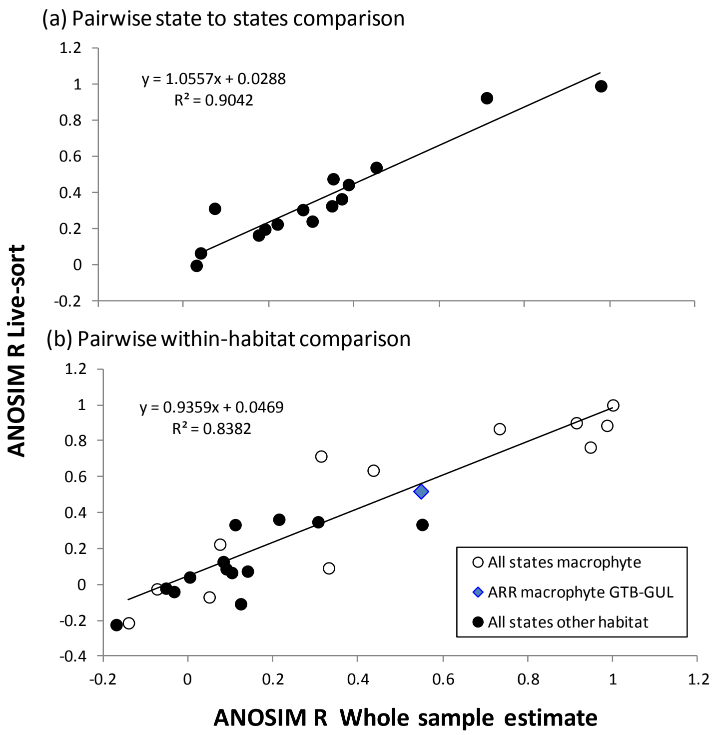

Figure 4 (a) State to state and (b) within-habitat relationships between ANOSIM R values calculated on live-sort or whole-sample-estimate data.

Taxa number comparisons between the two sample processing methods (Figure 5) showed a LS/WSE ratio of close to 1 for all state and local groups considered. This is despite the fact that taxa represented in LS and WSE components were not always the same.

These collective results show that groupwise comparisons of (i) macroinvertebrate community structure and (ii) taxa number, are preserved, whichever of the two sample processing methods is used. Thus, there is no evidence that the GTB-reference waterbody ANOSIM, similarity and taxa number values are confounded by the change in sample processing method that occurred after 2006.

Changes in aquatic vegetation amongst waterbodies over time

Macroinvertebrate communities may respond to changes in habitat and for GTB and similar ARR lentic waterbodies, aquatic plant communities constitute the major habitat for residency. Assessment of aquatic vegetation composition and relative abundance has accompanied billabong macroinvertebrate sampling for every year of the present study, using the same methodology first developed in 1995. Changes in these plant communities were examined to assess the extent, if any, to which these changes may also account for macroinvertebrate responses in GTB over time.

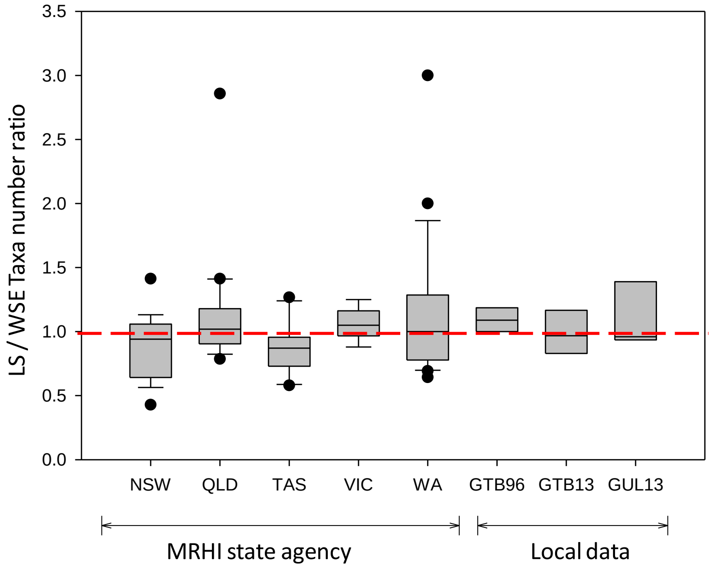

Figure 5 Boxplot showing comparison of LS-WSE (live-sort/whole sample estimate ratio) for the different MRHI state agencies and within GTB 1996 (10 samples), GTB 2013 (5) and Gulungul 2013 (5) samples. Box plots show median, 25th and 75th percentile, range within these interquartiles and outliers (asterisk).

Since 1995 and at each of the five locations within each of the 14 waterbodies sampled, macrophyte composition and relative abundance data have been collected. The visual-assessment methods are described in O'Connor et al. (1996). In summary, a visual assessment of the total percentage cover of submerged and emergent macrophytes in the sampling location is made, as well as the percentage abundance of individual macrophyte taxa (usually genus-level) present. Typically, percentage abundance of the different taxa were grouped according to structurally-similar plant forms, after the schema of Sainty & Jacobs (1994), i.e. 'floating attached', 'submerged not feathery', 'submerged and emergent feathery', 'free floating', 'emergent narrow leaf' and 'emergent broad leaf' forms. GTB plant communities were compared to the same (up to) 9 reference waterbodies as the macroinvertebrate analyses described above. Assessment of aquatic vegetation changes in GTB relative to reference waterbodies entailed comparative plots of relative abundance of structurally-similar plant forms, taxa number and mean (GTB-reference) similarity for community structure data. The taxa number and similarity plots were prepared in the same manner as applied to macroinvertebrate data (i.e. Figures 2 & 3B respectively); taxa number and similarity were plotted against antecedent (to sampling) Mg concentration. Of particular interest in the plots was whether changes in GTB plant communities, relative to reference waterbodies, were most evident in 2011, coincident with macroinvertebrate impacts and highest (on record) $MgSO_4$ contamination.

The analyses showed that, relative to reference waterbodies in 2011, GTB submerged vegetation was in low densities (Figure 6) and the average replicate taxa number was also low (Figure 7). GTB aquatic plant community structure also had low resemblance to that of reference waterbodies in 2011 (Figure 8). However, these same characteristics were also evident for GTB in 1996 and 2006 (Figures 6, 7 & 8), years when GTB macroinvertebrate communities were similar to reference. This provides evidence that key aquatic plant habitat in GTB was not responsible alone, at least, for the relatively low macroinvertebrate diversity observed in 2011.

Towards deriving closure criteria for Mg, EC and other solutes for Ranger receiving waters

Work conducted after 2013 sampling indicates that alternative possible causes, so far considered, of the significantly altered macroinvertebrate communities in GTB observed in 2011 (method and habitat change) cannot explain and account for the changes observed. Additional environmental variables associated with waterbody habitat and location will be sought for analysis to confirm these findings. Notwithstanding, together with the consistency of response observed in 2013 to the response-water quality relationship accrued after 2011 sampling, the increasing $MgSO_4$ contamination observed in GTB up to 2011 appears to be the most likely explanation for the impact observed in that year.

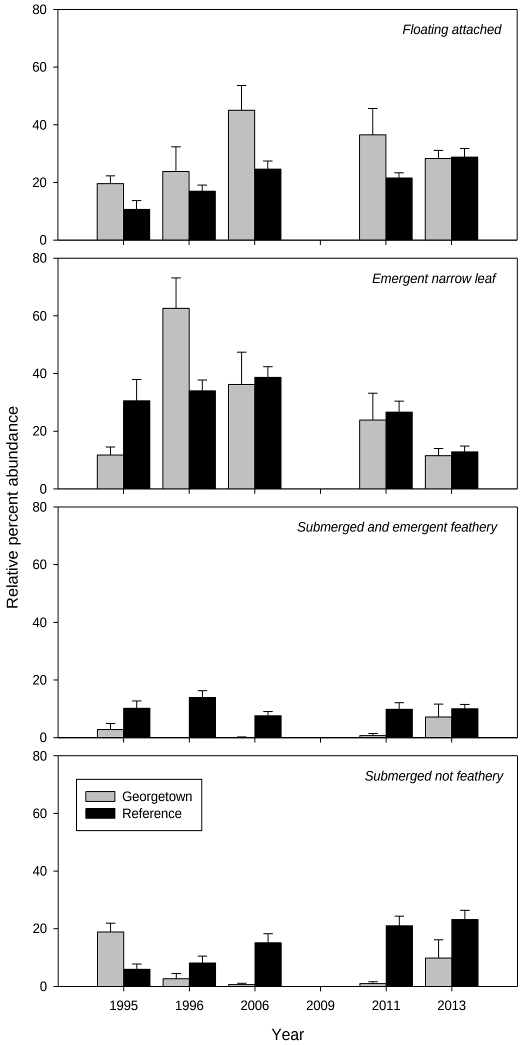

Figure 6 Histograms of mean, relative percent abundance of aquatic plants grouped by life-form in GTB and reference waterbodies for different years of sampling.

Billabong sampling will continue in future years, either periodically (e.g. every other year) and/or opportunistically in response to changes in water quality. The accruing data will be used to confirm the findings to date, while a more detailed examination of the taxa absent from, or in much-reduced abundance in, GTB in 2011, is underway to assess consistency in response to literature-reported sensitivities. Both approaches can be used to strengthen inferences made about water quality changes to biota in GTB.

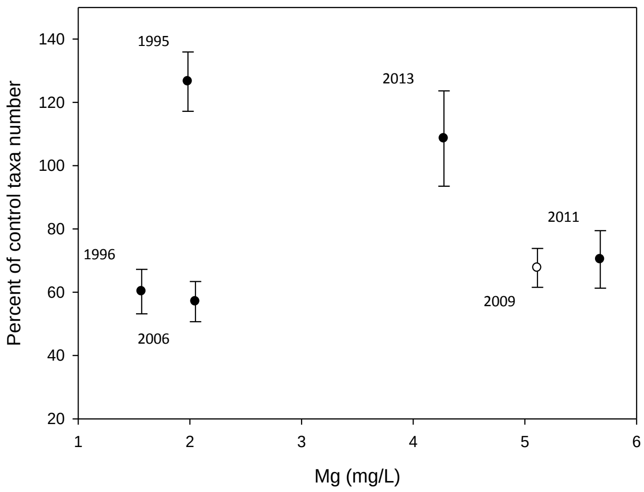

Figure 7 Mean (±SE) aquatic plant taxa number (as % of mean reference waterbody taxa number) in GTB in relation to year of sampling and median antecedent wet and dry season Mg concentration.

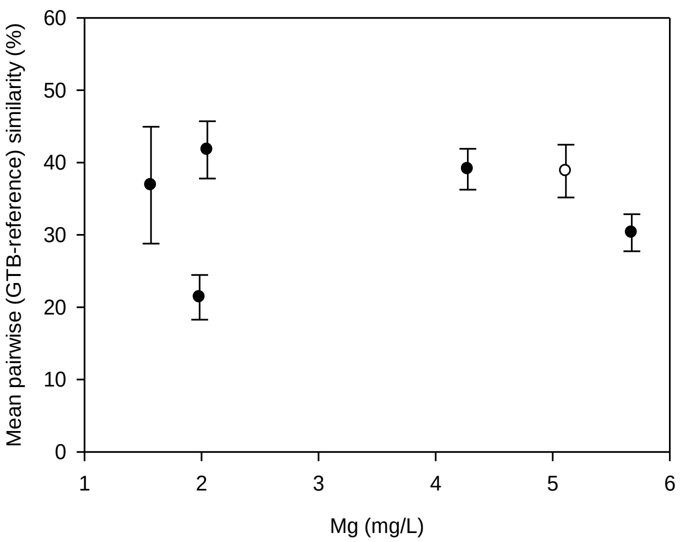

Figure 8 Mean (±SE) replicate pairwise Bray-Curtis similarity for GTB vs reference waterbody aquatic plant comparison, in relation to year and median antecedent wet and dry season Mg concentration.

A number of assemblage-based, statistical approaches for change-point determination are being assessed in a related field experimental, sediment quality criteria study for the ARR (Harford et al. 2013). While these same approaches can also be examined for the present study, the graphical results for GTB taxa number and community structure response measures relative to reference waterbodies appear unambiguous in identifying a change in macroinvertebrate communities at ~5 mg/L Mg (Figures 2 & 3B). This value is very close to the laboratory toxicity-determined guideline value of 3 mg/L, despite field and laboratory approaches using very different methods of derivation. Laboratory and field exposures, for example, are short- (up to 6 days) and longer-term (antecedent 12 months), respectively. Further, the laboratory value is based on a statistical extrapolation compared to the direct threshold determination of the field value. These supporting lines of evidence for closure criteria proposed for Mg and EC are presently being considered by an Aquatic Ecosystems Technical Working Group (AETWG) that reports to the Ranger Closure Criteria Working Group.

The AETWG is considering where the field and laboratory criteria would apply amongst creek channel and billabong receiving-water types. Organisms tested in the laboratory are mainly resident in billabongs, while apart from higher water temperatures observed in billabongs compared to creek channels, wet season water quality in creek channels and billabongs is similar (comparative data available but not provided here). This would suggest field- and laboratory-derived criteria could be applied to all wet season surface waters. For billabongs where surface waters persist throughout the year (unlike creek channels) the very different water quality observed naturally between wet and dry seasons necessitates derivation of two sets of criteria, one for each season. The antecedent dry and wet season (median) water quality values for 1995, 1996, 2006, 2009 and 2013 could serve this role, though decisions are required on how these values would be considered and integrated with those determined from laboratory and field threshold effects discussed above.

Antecedent dry and wet season (median) values for U in GTB for 1995, 1996, 2006, 2009 and 2013 are lower than laboratory-determined guideline values, and support 'no effects'. Values exceed those recorded in reference waterbodies, nevertheless, and so could be considered amongst other U closure criteria being determined for surface waters.

The AETWG is also considering the derivation of different criteria that would apply on and outside of the RPA and whether or what criteria would apply for different phases of the rehabilitation trajectory. This is further considered below.

Acclimation of biota to $MgSO_4$ in other on-site waterbodies

Noting the presence of *Hydra* (and other freshwater species used in ***eriss***' toxicity testing programme) in Coonjimba (CJB) and Ranger Retention Pond 1 (RP1) adjacent to Ranger, at concentrations well above the laboratory-determined guideline, McCullough et al. (2008) sought to determine the basis for this residency. They noted the same responses to Mg in *Hydra* sourced from CJB and a reference billabong (i.e. lack of genetic adaptation in CJB *Hydra*) and with further work, concluded that CJB tolerance was due to Ca amelioration (i.e. Mg:Ca ratio <9:1 in CJB and thereby protective). Apart from ionic protection of this type, similar high tolerances without genetic adaptation are often ascribed as evidence of physiological tolerance (e.g. Weis & Weis 1989) or acclimation.

Further evidence of acclimation in CJB and RP1 macroinvertebrate communities can be shown by plotting, together with the GTB results from Figure 3A, ANOSIM R values for CJB and RP1 comparisons with reference waterbodies with associated antecedent (to sampling) Mg concentrations. These results are shown in Figure 9 where it is evident that group separation in CJB and RP1 are elicited at increasingly higher Mg concentrations, respectively, than for GTB. Thus, the fauna of these waterbodies has a higher tolerance to Mg than GTB fauna. Mg:Ca ratios in GTB, CJB and RP1 are, respectively, ~3.5:1, 8:1 and 7.5:1. Thus, Ca amelioration (i.e. Mg:Ca ratio <9:1) cannot explain this greater tolerance because the prevailing Mg:Ca ratio in GTB has been much less than CJB and RP1 for all sampling years and hence, and according to van Dam etc (2010), should actually confer *greater* protection. Other (unknown) antagonistic ionic interactions in GTB may be interfering with the protective role of Ca. This requires further study.

The different exposures to minewaters for the three waterbodies might also explain the different tolerances of resident macro invertebrate fauna. Greatest to least spatial connectivity of waterbodies to Magela Creek follows the order GTB, CJB then RP1. RP1 has no direct connectivity to Magela Creek while CJB backflows occasionally at high creek flows. GTB both backflows regularly and becomes a channel of Magela Creek itself at high flow events due to its very close proximity to the adjacent creek channel. GTB's exposure to mine-derived waters in the wet season is intermittent due to its closer contact and exchange with flow in Magela Creek, though since 2006, these 'partial-cleansing' events during backflow and high creek-flow events have reduced in frequency. Conversely, CJB and RP1 have constant exposure to minewaters with negligible flushing events. It is possible that tolerance and acclimation of the biota in CJB and RP1 are only maintained by constant exposure to mine wastewaters and this is the explanation for results observed (Figure 9).

Because taxonomic identification of macroinvertebrates in this billabong study is conducted at the family-level, another explanation for the higher tolerances observed in CJB and RP1 is species or genus replacement, i.e. sensitive taxa being replaced by more tolerant taxa from the same family. Only limited species-level identifications have been conducted, most notably Chironomidae in 1995 sampling (O'Connor et al. 1996). This study demonstrated a preference of a few species from the genera *Tanytarsus*, *Larsia* and *Polypedilum* for high EC waters, but none of the 45 chironomid species was unique to either waterbodies receiving mine wastewaters or reference waterbodies with no mine influence. Persistence of taxa in waterbodies receiving Ranger mine wastewaters appears best explained by physiological acclimation.

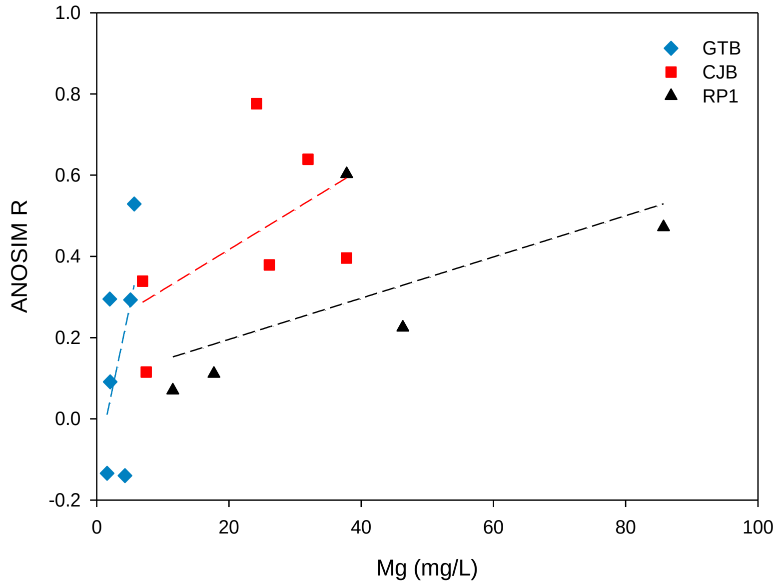

Figure 9 GTB, Coonjimba Billabong (CJB) and Ranger Retention Pond 1 (RP1) vs reference waterbody ANOSIM R values, in relation to median antecedent wet and dry season Mg concentration.

From the results presented in Figure 9, the macroinvertebrate communities of CJB and RP1 in 1995 and 1996, when median annual Mg concentrations approached 20 mg/L (RP1), were similar to reference (interpretation of ANOSIM R values <~0.3; Clarke & Gorley 2006). As a relative percentage of taxa number occurring in reference waterbodies, however, taxa number in both waterbodies for both years, was less than 100% (Figure 10). It is possible that for waterbodies lying within the RPA and constantly exposed to minewaters (including CJB), Mg values in this range may serve as transitional criteria, along a trajectory of continual water quality improvement.

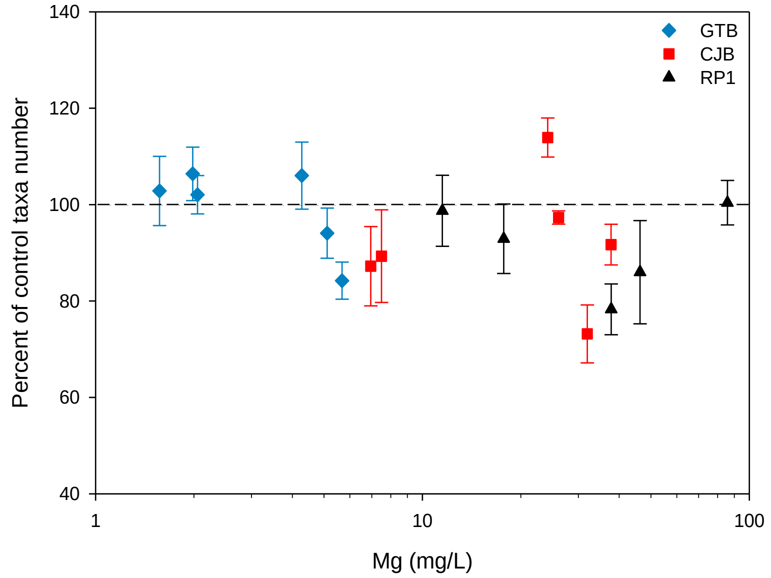

Figure 9 GTB, CJB and Ranger RP1 taxa number, as a percent of control waterbody taxa number, in relation to median antecedent wet and dry season Mg concentration.

Conclusions and further studies

Further sampling is needed to improve the understanding of contaminant-response relationships amongst waterbodies receiving mine wastewaters. The preceding assessments assume different macroinvertebrate responses to minewater contamination in GTB, CJB and RP1. Alternatively, it is possible that responses are a continuum across the range of Mg observed in the three waterbodies, when collective data are plotted together (Figures 9 & 10). Further taxonomic and water quality information may assist in this assessment, and thereby improve understanding of a suitable trajectory-based rehabilitation/restoration model for these systems.

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Development of turbidity closure criteria for receiving surface waters following Ranger minesite rehabilitation

A George, S Mueller[1], S Mitrovich[1] & C Humphrey

Background

Georgetown Billabong (GTB) is a natural waterbody lying in very close proximity to Ranger and is one of the first waterbodies to receive runoff from the minesite. The billabong experiences natural increases in turbidity over the dry season as a consequence of (mainly) wind-induced re-suspension of the silt-clay sediments, exacerbated by evaporation and decreasing water depth. Biological responses associated with these natural increases in turbidity have been studied to assist in developing local water quality criteria for this stressor, including closure criteria for the Ranger minesite during and after rehabilitation.

Biological-effects information are being sought in GTB from the relationship between turbidity and measures of phytoplanktonic biomass and productivity (Buckle et al. 2010, George & Humphrey 2013). In the dry season of 2009 (Buckle et al. 2010) and since the dry season of 2012 (George and Humphrey (in prep), turbidity, chlorophyll-a and other water quality variables have been collected in GTB. The aim of the studies has been to elucidate the range of (natural) turbidity values in GTB that inhibit phytoplankton biomass (indirectly represented by chlorophyll-a) so that these ranges can be used to inform the development of closure criteria. The data show that despite an absence of extreme turbidity events (150+ NTU), an inverse relationship between turbidity and chlorophyll-a is typical when turbidity reaches and exceeds sustained (high) threshold values in the mid-late dry season (George & Humphrey in prep).

These same studies have indicated a peak in chlorophyll-a in GTB during the mid dry season. More recent work by George & Humphrey (2013) found that a short spike in chlorophyll-a was associated with the onset of wet season flows, just after turbidity values had fallen. Such a relationship suggested that phytoplankton biomass may have a secondary, though short, response to rapid changes in turbidity, in the latter case, elicited by a return to higher water clarity with wet season re-wetting and inundation of the billabong.

Phytoplankton identification and enumeration conducted on three samples associated with the spikes in chlorophyll-a from the 2012–13 wet season suggested that the greatest diversity in phytoplankton may correspond to higher turbidity, while also suggesting the lowest diversity was associated with flushing flows and reductions in turbidity (George & Humphrey 2013). The shifts in phytoplankton diversity between the three samples highlighted the need to better ascertain the representativeness of chlorophyll-a (an indirect measure of phytoplankton biomass) as an indicator of biological activity and biodiversity more generally. A broader examination of phytoplankton community structure in GTB was initiated to determine whether relationships were evident between concurrent changes in chlorophyll-a concentrations, turbidity and phytoplankton biovolumes. The results aimed to identify environmental drivers for changes in phytoplankton communities, and where turbidity was implicated, improve knowledge about the responses of phytoplankton communities.

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Methods

Over the past two years, water samples for phytoplankton analyses have been routinely collected in conjunction with monthly and fortnightly field-based sampling (see George and Humphrey (2013). A total of 37 preserved samples collected between November 2012 and April 2014 were provided to the University of Technology Sydney, where phytoplankton were identified and enumerated. Biovolumes were determined by measuring the dimension of 20 cells of each taxon using 400 times magnification. UTS analysed the data using multivariate techniques. In particular, two multivariate ordination techniques, Non-metric Multidimensional Scaling (NMDS) and Canonical Correspondence Analysis (CCA), explored patterns amongst the phytoplankton samples with vectors of taxa and/or environmental variables overlain on the ordinations to explore associations with the samples. Results presented below are the first analyses conducted on the data. Additional analyses exploring some of the relationships presented below will be further developed in 2015.

Preliminary phytoplankton community results

The greatest biovolumes of phytoplankton (cell volume in mm^3/L) corresponded to the mid-late dry season of 2013. Based on biovolume, dinoflagellates were the most abundant taxa assemblage during this period, followed by euglenoids and green algae (Mueller et al. in prep) (Figure 1). Generally, variation and abundance in biovolume were greatest at the 10 cm sampling depth, with biovolumes nearly two-fold greater than at the 50 cm and 100 cm depths (data for depths >10 cm not provided here). Data presented in George & Humphrey (in prep) indicate that the most dynamic zone of activity is the upper 10 cm of the water column and that changes in turbidity are likely to impact phytoplankton communities that are resident at this depth. Life history characterisation of the communites was not included in the preliminary analyses and should be evaluated further to extrapolate the importance of the observed communities and assemblages.

To assess relationships between phytoplankton and turbidity, three turbidity categories were applied, which aligned with turbidity ranges corresponding to changes in chlorophyll-a values (e.g. 30–70 NTU) (George & Humphrey in prep). The ranges applied to the phytoplankton analyses included: <10 NTU, 10-70 NTU and >70 NTU. Ordination techniques illustrate distinct phytoplankton taxa groups associated with each of the three pre-defined turbidity categories (Figure 2), with no overlap between the defined zones. While it appears turbidity has a structuring role for phytoplankton assemblages (Mueller et al. in prep), in addition to its influence on biomass (viz chlorophyll-a concentrations (George & Humphrey in prep), seasonal changes in turbidity are also highly correlated with other environmental variables (not assessed here) and, hence, further analyses are required to confirm the relationships observed in these data.

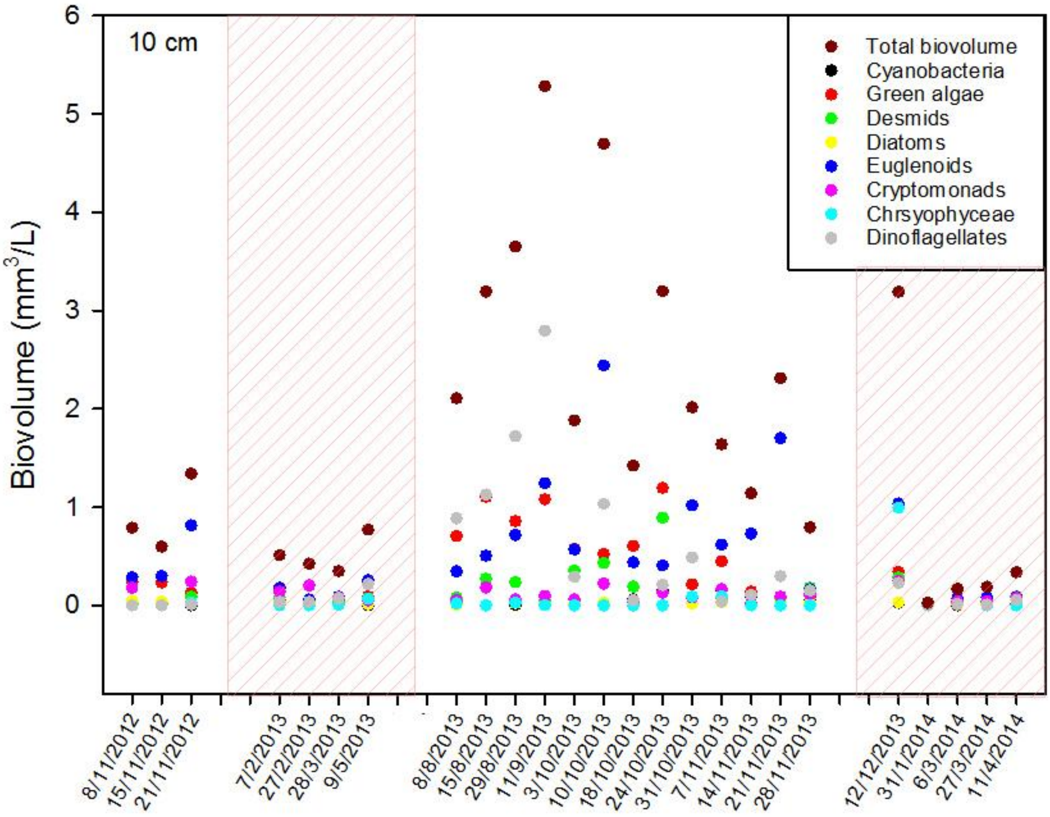

Figure 1 Phytoplankton biovolume change in GTB at the surface (10 cm) between November 2012 and April 2014. Sample dates provided for analyses are indicated by the dates on the X axis. The dates placed closely together are within the dry seasons with dates spaced further apart representing wet season sampling occasions.

Phytoplankton biovolumes were analysed in relation to other water quality variables including nutrients (total nitrogen, total phosphorus, phosphate, ammonium and nitrate), metals, light and dissolved oxygen to elucidate if other water quality variables were confounding the turbidity responses. Field EC data have not yet been analysed in association with phytoplankton data. While trace metals alone did not influence phytoplankton (results not provided here), some samples showed association with particulate concentrations of Mn, Al, U (co-linear variables) and Fe (Figure 3), while also showing a shift in dominance of some taxa related to metals. The significance of these associations is unclear from such preliminary analyses and requires additional consideration. In particular, the association of phytoplankton communities and particulate U will need to be evaluated. The ordination results also suggest that light availability, phosphate and dissolved oxygen were not strong drivers of phytoplankton change (Figure 4).

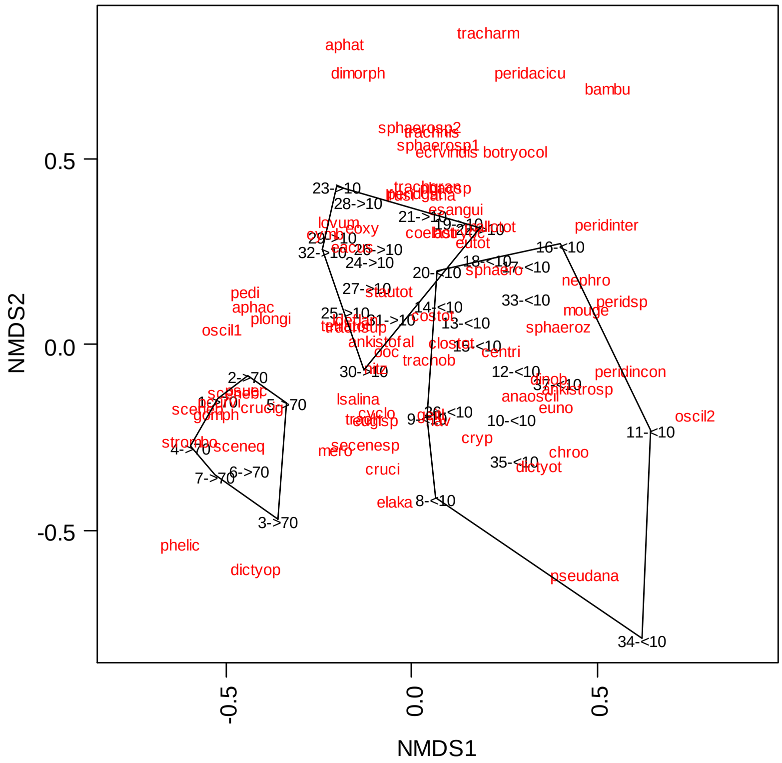

Figure 2 Non-Metric Multidimensional Scaling of GTB phytoplankton samples (black labels representing sample number and turbidity category) overlain with abbreviations of genera and species (in red) (Stress=0.19). The convex hulls (polygons) group the turbidity categories (<10, 10-70 and >70 NTU).

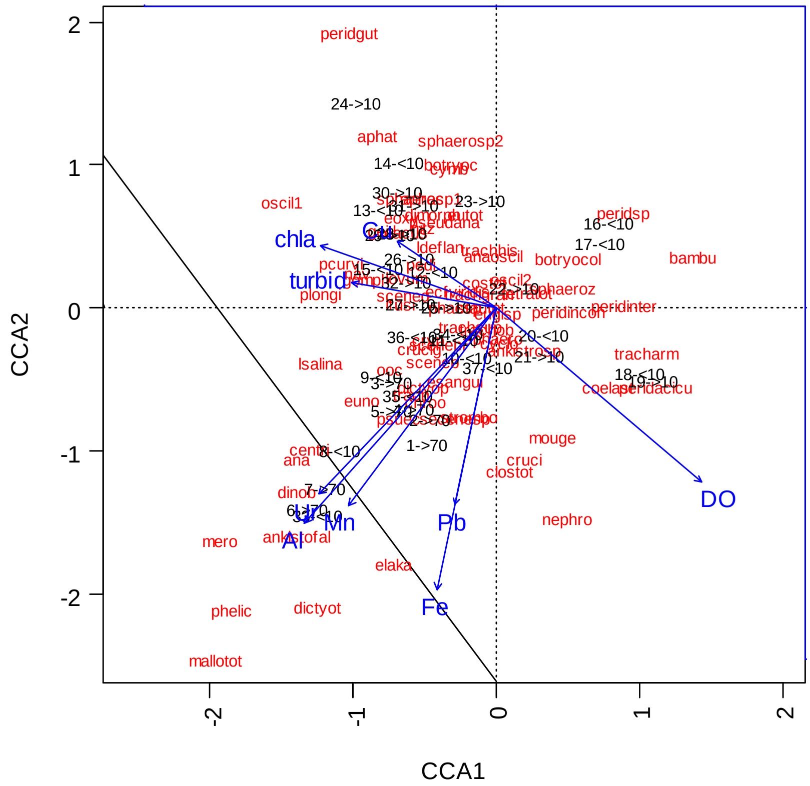

Figure 3 Canonical Correlation Analysis using particulate trace metal concentrations, dissolved oxygen, water temperature, chlorophyll-a and turbidity as constraints. Arrows shows the direction of the environmental gradient, and length of the arrow is proportional to the correlation between the variable and the ordination.

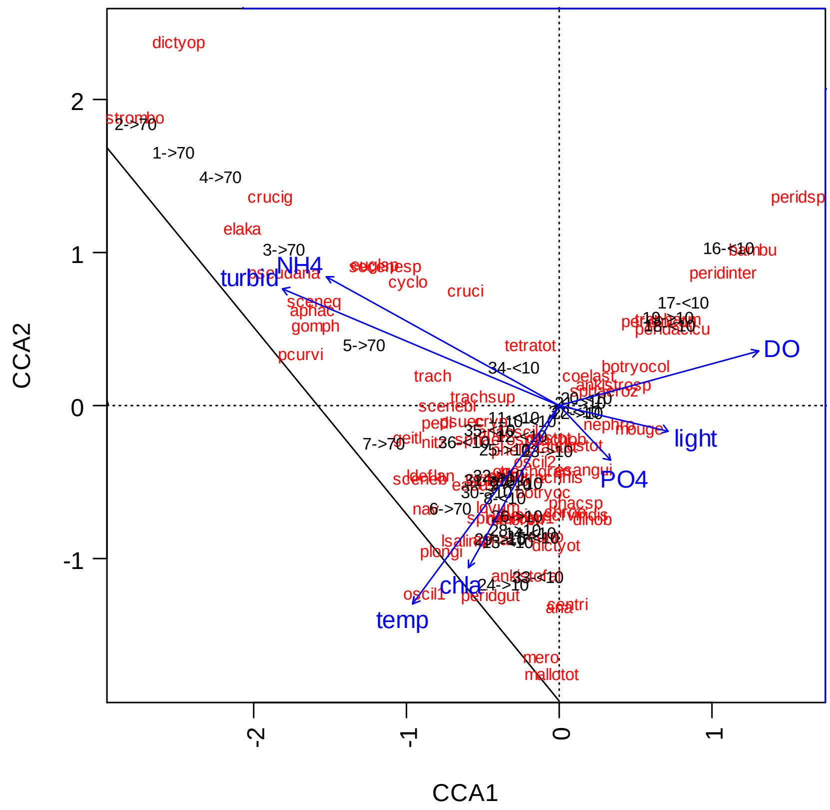

Figure 4 Canonical Correlation Analysis applying water quality parameters. Arrows shows the direction of the environmental gradient, and length of the arrow is proportional to the correlation between the variable and the ordination.

Conclusions

The current phytoplankton analyses appear to provide support for the conclusions from George and Humphrey (in prep) regarding the potential role of turbidity in altering phytoplankton communities in GTB. Initial data analyses have focused on turbidity and phytoplankton community structure relationships. Additional analyses on the responses of phytoplankton to turbidity (and other minewater contaminants) that will inform mine closure and rehabilitation include:

i. Improved assessment of the importance of turbidity relative to other measured environmental variables in structuring phytoplankton communities;

ii. Improved characterisation of particular phytoplankton species and assemblages and evaluation of their responses across turbidity gradients (e.g. confirmation from extended spatial and interannual studies);

iii. Assessment of the responses of phytoplankton communities to other key mine water contaminants (Mg, U, Mn) that may inform the development of closure criteria for these variables;

iv. Identification of any relationships between chlorophyll-a and phytoplankton viz common community summaries, including taxa number and total abundance; and

v. Assessment of whether changes in phytoplankton community structure or in community summaries are occurring at lower turbidity thresholds than currently determined for chlorophyll-a. Such a finding would require a refinement of the currently-proposed closure criteria for turbidity (George & Humphrey, in prep).

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Aquatic ecosystem establishment

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Background

In 2012, a review of the Key Knowledge Needs (KKNs) for the Alligator Rivers Region (ARR) identified that components related to aquatic ecosystems were scattered across a large number of KKNs and project areas, with the result being a diluted strategic approach to rehabilitation of aquatic ecosystems. In response to this, it was proposed during the ARRTC 29 meeting that a targeted KKN or research programme be established to specifically address the needs of aquatic ecosystem rehabilitation. These efforts were deemed to be best met in a re-organisation of KKNs into two themes, provision of services (e.g. contaminant amelioration) and characterisation and functionality. The initial intent was to develop a research programme focused on ecosystem establishment that accorded with the relevant Environmental Requirements. This was to include any necessary rehabilitation of receiving water environments as well as closure criteria development for aquatic environments. For the Ranger Project Area, efforts could include, but are not restricted to: research associated with waterbodies constructed for water treatment; possible construction of, and influences on, permanent waterbodies; rehabilitation, if required, of natural waterbodies; and re-establishment of creeklines. While waterbodies associated with the Ranger Project Area will certainly be included, the full spatial extent of the programme has not yet been determined.

Progress to date

At the outset, a joint project between ERA and SSB within the Aquatic Ecosystem Establishment (AEE) programme, "Aquatic ecosystem establishment knowledge assessment and evaluation", was initiated. This project aimed to review existing available literature on aquatic ecosystems of the Alligator Rivers Region and thereby identify key areas requiring research to meet the rehabilitation expectations and needs for the Ranger minesite. The intent was that the literature review would establish the parameters and future direction of the overarching AEE programme and identify knowledge gaps to provide to the Closure Criteria Working Group (CCWG). The review has since evolved to provide guidance to other parallel activities, including the ecological processes and closure assessments. A review of available literature is ongoing with datasets for aquatic macrophytes being compiled.

A key knowledge gap was identified within the literature review pertaining to aquatic vegetation. The Supervising Scientist has monitored water quality, sediments, macroinvertebrates and fish as key components of aquatic ecosystems, however, aquatic vegetation as a component on its own has not been studied since the 1980s.

The early published flora list compiled in Cowie and Finlayson (1986) encompassed work from the 1970s and 1980s. Even work published during the 1990s is based on data collected for Finlayson et al. (1989) (see Finlayson et al. 1992, 1994, Finlayson et al. 1997). A limitation of the information is that the period over which such data were collected was a period of high disturbance associated with buffalo activity. Since that time vegetation work has shifted focus to Ranger-specific, terrestrial vegetation (Ashwath et al. 1994, Brennan 1994, Bayliss et al. 2004, Brennan 2005), while aquatic vegetation has only received attention during the last few years as technology has improved aerial capacity to view emergent and floating aquatic vegetation (e.g.

unmanned aerial systems technology). There is a need to differentiate the types of information that can be obtained from new technology and that gained from on-ground research. Integration of information will be most beneficial for understanding the requirements for successful aquatic ecosystem establishment.

A meeting was held between ERA and ***eriss*** to evaluate current progress of this programme and determine a revised focus for intended work. Individual projects for addressing Aquatic Ecosystem Establishment are now being considered within the context of ecological processes. For an ecosystem processes approach for rehabilitation to be successful, the key measures need to be strongly underpinned by targeted research which improves understanding of the linkages between monitoring variables and ecosystem processes.

Within this context, a number of projects have been identified that could begin to address the limited information available for aquatic vegetation:

1. Analysis and evaluation of historical aquatic vegetation data.
2. Re-survey of the sites used in the early vegetation reports.
3. Analysis of water quality influences upon aquatic vegetation seedbanks.
4. Defining aquatic analogue sites.

Dealing with these in turn:

Project 1 – Analysis and evaluation of historical aquatic vegetation data:

Objective: Describe the spatial and temporal patterns of aquatic plant communities in ARR waterbodies.

Although existing ***eriss*** data from early aquatic vegetation surveys have been evaluated and mapped, full analysis and description of seasonal and interannual variations are required. Semi-quantitative aquatic vegetation data have been collected over the years as part of monitoring programmes for macroinvertebrates and fish while quantitative data are available from the biennial ***eriss*** popnetting programme. In addition, data collected and evaluated by ERA may also be examined. Amalgamating these data may provide a dataset suitable for:

(i) assessing temporal and spatial variation in aquatic vegetation,

(ii) determining if community typology and classification are evident, and

(iii) seeking environmental correlates of community structure, including water quality variables.

Project 1 is proceeding in ***eriss***' current workplan for Q2 2014–15. The results of these analyses will inform further proposals.

Project 2 – Re-survey of the sites used in the early vegetation reports:

Objective: Assessment of aquatic vegetation recovery from historical feral animal disturbance to determine shifts in vegetation baselines.

To support an amalgamated dataset, it would be beneficial to re-survey aquatic vegetation at each of the sites cited in the early vegetation reports (reports by Finlayson et al., multiple years). The current macroinvertebrate and fish monitoring programmes sample billabongs annually or bi-annually. However, these are only a subset of the sites surveyed in the early reports. Re-surveying the sites would allow comparisons of vegetation communities between a period of significant disturbance (feral buffalo) and 30 years of post-disturbance recovery. This should provide a general indication of the rate of recovery expected following rehabilitation management activities.

Potential commencement planned for late 2015–16.

Project 3 – Analysis of water quality influences upon aquatic vegetation seedbanks:

Objective: Evaluate aquatic plant biodiversity between on-site and off-site aquatic ecosystems.

This project addresses the question of how variation in water quality affects germination of sediment seed banks in aquatic ecosystems. As the mine moves into the rehabilitation phase, there is a need to understand what vegetation may eventually be found on the rehabilitated site if aquatic ecosystems form naturally or through active management (e.g. re-establishment of creeklines, construction of water treatment waterbodies). Water quality on site will be different than surrounding areas, and highly disturbed sites often exhibit different seedbanks compared to surrounding natural areas. Identifying such effects on the emergence and establishment of aquatic vegetation is essential for applying adaptive management principles within a rehabilitation context. The proposed seedbank study takes an experimental approach and thereby reduces uncertainty in the cause of species absences from surveyed mine-disturbed environments. Such data will complement the visual surveys conducted by ERA and may provide information about the causes of species missing from the current communities.

Potential commencement planned for late 2015–16.

Project 4 – Defining aquatic analogue sites:

Objective: Evaluate and catalogue the characteristics to be applied for measuring aquatic ecosystem recovery.

Analogue sites have been used in developing the rehabilitation plan for the terrestrial landform and its associated revegetation. Such an approach may be useful for aquatic ecosystems. This project is only in the early stages of conceptualisation, but it would aim to integrate and evaluate the natural variability of aquatic ecosystem characteristics in order to define acceptable levels of change in biodiversity and ecosystem health for rehabilitation. Results from Projects 1–3 would inform this project.

Commencement time for this project has yet to be determined.

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Ranger revegetation

C Humphrey, M Gardener[1], P Lu[2] & R Bartolo

This summary paper provides an update on research and monitoring activities associated with Key Knowledge Needs (KKNs) relevant to Ranger revegetation. Relevant KKNs include:

- 2.5 Ecosystem establishment
  - 2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective
  - 2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites
  - 2.5.3 Establishment and sustainability of ecosystems on mine landform
- 2.6 Monitoring
  - 2.6.1 Monitoring of the rehabilitated landform
  - 2.6.2 Off-site monitoring during and following rehabilitation

Background and progress to date

For each of the major rehabilitation activities required for closure at Ranger, stepwise processes for planning and approvals are required (ERA 2014). These are shown in Figure 1 (left hand column). A key component of rehabilitation and closure planning is the development of closure criteria. These measurable and quantifiable benchmarks are the targets against which the long term success and sustainability of rehabilitation will be measured. They will serve as the basis for issuing of a close-out certificate (ERA 2014). For revegetation, the development of closure criteria, together with development of assessment criteria for monitoring and sign-off, will be considered by a Flora and Fauna Technical Working Group (TWG) that will report to the Ranger Closure Criteria Working Group (CCWG). The steps that the TWG must consider in developing closure criteria are shown in Figure 1 (right-hand column).

At ARRTC 31 (November 2013), ERA articulated a trajectory approach for assessing closure criteria for monitoring and sign-off (Paulka & Humphrey 2013). This approach has been further refined (ERA 2014) and is demonstrated in Figure 2. Trajectories are applicable to any measurement endpoint that is expected to be reached after a (typically modelled) period of time from initial establishment. Where short-term achievement is likely, the trajectory provides a management tool to demonstrate to stakeholders that the rehabilitated area is behaving as predicted and is moving through the stabilisation and monitoring phase towards the post-closure phase – see left-hand panel of Figure 2. Milestones along the path may be selected, with deviations from the milestone triggering mitigating actions. Where a measurement endpoint will not be achieved in the short term, there is a need to model the trajectory carefully, then select points on the modelled pathway that would culminate in the 'pseudo' final, agreed criteria – see right-hand panel of Figure 2.

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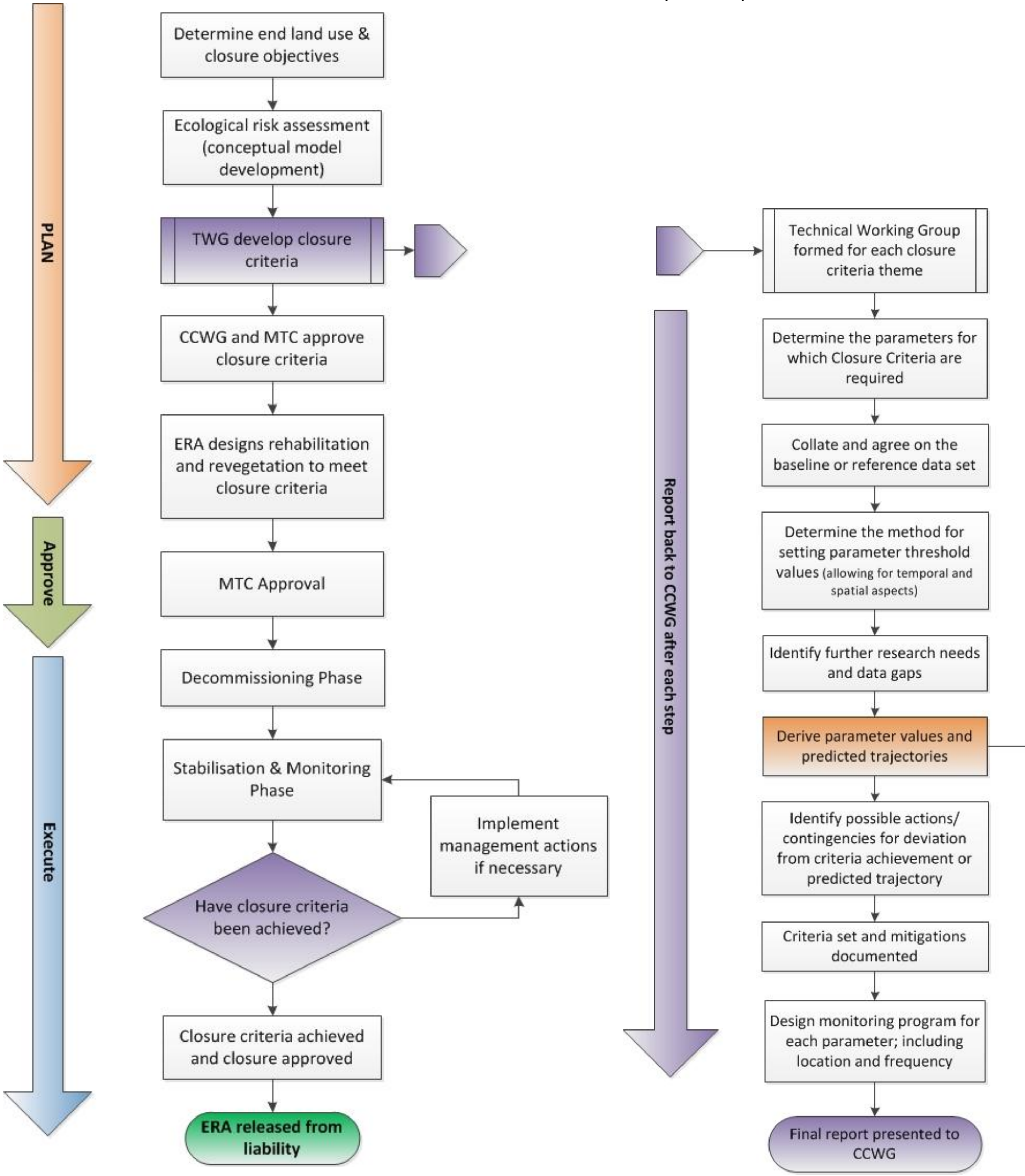

Figure 1. Closure process for Ranger, including development of closure criteria. From ERA (2014).

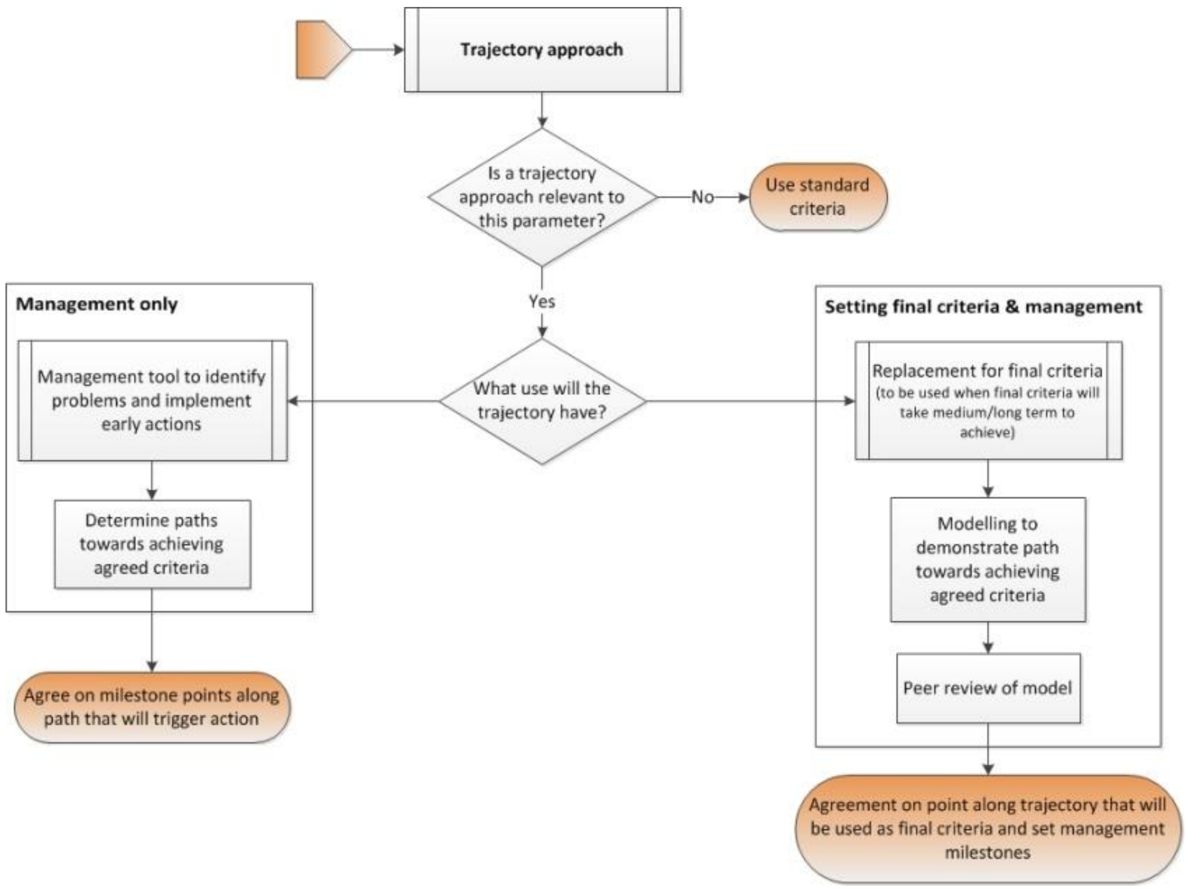

Figure 2. Use of trajectory approach in closure criteria From ERA (2014).

For ARRTC 31, Humphrey & Paulka (2013) re-stated the conceptual and ecological model for revegetation of the restored Ranger minesite that is the basis for trajectory development for this (revegetation) activity. Such models are important to identify in order to understand recovery/restoration trajectories, understand and predict potential thresholds of change towards undesired states, and to determine if and when appropriate management intervention is required and the type of intervention (Hobbs & Suding 2009).

Alternative stable states (or state-and-transition) models of ecosystem dynamics have been recognized as relevant to revegetation programmes in wet-dry tropical savannas, including Ranger (see Bayliss & Gardener 2006). As applied to restoration, this particular model has been informed by Australian rangelands ecology research, depicting shifts between natural and degraded states (e.g. Hobbs & Suding 2009). State and transition models identify processes that cause transition between defined states and in the context of restoration, thresholds may be reached that prevent recovery without significant management intervention. For revegetation efforts in the wet-dry tropics, the interaction between early establishment of high fuel-load grasses and/or acacias and intense fires can lead to a biotic-abiotic positive feedback that prevents establishment of mature eucalypt woodlands (see Groote Eylandt, Nabarlek and Ranger case studies in Bayliss & Gardener (2006).

This 'deviant' grass-acacia state was identified early in revegetation trials at Ranger and preventative steps were developed as part of the Ranger revegetation strategy (Reddell & Meek 2004).

On the basis of (i) the original Ranger revegetation strategy (Reddell & Meek 2004), (ii) ongoing refinements to the strategy, particularly through learnings from the Ranger trial landform (Gellert 2012), and (iii) practical understanding of state and transition responses, a trajectory-based model for Ranger revegetation can be readily conceptualised – see Figure 3.

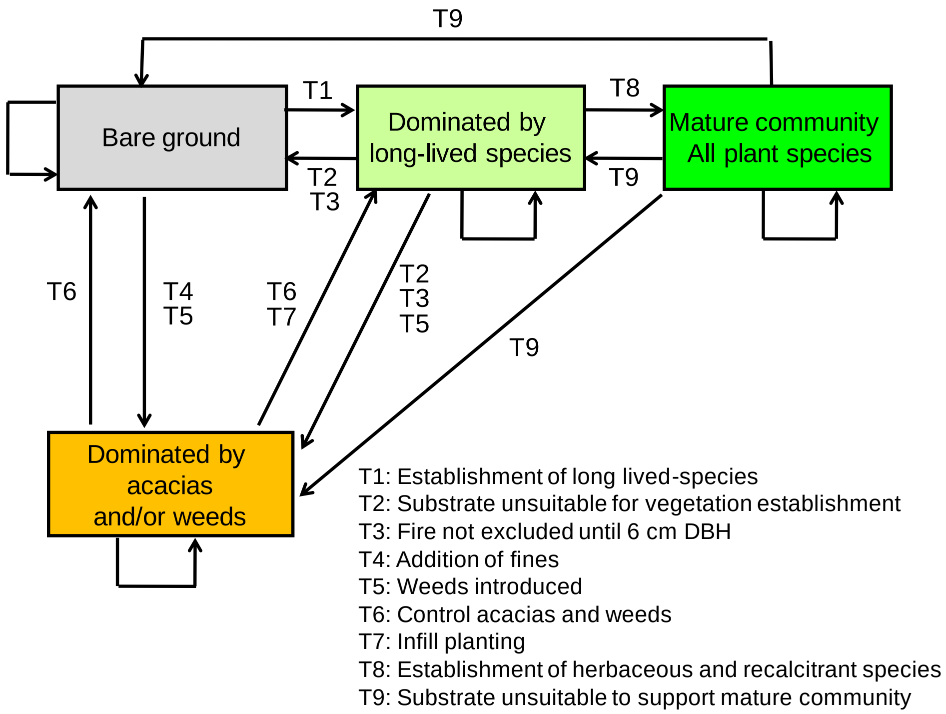

Figure 3. General trajectory for revegetation of the Ranger minesite embedding transitions (T) or drivers to move between states (viz state and transition model for ecosystem dynamics)

The model for successful development of a mature vegetation community for the site is highly dependent upon several assumptions:

1. Landform cover built to specifications (waste rock with no additional fines);
2. Site is revegetated using tubestocks, with supplementary irrigation needed for 3-6 months if planted during the dry season;
3. Sufficient water and nutrients in waste rock cover available for vegetation establishment and reproduction;
4. Fire excluded until plants are 5-years old, then managed thereafter; and
5. Aggressive acacias and weeds controlled.

Elements of the original revegetation strategy (Reddell & Meek 2004), knowledge subsequently acquired to the present (2014) and potential risks and key knowledge gaps that require further work, are provided in Table 1.

Table 1. Elements of the Ranger revegetation strategy with key learnings, risks and identified research needs, to 2104.

Reddell & Meek (2004)	Knowledge (2014)	Potential risks and further work
1. Determine likely physical and chemical characteristics of final landform • Identify constraints to vegetation establishment and long-term success	Trial landform indicating no short-term constraints of landform substrate for plant growth. ERA's water balance study indicates Potential Available Water (PAW) on the trial landform (4 m of waste rock) is lower than analogue sites; whether this is the reason for signs of stress amongst some species on the trial landform (e.g. *E. miniata*) is unknown. Partially compacted subsurface layers or thicker layer of waste rock cover will improve the local PAW.	Water balance studies could suggest risks to long term sustainability of the vegetation (at maturity), especially where thickness of waste rock is less (i.e. thinner) than a particular – and unknown – threshold (this requirement is currently being assessed). Water availability may similarly be limited for vegetation establishment on steeper slopes. **Further studies**: Use Pit 1 capping and re-vegetation to trial performance of vegetation over various waste rock layer designs (thicknesses, partial compacted subsurface layer) and slopes.
2. Identify suitable vegetation types for establishment	Analogue studies have identified key plant community types and species densities	Rehabilitation targets wrt analogues yet to be negotiated in light of (i) plant-environment relationships determined from analogue studies, and (ii) recent learnings from trial landform and plant propagation.
3. Establish vegetation directly into landform substrate	Additional laterite mixed into waste rock (appearing at surface) facilitated weed and other fire-prone and vegetation-smothering establishment on the trial landform. Waste rock <u>only</u> substrate appears to be a suitable medium, for at least early establishing plants (4-5 years).	Waste rock mixed with 'contaminated' laterite introduced weeds and aggressive acacias (*Acacia holosericea*) on the trial landform. Even if laterite materials were weed-propagule free, surface laterite could still facilitate establishment of weeds blown in from nearby infested areas. Laterite mix layer *below* a layer of clean waste rock would negate those risks.
4. Maximise surface roughness and 'patchiness' during site preparation.	Several advantages to the inclusion of riplines have been found. They increase available microhabitats, nutrient cycling and ecosystem patchiness, and trap leaf litter on leeward side. Riplines, also reduce short term erosion, thus retaining fines and nutrients for plant use.	**Further studies**: Nutrient cycling and proxies have not yet been studied on trial landform. **Further consultation**: Landowners have concerns about rough surfaces left after landform formation for traversing country. Large size rocks brought to surface while ripping (creating useful habitat structure) is purported to be less a problem than riplines themselves. Further negotiation is required; for example, landowners may be satisfied that by the time vegetation is established (10-20 years), weathering of rocks and smoothing out of the riplines will have occurred. Further, adequate spacing of riplines (e.g. 10 m apart) would provide greater ease of traversing.
5. Use seed and propagation material collected within 30 km of Ranger for all species.	Limited seed stock available within recommended radius. This has lead to a provenance radius re-assessment with an expanded radius proposed.	**Further consultation**: Expanded provenance radius presented to stakeholders; awaiting endorsement of KNP and GAC
6. Focus on initially establishing diverse range of the long-lived 'framework' species.	Informed by analogue studies and confirmed by trial landform revegetation. With few exceptions (see 10 below), no significant impediments to propagation of most long-lived 'framework' species.	See 10 below.
7. Introduce a range of mycorrhizal fungi from local environments to aid in the establishment of framework species	Currently limited although suites of mycorrhizal fungi are incorporated in the potting mix used for the tubestock	Addition of mycorrhizal fungi not tested on trial landform.

Table 1. Cont.

Reddell & Meek (2004)	**Knowledge (2014)**	**Potential risks and further work**
8. Avoid use of high densities of acacia species	Certain aggressive acacias cause deviant revegetation trajectory. Even local (to the area) acacias may need to be planted at lower densities than they occur in surrounding analogue sites.	Poor knowledge on controlling acacias and their seedbank though on the trial landform, *Acacia holosericea* has not established itself on the waste-rock-only surface (though is present on lateritic surfaces).A screening process is required to identify local acacias that may pose a risk to initial vegetation establishment.
9. Avoid actively re-introducing grasses and vigorous herbs during the first year	All Ranger revegetation and landform trials demonstrate that sites with weeds (as a proxy for native grasses and herbs) impede the establishment of framework species. Delay introduction of ground cover for at least 5 years after establishment of the framework species. This avoids potential fire damage from high fuel-load grasses and reduces competition.	No initial ground cover increases erosion risks. Erosion risks can be mitigated by use of strategic riplines.
10. Use nursery-grown planting stock to establish framework species.	Direct seeding on trial landform had very limited success though kapok, and some grevillea and acacia species could be directly seeded into waste rock. If tubestock planted during the dry season, 3-4 months of irrigation appears to be adequate for establishment. Volunteers / self-recruits (>10 species to date) and recalcitrants (e.g. emu apples [*Owenia*]) informing nursery efforts, while an understanding of specific planting requirements of some species is developing (e.g. *E. miniata* should not be planted at the bottom of rip lines; best establishment/survival of *Xanthostermon paradoxus* is in patrial shade).	Inability of propagation some recalcitrant species has implications regarding meeting the ERs (i.e. species composition /densities matching natural surrounding environment). See 2 above. Notwithstanding, vast majority of framework species can be successfully propagated.
11. Apply fertilisers in a strategic manner using formulations and delivery methods that maximise their effectiveness.	For trial landform, osmocote slow release fertiliser (low P) for natives was hand-broadcasted to side of plants during two wet seasons subsequent to planting.	**Further assessment**: potential eutrophication risks to receiving water environments (including native 'pests' in establishment sumps/wetlands, e.g. *Typha*). Plant and soil nutrients should be monitored and adequacy and 'downstream' risks assessed
12. Rigorous control of potentially threatening weed species.	Weed species can be controlled but if programme is halted, weeds can quickly recover. Weeds much less of a concern for waste-rock-only substrates.	Adhere to, regularly review, ERA's Five-year weed management plan and strategy. Essential to develop completion criteria embedding this strategy. Includes management of native, but non-endemic and aggressive, *A. holosericea*, around minesite. Control of exotics should based on risk, particularly that of fire and competiveness to tubestocks.
13. Exclude fire from revegetation areas during the first three to five years after establishment	Framework species must achieve a minimum diameter of 6 cm to withstand fire.	On the waste rock substrates of the trial landform, there have been insufficient grasses to carry a fire in the 6 years of its establishment. Hence risks from fire damage are low.
14. Design and implement a rigorous and scientifically-based monitoring strategy for on-going evaluation of the performance of the revegetation	Will be developed by Flora and Fauna subgroup of Ranger Closure Criteria Working Group	Further work includes the monitoring of Jabiluka revegetation using an Unmanned Aerial System (UAS). The methodology developed will be applied to subsequent revegetation efforts at Ranger.

Conclusions and further work

Revegetation knowledge to guide planning for rehabilitation of the Ranger minesite is well advanced. The knowledge and advice, together with development of assessment criteria for monitoring and sign-off, will be taken to the Flora and Fauna Technical Working Group (TWG), that will report to the Ranger Closure Criteria Working Group.

Key knowledge acquired to date includes:

1. Detailed analogue characterisation to inform species composition and abundances (see review in Humphrey et al. (2014))
2. Nursery requirements for native trees and shrubs,
3. Practical learning from the Ranger trial landform, and
4. A prescribed revegetation strategy that, if implemented correctly, mitigates against undesired ecological states.

Hereafter, the Flora and Fauna TWG will build on this knowledge and for each of the 14 key elements of the revegetation strategy, where applicable, will derive the following to effectively monitor and assess revegetation success:

- Quantifiable closure criteria
- Pragmatic monitoring methodology, including UAS technology
- Performance standards
- Guidelines for acceptance, and
- Corrective actions.

The most recent revegetation programme by ERA has been directed at Jabiluka, while rehabilitation of Magela and Djalkmara Land Application Areas (LAAs) will progress within several years. Cumulative knowledge and learnings will inform the LAA revegetation programme though the challenges faced for disturbed natural woodlands of the LAA differ from those of the waste-rock-formed final landform. Areas within the LAA have high densities of native, but non-local and aggressive, *Acacia holosericea*, as well as speargrass, that pose a risk to successful revegetation (see strategy elements 8, 9 and 12 from Table 1). It will be important to manage *A. holosericea* and speargrass in these areas as part of site preparation for rehabilitation efforts, to ensure revegetation success.

Finally, for revegetation of the final landform, water balance studies have indicated potential risks to long term sustainability of the vegetation (at maturity) where thickness of waste rock is less (i.e. thinner) than a particular – and unknown – threshold (see strategy element 1 from Table 1). Water availability may similarly be limited for vegetation establishment on steeper slopes. Further studies are required to assess these risks. Pit 1 capping and re-vegetation provide the opportunity to trial performance of vegetation over various waste rock thicknesses, slopes and partially compacted subsurface layers.

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Post-rehabilitation radiation exposure due to Aboriginal bush foods from terrestrial ecosystems

Che Doering & Andreas Bollhöfer

Introduction

The current authorisation for the Ranger mine requires that all mining and processing activities must cease by 2021 and that site rehabilitation must be completed by 2026. Rehabilitation of the mine will result in the re-establishment of terrestrial ecosystem and eventual return of the land to traditional owners. Traditional owners using the rehabilitated Ranger mine site and adjacent areas of Kakadu National Park for cultural activities may incur a radiation dose from the site. This dose must not exceed the public dose limit of 1 mSv in a year above the pre-mining background (ARPANSA 2005, Commonwealth of Australia 1999).

A radiological dose model is currently being developed to estimate above background doses to Aboriginal people from the rehabilitated Ranger mine site. The ingestion of radionuclides accumulated in bush foods is to be included as an exposure pathway within the dose model to account for traditional living scenarios. A common approach used in dose models to estimate radionuclide accumulation in food items is to use concentration ratios (CRs). These can be expressed as the ratio of the radionuclide activity concentration in the food item (plant or animal tissue) to that in the relevant environmental medium (soil for terrestrial ecosystems) (IAEA 2010, Martin et al. 1998).

eriss has substantial data on radionuclide activity concentrations in terrestrial wild plants and animals and in soils collected from the ARR. This data has amassed from radioactivity measurements of field samples collected over several decades. The data have been collated and CRs calculated for plant- and animal-based bush foods from terrestrial ecosystems. The CRs may be used in combination with other parameters to estimate above background ingestion doses due to the rehabilitated Ranger mine site.

Data and analysis

Radionuclide data were collated from ***eriss*** publications (Martin et al. 1995, Medley et al. 2013, Ryan et al. 2005, Wasson 1992) and a large set of unpublished analysis results. Data of ERA (RUM 1987) and others (White & Gigliotti 1985, White & McLeod 1985, White et al. 1985) were also included to produce a comprehensive dataset from which CRs could be derived. Data collation focused on retrieval of individual sample results rather than summary statistics. In total, data for 164 terrestrial plant samples, 165 terrestrial animal samples and more than 1500 soil samples were collated.

The collated data were organised around a number of biota groups considered representative of common bush food types. These biota groups included bandicoot, buffalo, goanna, pig, wallaby, fruit and yam. The BRUCE tool (Doering 2013) was then used to calculate CRs from the data on a wet mass biota tissue to dry mass soil basis. For a small number of biota-tissue-radionuclide combinations the only tissue activity concentration data available were less than detection limit values. A weighting factor of 0.5 was applied to the tissue activity concentration to calculate CRs for these combinations.

Results and discussion

CR results

CR results for more than 100 biota-tissue-radionuclide combinations were determined. Figure 1 shows the CR geometric mean and standard deviation for each combination. The sample count was low for most combinations and precluded rigorous statistical treatment of the results. Nevertheless, the following general trends were observed:

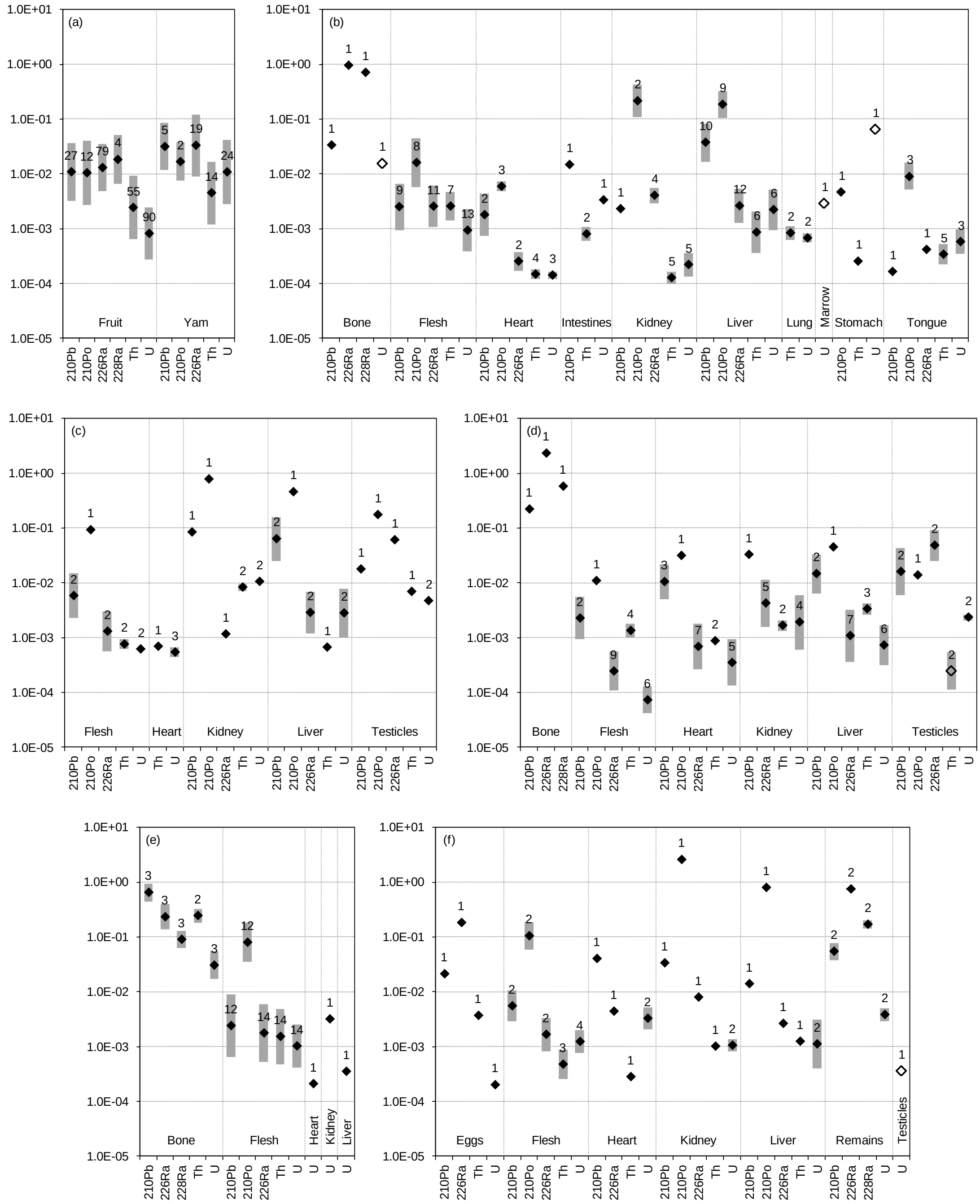

Figure 1 CR geometric means and standard deviations (Bq kg^{-1} wet mass biota tissue per Bq kg^{-1} dry mass soil) for (a) fruit and yam; (b) buffalo; (c) bandicoot; (d) wallaby; (e) pig; and (f) goanna. The number above each diamond indicates the number of results on which values are based. Values plotted with an open diamond indicate that a less than weighting factor of 0.5 was applied to the tissue radionuclide activity concentration to calculate the geometric mean.

1. *CRs for yam higher than fruit.* Yams are root vegetables and uptake radionuclides directly from the soil. Fruits on the other hand sequester radionuclides translocated from the root of the plant following uptake from the soil. Hence the transfer pathway to fruits is effectively longer than for yams and so there is greater possibility of immobilisation of the radionuclide along this pathway by physical and chemical processes.
2. *CRs for ^{210}Po in animal tissue higher, and for U and Th lower, than other radionuclides.* The higher CRs for ^{210}Po may be from elevated ^{210}Po concentrations in the above-ground parts of plants and grasses consumed by foraging animals due to atmospheric deposition and foliar adsorption of this radionuclide as suggested by Johansen and Twining (2010). ^{210}Po is present in the atmosphere from the decay of ^{222}Rn, a natural radioactive gas released to the atmosphere by soils and rocks. The lower CRs for U and Th were potentially due to low gastrointestinal fractional absorption by animals. Gastrointestinal fractional absorption follows the order Th < U < Ra < Pb < Po (IAEA 2010). This may also contribute to the higher CRs for ^{210}Po.
3. *CRs for ^{210}Pb and ^{210}Po in liver and kidney higher than flesh.* Comparative results for bandicoot, buffalo, goanna and wallaby showed that CRs for ^{210}Pb and ^{210}Po in liver and kidney were higher than flesh by approximately one order of magnitude. The liver and kidneys act as filters for substances passing through the body of an animal and are known to accumulate various metals and pollutants, including radionuclides. Seiler and Wiemels (2012) and references therein noted that ^{210}Po activity concentration in kidney and liver of terrestrial mammals is generally one to two orders of magnitude greater than in flesh, and similar findings were reported by Johansen and Twining (2010) for kangaroo and sheep from semi-arid Australian environments.
4. *CRs for ^{226}Ra in bone much higher than flesh.* The bone ^{226}Ra CR was greater than the flesh ^{226}Ra CR in buffalo and pig by two orders of magnitude and in wallaby by four orders of magnitude. A recent review of the environmental behaviour of radium (IAEA 2014) and references therein show that the accumulation of ^{226}Ra in bone is typically orders of magnitude higher than in soft tissues of terrestrial animals. The similar chemical properties of group 2 elements means that radium essentially follows the same metabolism as calcium, and so is preferentially incorporated in bone and other calcified tissue. This may also explain the higher observed ^{226}Ra CR for goanna egg, which was two orders of magnitude higher than that for goanna flesh.

Dose assessment context

Terrestrial plants and animals may accumulate radionuclides above background levels if they are exposed to the rehabilitated Ranger mine site. This applies to plants growing on the site and animals having a home range that overlaps in whole or part with the site. Aboriginal people may receive an above background ingestion dose if they consume the tissues of these exposed plants or animals as part of a traditional diet. The goal in calculating CRs is to estimate the annual above background ingestion dose that an Aboriginal person could potentially receive from the accumulation of radionuclides in terrestrial bush foods due to the rehabilitated site.

This dose can be estimated for each bush food as:

$$Dose\ (mSv) = S \times \sum (CR \times DC) \times M \times F_E \times F_H$$

where:

- S is the above background soil activity concentration (Bq kg^{-1}) of ^{238}U in secular equilibrium with its progeny on the surface of the rehabilitated Ranger mine site;
- $\Sigma(CR \times DC)$ is the sum of the products of the CR and the corresponding radionuclide ingestion dose coefficient (DC, mSv Bq^{-1}) for ^{238}U, ^{234}U, ^{230}Th, ^{226}Ra, ^{210}Pb and ^{210}Po;
- M is the wet mass (kg) of the bush food consumed in a year;
- F_E is the average fraction that an animal is exposed to the rehabilitated site; and
- F_H is the average fraction spent hunting in the zone from within which animals may be exposed to the rehabilitated site.

The parameters S, $\Sigma(CR \times DC)$ and F_E determine the above background radionuclide activity concentrations in plants and animals exposed to the rehabilitated Ranger mine site. Values for these parameters are now relatively well known – above background soil activity concentrations of ^{238}U series radionuclides on the surface of the rehabilitated site can be reasonably estimated from the preferred rehabilitation strategy and work done by Bollhöfer et al. (2014), CRs for terrestrial bush foods have been calculated (Figure 1), ingestion dose coefficients have been determined by the International Commission on Radiological Protection (ICRP 1996), the average fraction that an animal is exposed to the rehabilitated site can be estimated from the size (radius) of the site and animal home ranges. The parameters M and F_H determine the fraction of contaminated diet. Their values are less well known because they depend on post-rehabilitation land use for hunting and gathering and future dietary habits. Agreed land use scenarios and reference diet for rehabilitation planning of the Ranger mine do not yet exist but are being developed through a cultural closure criteria working group that consults with Aboriginal advocacy groups on cultural issues important to rehabilitation.

Table 1 gives an example ingestion dose calculation for the consumption of selected bush foods by an Aboriginal adult for an assumed traditional living scenario and set of circumstances for the rehabilitated Ranger mine site. This example calculation has been provided to illustrate the logistics and mechanics of estimating the above background dose from terrestrial bush foods; parameter values and results shown in Table 1 are indicative only and do not necessarily represent final dose estimates. The assumptions made in the example calculation were:

1. *Rehabilitated site general characteristics*. The area of the rehabilitated Ranger mine site was assumed to be 8 km^2. This is the approximate size of the mine disturbed area requiring rehabilitation. The surface of the rehabilitated site was assumed to be covered with mine waste rock and shaped as near as practicable to resemble the pre-mining landscape. This is currently the preferred rehabilitation strategy for mine disturbed areas (ERA 2013). The rehabilitated site was assumed to be revegetated using local native plant species to broadly match adjacent areas of Kakadu National Park as required by the Environmental Requirements for the Ranger mine (Commonwealth of Australia 1999). The site was assumed to be used as habitat by terrestrial animals similar to other parts of Kakadu National Park and to equally support the growth of plant species included in traditional diet.

2. *Rehabilitated site soil radionuclide activity concentrations.* Ranger waste rock has a maximum ore grade equivalent to approximately 2100 Bq kg^{-1} ^{238}U (Lawrence et al. 2009) and lower average ore grade. Average ^{238}U activity concentration in waste rock used to cover the rehabilitated site was assumed to be half this maximum ore grade and ^{238}U series radionuclides in the waste rock cover were assumed to be in secular equilibrium. The average pre-mining soil ^{238}U activity concentration of the site has been determined from Bollhöfer et al. (2014) to be approximately 360 Bq kg^{-1}. This implies an average above background activity concentration of ^{238}U in secular equilibrium with its progeny of 690 Bq kg^{-1} on the surface of the rehabilitated site.
3. *Diet.* In lieu of an agreed reference diet for rehabilitation of the Ranger mine site, a diet previously developed by ***eriss*** from consultation with Aboriginal people and general observations of traditional dietary habits was used (Ryan et al. 2008). Bandicoot was not included in the ***eriss*** diet, and so it was assumed that the annual consumption of bandicoot flesh was one quarter that of wallaby flesh.
4. *Post-rehabilitation land use for hunting and gathering.* Aboriginal people leading a traditional lifestyle in the vicinity of the rehabilitated Ranger mine site were assumed to hunt over an area of approximately 300 km^2. This is half the nomadic range observed by Altman (1984) for a group of Aboriginal people leading a traditional lifestyle near Momega outstation, approximately 85 km east of the Ranger mine. Anecdotal evidence suggests the nomadic range in the vicinity of the rehabilitated Ranger mine site will probably be less than for the area around Momega outstation due to greater abundance of bush food resources in the Magela Creek corridor.
5. *Animal exposed and hunting fractions.* The animal exposed and hunting fractions were calculated by assuming the rehabilitated site is a circle of radius 1.6 km, the home range of an animal is a circle of radius typical of the particular species and the hunting range is a circle of radius 10 km. The animal exposed fraction represents the portion of the animal home range overlapping with the rehabilitated site and the hunting fraction represents the portion of the hunting range overlapping with the zone from within which an animal may be exposed to the rehabilitated site.

From these assumptions, the annual above background dose from animal-based terrestrial bush foods would be of the order of 0.1 mSv and come largely from buffalo flesh and organs. The annual above background dose from plant-based terrestrial bush foods would be of the order of 0.01 mSv and come almost exclusively from yams. However, buffalo have been significantly eradicated from Kakadu National Park in recent decades, making them potentially less available as a bush food resource in the Magela Creek corridor. The implication is that the dietary intake of wild buffalo sourced from around the rehabilitated Ranger mine site may be less than assumed, and so the dose from buffalo may be lower than estimated. Similar consideration of the 'real-world' availability of other terrestrial bush foods in the Magela Creek corridor should be applied when interpreting assessment results in the context of dose limits.

Table 1 Example ingestion dose calculation: estimated annual average above background dose to an aboriginal adult.

Biota	Tissue	S (Bq kg^{-1})	Σ(CR×DC) (mSv Bq-1)	M (kg)	F_E	F_H	Dose (mSv)
Bandicoot	Flesh	690	1.2×10^{-4}	5	6.9×10^{-1}	2.5×10^{-2}	6.9×10^{-3}
Buffalo	Flesh	690	2.2×10^{-5}	146	7.4×10^{-2}	1.1×10^{-1}	1.8×10^{-2}
Buffalo	Kidney	690	2.6×10^{-4}	18	7.4×10^{-2}	1.1×10^{-1}	2.6×10^{-2}
Buffalo	Liver	690	2.5×10^{-4}	18	7.4×10^{-2}	1.1×10^{-1}	2.5×10^{-2}
Goanna	Flesh	690	1.3×10^{-4}	2	6.9×10^{-1}	2.5×10^{-2}	3.1×10^{-3}
Pig	Flesh	690	9.8×10^{-5}	25	1.0×10^{-1}	8.7×10^{-2}	1.5×10^{-2}
Wallaby	Flesh	690	1.5×10^{-5}	20	6.9×10^{-1}	2.5×10^{-2}	3.6×10^{-3}
Fruit	Flesh	690	2.4×10^{-5}	3	1.0×10^{0}	1.9×10^{-2}	9.3×10^{-4}
Yam	Tuber	690	5.3×10^{-5}	20	1.0×10^{0}	1.9×10^{-2}	1.4×10^{-2}

Conclusions

Terrestrial wild plants and animals are an important part of traditional Aboriginal diet in the ARR. They are also a potential vector for above background public radiation exposure following rehabilitation of the Ranger mine. CRs have been determined for radionuclides significant to uranium mining for terrestrial wild plants and animals important to traditional Aboriginal diet of the ARR. These CRs have been used in combination with other parameters in an example ingestion dose calculation to illustrate the dose assessment process for the terrestrial bush food exposure pathway. Lack of agreed land use scenarios and reference diet currently impedes the final estimation of above background dose due to bush foods sourced from terrestrial ecosystems.

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A catalogue of traditional plant foods in Kakadu National Park

G Fox

Introduction

Early studies on bush food types, their culinary preparation and consumption by Aboriginal people were undertaken in Arnhem Land, and include information gathered during an American-Australian Scientific Expedition in 1948 (Specht 1958, McArthur 1960), a study of the use of plants by Aboriginal people in the Oenpelli region (Smyth & von Sturmer 1981) and a bush food study conducted over 12 months at Momega outstation in the Maningrida area (Altman 1984). Some information relevant to the bush foods of the local Aboriginal people of Kakadu National Park (KNP) is also available in two unpublished reports . These reports include descriptions of a broad range of plants and their different uses by the Gundjeihmi people, including food, medicine, tools and implements, and cultural and spiritual purposes. Thus, Chaloupka & Giuliani (1984) identified 111 species of plants as food resources for the local Aboriginal community, while Lucas & Russell-Smith (1993) identified 127 species. A combined total 147 edible plant species were identified from both reports, together with other valuable information about usage, collection and preparation.

In order to capture all current knowledge pertaining to the traditional plant food usage of the local Aboriginal people of Kakadu National Park, and to collate this knowledge in a single report, a project to catalogue the 147 identified plant foods was initiated. The gathered information has been used for selection of representative organisms required for a radiation dose assessment of the environment (Doering et al. this report) and will be included as metadata in the BRUCE tool, through links to the description sheets for individual plant species.

The catalogue will be an easy-to-use resource for field identification of specimens required for analysis and, as a further aid, a photo database will also be created of the complete 147 identified plant foods. The catalogue will be used as a tool to effectively communicate with local aboriginal people about those foods which have been collected and analysed, and those viewed by Aboriginal people still needed for analysis.

Methodology

The descriptions of each plant species are treated in two parts:

Aboriginal knowledge: Information was mostly taken from Chaloupka & Giuliani (1984) and Lucas & Russell-Smith (1993). Where information was lacking or scarce, additional information was obtained from other reports (Blake et al. 1998 McArthur 1960, Specht 1958, Wiynjorrotj et.al. 2005). It should be noted that knowledge may vary slightly between different Aboriginal groups, so information was restricted to just those Aboriginal groups adjacent to KNP, i.e. Iwaidja from Cobourg Peninsula, Kunwinjku from Arnhem Land and Jawoyn from southern areas of KNP and Katherine. Aboriginal names are in the Gundjeihmi language, the most commonly-used language in the KNP area. Photos of the plant parts eaten, food importance, food preparation, season that the plant is eaten, together with any pertinent information about collection of the plants, are catalogued.

Botanical knowledge: Information was taken from a variety of (referenced) sources and is presented in plain language to enable the non-botanist to identify each plant species with reasonable accuracy. Information presented includes; habit, bark, leaves, flowers, fruit, flowering and fruiting phenology and habitat.

Results

At this stage, 127 of the 147 edible plant species have been photographed and catalogued, with preliminary description sheets for most of these species prepared. It is planned to continue the photography and description of the remaining species in the coming months; most of these species typically produce edible fruits or vegetative parts late into the 'build up' and wet season. Examples of how the information may ultimately be presented are provided in the following pages.

Cartonema parviflorum

anmarridjak

HERB

Tubers form at base of stem

Small tubers to 1.2 cm diameter

YAM: **Important Food.** Gudjeuk - Banggerreng (January – April).

Small underground corms are roasted briefly on hot coals. Fresh corms have a strong spicy smell and many are necessary to provide a good meal. Bags of them are collected to be consumed during ceremonies.

Habitat: Lowland Woodland, Sandstone Woodland.

Sources of Information. Chaloupka & Giuliani 1984, Lucas & Russell-Smith 1993.

Cartonema parviflorum

Family: Commelinaceae
Habit: slender herbaceous perennial to 25 cm tall.
Tuber: root tuber obovate to globoid 1-1.2 cm diameter, buried 3-5 cm; glabrous, surrounded by several loose, brown, papery sheaths. Tubers replaced annually.
Leaves: grass-like, linear, 8-15 cm long, hairy; each tapering to a fine point.
Flowers: white to cream with three petalled, loosely placed along stem. Sepals persist after the petals fall, giving the plant a spiky appearance.
Habitat: open woodland in damper areas.
Flowering: December – August.
Sources of Information. Levitt 1981, Pate and Dixon 1982, Specht 1958.

Multiple stemmed herb to 25 cm high

Three petalled cream coloured flowers

Nymphaea macrosperma
maardjakalang

AQUATIC HERB

Yams ready for cooking

Seeds are larger than the two other species of *Nymphaea* eaten

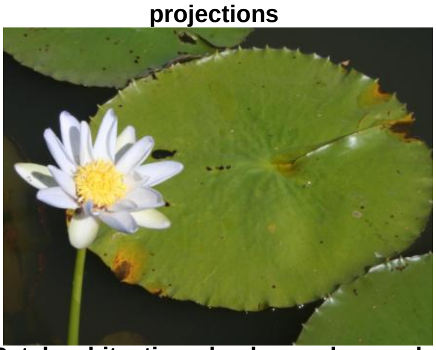
Stems peeled and eaten like celery

YAM: Staple Food. Banggereng – Gunumeleng (April – December).
Yams are dug out of the mud, peeled, then cooked gently on hot coals for 10-15 minutes and eaten.
SEEDS & SEED HEADS: Staple Food. Banggereng – Gurrung (April – September).
Seed head can be eaten raw, lightly roasted on the coals or ground and prepared into a very nutritious damper which is wrapped in waterlily leaves then in paperbark and cooked on the coals.
STEMS: Eaten Food. Banggereng – Gurrung (April – September).
Flower/fruit stems eaten raw or lightly cooked on coals after peeling.

Habitat: Floodplain.
Sources of Information. Chaloupka & Giuliani 1984, Lucas & Russell-Smith 1993.

Nymphaea macrosperma
Can be identified from the two other lily species by the margins of the leaves mostly being toothed or serrated and having spine like projections on the tip of the teeth (see photo).
Family: Nymphaeaceae
Habit: perennial floating aquatic herb.
Leaves: smooth, glossy green above, ovate-elliptic to orbicular, 17-38 cm long x 15-31 cm wide, distinctly toothed on leaf margins to 3 mm long, with spine like projections, floating on water surface.
Flowers: large, petals 10-18, white, tinged pale purple or pale blue to dark purple, numerous yellow stamens.
Fruit: spongy, roundish, capsule about 25-35 mm across, maturing underwater, breaking up irregularly when ripe, containing many green, ellipsoidal seeds to 3.5 mm long with numerous longitudinal rows of short hairs.
Habitat: usually fringing permanent billabongs and semi-permanent back-water swamps on floodplains, grows in water to 2.5 metres deep.
Flowering: March – December.
Fruiting: July – October.
Sources of Information. Cowie et al. 2000, Flora of Australia Volume 2 2007.

Underside of leaf. Distinctly toothed leaf margins with spine like projections



Petals white, tinged pale purple or pale blue to dark purple

Pandanus spiralis
anyagngarra

TREE


Kernels embedded within the woody fruits


Pale bases from newest leaves cut from stem and eaten


Fresh fruits chewed and sucked for juice. Fruit crushed, soaked in water and the liquid drunk

SEEDS: **Eaten Food.** Gurrung (August – September).
Kernel extracted from fruits and eaten raw or roasted.

LEAF BASE: **Eaten Food.** Throughout the year
White, moist bases of fronds can be eaten.

FRUIT: **Eaten Food.** Gurrung (August – September).
Fresh fruits chewed and sucked for the juice. The skin of the fruit is also crushed, soaked in water and the liquid drunk.

Habitat: Lowland Woodland, Sandstone Woodland.
Sources of Information. Chaloupka & Giuliani 1984, Lucas & Russell-Smith 1993.


Fruit stalk (peduncle) can also be eaten after being lightly roasted

Pandanus spiralis (Pandanus)

Family: Pandanaceae
Habit: distinctive tree to 8 metres high with prickly leaves and spirally marked trunk, does not have prop roots, often in small clumps, male and female plants.
Leaves: crowded spirally towards end of branches, stiff, thick, leathery, V-shaped, widest at base tapering to elongated pointed tip, 1.5-2 m long x 4-7 cm wide at base, bent downwards about third of length, numerous longitudinal veins, numerous small thorns along margins and back of mid-rib diminishing in size towards the apex.
Flowers: male and female flowers on separate plants, small, white, inconspicuous, in dense terminal spikes, 3-7 cm long, enclosed by leaf-like bracts.
Fruit: large, roundish composite clusters pendulous below the crown, 15-20 cm across, containing 8-25 individual wedge-shaped, woody fruits 50-70 mm x 50-80 mm, each with 7-10 edible seeds, deep orange-red when ripe but seeds eaten when fruit turns a brown colour.
Habitat: swamps, billabongs and fringes of floodplains, beside freshwater streams, open forest and woodland.
Flowering: April - July.
Fruiting: June – October.

Sources of Information. Brock 2007, Cowie et al. 2000, Petheram & Kok 2003.


Distinctive tree

Capparis umbonata

gaiwom / gaibam / baiwal

SHRUB or TREE

FRUIT: **Eaten Food**. Gudjeuk (January – March).
Fruit is eaten when ripe, green and soft.
Habitat: Lowland Woodland.
Source of Information. Lucas & Russell-Smith 1993.

Fruit born on very long stalk

Orange flesh

Capparis umbonata (Wild Orange)

Family: Cappparaceae
Habit: slender tree to 7 metres high with pendulous leaves and branches, small spines along young branches.
Bark: grey, hard and rough.
Leaves: alternate, variable in shape and size, pendulous, smooth, leathery, long and narrow or curved and elongated tapering towards base, 100-230 mm long x 7-30 mm wide, dull dark green, prominent mid-rib, round or pointed tip, petioles 10-30 mm long.
Flowers: large, white to pale yellow, sweetly scented and open at night, with numerous long stamens and long protruding style, 50-100 mm long, up to 6 flowered in inflorescences towards ends of branches. Flower stalk 20-55 mm long.
Fruit: smooth, globular, 30-50 mm diameter, on long stalk, green or green-yellow when ripe, usually containing more than ten brown seeds to 8 mm long.
Habitat: scattered in open forest and woodland, sandstone country in shrubby woodland on the plateau or slopes. Occasionally in monsoon vine thickets.
Flowering: Most months.
Fruiting: Most months.
Source of Information. Brock 2007, Flora of Australia Volume 8, 1982, Petheram & Kok 2003, Short 2011.

Flowers with long white stamens

Rough bark

Narrow leaves

Outstanding work

Writing of the report is continuing. Some botanical information for, and photography of, some of the species is still incomplete, but description of most of the species has been completed. The existing plant descriptions need to be internally validated before final publication.

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Environmental media concentration limits for terrestrial wildlife

C Doering & A Bollhöfer

Introduction

The environmental exposure of wildlife to above-background radiation sources resulting from human activities has been included as a distinct exposure category in contemporary radiation protection recommendations from national (ARPANSA 2014) and international (ICRP 2007) bodies and an environmental radiological protection framework developed (ICRP 2008, ICRP 2009, ICRP 2014). Radiation exposure to wildlife from radionuclide contaminated environments should be considered and assessed where the potential for environmental effects exist or where otherwise required to do so. Uranium mine rehabilitation is a human activity that may lead to the environmental exposure of wildlife if average radionuclide concentrations on the rehabilitated landform are greater than pre-mining. This is the likely scenario for Ranger uranium mine, where the preferred rehabilitation strategy is to use waste rock material as the surface cover on the rehabilitated landform.

Environmental exposure is typically quantified as the absorbed dose rate to wildlife from the radionuclide contaminated environment, and can be placed in a risk context by comparing with a screening dose rate. The screening dose rate is an absorbed dose rate value that provides some level of protection to wildlife. The commonly used ERICA Tool screening dose rate is 10 μGy h^{-1} and is intended to conservatively provide 95% species protection based on species sensitivity distribution analysis (Garnier-Laplace et al. 2008). Exposures below this screening dose rate suggest low risk. Those above it are not necessarily harmful or unacceptable, but highlight that a more detailed assessment of the exposure situation may be necessary in order to determine this.

The environmental media concentration limit (EMCL) for terrestrial ecosystems is the soil radionuclide activity concentration that would result in an absorbed dose rate to wildlife equal to that of the screening dose rate and can be calculated as (Brown et al. 2008):

$$EMCL\ (Bq\ kg^{-1}) = \frac{SDR}{F}$$

where:

SDR (μGy h^{-1}) is the screening dose rate; and

F (μGy h^{-1} per Bq kg^{-1}) is the dose rate to the organism per unit activity concentration of the radionuclide or mix of radionuclides in the soil of the contaminated environment, and is calculated from organism radionuclide concentration ratios (CRs) and dose coefficients.

Comparison of measured or predicted above background soil radionuclide activity concentrations in the contaminated environment with the EMCL can be used to establish the risk context. The convenience of the EMCL in uranium mine rehabilitation is to provide guideline values for soil radiological quality for wildlife-based protection that can be considered at the landform design stage.

Method

Radionuclides

The Ranger rehabilitated landform was assumed to be covered with a waste rock substrate. ^{238}U series radionuclides were assumed to be in approximate secular equilibrium in the substrate as waste rock is a natural (unprocessed) material. The ERICA Tool (Brown et al. 2008) was the used for the analysis. The tool includes progeny radionuclides in the dose coefficient of their parent if their half-life is less than 10 days. The specific radionuclides required for ^{238}U series secular equilibrium were ^{238}U, ^{234}Th, ^{234}U, ^{230}Th, ^{226}Ra, ^{210}Pb and ^{210}Po.

Representative organisms

Local wildlife species spanning a range of different organism types were selected for environmental exposure assessment and calculating EMCLs for the Ranger rehabilitated landform (Table 1). Their selection considered exposure likelihood, ecological importance, Aboriginal cultural factors and threatened species. The selected local wildlife species are considered preliminary at this stage and are pending endorsement from the cultural closure criteria technical working group.

Reference organisms

Reference organism dosimetric models for each selected local wildlife species were defined in the ERICA Tool based on typical body shape proportions and mass. Some of the species were matched to existing reference organisms in the tool, while for others a new reference organism was created (Table 1). Reference organism geometries were simple ellipsoids, since more complex configurations are not supported by the ERICA Tool. Internal and external dose coefficients for the reference organisms were generated in the ERICA Tool's dosimetric module (Ulanovsky et al. 2008).

Table 1 Body shape proportions and mass used for the reference organisms in the ERICA Tool (A is the length of the ellipsoid major axis, B and C are the lengths of the ellipsoid minor axes).

Organism		Body shape proportions and mass			
Representative	Reference	A (cm)	B (cm)	C (cm)	Mass (kg)
Green tree frog	ERICA – Amphibian	8	3	2.5	3.14×10^{-2}
Corella	ERICA – Bird	30	10	8	1.26
Spear grass	ERICA – Grasses & Herbs	5	1	1	2.62×10^{-3}
Green ant	New – Ant	3.2	0.32	0.32	1.74×10^{-4}
Sand goanna	New – Goanna	100	5	5	1.3
Olive python	New – Snake	250	5	5	3.3
Cathedral termite	New – Termite	3.2	0.32	0.32	1.74×10^{-4}
Agile wallaby	New – Wallaby	70	20	20	15
Ironwood	ERICA – Tree	1000	30	30	471

Occupancy factors

Occupancy factors were assigned to each organism for time spent on the rehabilitated landform and in different ecosystem compartments. Landform occupancy was assumed to be 100% for all organisms in order to maximise their environmental exposure and ensure a conservative EMCL. This occupancy assumption is not unrealistic as most of the selected local wildlife species have a home range that is much smaller than the area of the planned rehabilitated landform. The one exception is the corella which is more mobile and has a larger home range. Ecosystem compartment occupancy factors were also selected to maximise environmental exposure. All organisms were assumed to spend 100% of their time on soil except for the cathedral termite which was assumed to spend 100% of its time in soil.

Concentration ratios

Organism-to-soil CRs (Table 2) to back-calculate the component of the EMCL related to internal exposure were obtained by one of three methods:

- *Most preferred:* obtain tissue-to-soil CRs from the BRUCE Tool (Doering 2013) and convert to organism-to-soil CRs using the wholebody to tissue factors reported in Yankovich et al. (2010).
- *Next preferred:* use organism-to-soil CRs from the wildlife transfer database (Copplestone et al. 2013) and its base references.
- *Least preferred:* use default organism-to-soil CRs for the closest matching terrestrial reference organism in the ERICA Tool (Beresford et al. 2008).

Table 2 Organism-to-soil CRs used to quantify radionuclide transfer to reference organisms[1,2,3]

Organism	Pb	Po	Ra	Th	U
Green tree frog	AM: 1.2×10^{-1} SD: 5.2×10^{-1} N: 24	AM: 2.8×10^{-3} SD: na N: na	AM: 3.6×10^{-2} SD: na N: na	AM: 3.9×10^{-4} SD: na N: na	AM: 5.0×10^{-4} SD: na N: na
Corella	AM: 4.9×10^{-2} SD: na N: 1	AM: 1.0×10^{-2} SD: 2.9×10^{-3} N: 5	AM: 9.7×10^{-3} SD: na N: 1	AM: 3.9×10^{-4} SD: 9.4×10^{-5} N: 20	AM: 5.0×10^{-4} SD: 1.1×10^{-4} N: 20
Spear grass	AM: 2.4×10^{-2} SD: na N: 1	AM: 3.1×10^{-1} SD: 4.9×10^{-1} N: 71	AM: 2.3×10^{-3} SD: na N: 1	AM: 2.3×10^{-4} SD: na N: 1	AM: 3.5×10^{-4} SD: na N: 1
Green ant	AM: 4.0×10^{-1} SD: 4.7×10^{-1} N: 561	AM: 2.8×10^{-3} SD: na N: na	AM: 6.1×10^{-2} SD: 2.8×10^{-2} N: 7	AM: 1.8×10^{-2} SD: 5.0×10^{-3} N: 4	AM: 1.8×10^{-2} SD: 5.0×10^{-3} N: 4
Sand goanna	AM: 2.8×10^{-2} SD: 1.9×10^{-2} N: 2	AM: 2.4×10^{-1} SD: 1.5×10^{-1} N: 2	AM: 7.6×10^{-2} SD: 6.1×10^{-2} N: 2	AM: 1.7×10^{-3} SD: 1.1×10^{-3} N: 3	AM: 3.6×10^{-3} SD: 1.8×10^{-3} N: 4
Olive python	AM: 2.8×10^{-2} SD: 1.9×10^{-2} N: 2	AM: 2.4×10^{-1} SD: 1.5×10^{-1} N: 2	AM: 1.6×10^{-1} SD: na N: 1	AM: 3.4×10^{-3} SD: na N: 1	AM: 2.4×10^{-3} SD: 3.6×10^{-4} N: 2
Cathedral termite	AM: 4.0×10^{-1} SD: 4.7×10^{-1} N: 561	AM: 2.8×10^{-3} SD: na N: na	AM: 6.1×10^{-2} SD: 2.8×10^{-2} N: 7	AM: 1.8×10^{-2} SD: 5.0×10^{-3} N: 4	AM: 1.8×10^{-2} SD: 5.0×10^{-3} N: 4
Agile wallaby	AM: 4.7×10^{-3} SD: 5.1×10^{-3} N: 2	AM: 2.2×10^{-2} SD: na N: 1	AM: 1.6×10^{-2} SD: 1.6×10^{-2} N: 9	AM: 6.6×10^{-3} SD: 1.9×10^{-3} N: 4	AM: 4.5×10^{-4} SD: 2.7×10^{-4} N: 6
Ironwood	AM: 7.6×10^{-2} SD: 1.1×10^{-1} N: 42	AM: 3.8×10^{-2} SD: 2.2×10^{-2} N: 20	AM: 6.8×10^{-4} SD: 7.5×10^{-4} N: 20	AM: 1.1×10^{-3} SD: 1.1×10^{-3} N: 85	AM: 6.8×10^{-3} SD: 1.4×10^{-2} N: 521

1 Values shown are arithmetic mean (AM), standard deviation (SD) and number of samples (N)

2 Values with no shading obtained from the most preferred method, values with light grey shading obtained by the next preferred method, values with dark grey shading obtained by the least preferred method

3 na indicates value not available

Analysis approach

EMCLs were back-calculated against a screening dose rate of 10 μGy h^{-1} for an anticipated mix of ^{238}U in secular equilibrium with its progeny in the waste rock cover of the rehabilitated landform. The ERICA Tool was used to calculate the dose rate to the organism per unit activity concentration of ^{238}U series radionuclides in waste rock (*F*-value) in two ways:

- *Tier 3 analysis*. This approach was used to calculate a highly conservative *F*-value and followed the general approach described in the ERICA Tool help file and Brown et al. (2008). Soil-to-organism CRs were entered as a distribution – lognormal for those with a mean and standard deviation and exponential for those based on a single value only. Soil activity concentrations were entered as 1 Bq kg^{-1} for all radionuclides. The *F*-value was selected as the 95th percentile value of the output dose rate distribution from the analysis. The EMCL was then determined for the limiting organism, which was the organism with the highest *F*-value.

- *Tier 2 analysis*. This approach was used to calculate a typical *F*-value. Organism-to-soil CRs were entered as the mean value only, not a distribution. Soil activity concentrations were entered as 1 Bq kg^{-1} for all radionuclides. The *F*-value was selected as the deterministic output dose rate value from the analysis. An EMCL was then determined for each organism.

Results and discussion

Figure 1 shows the highly conservative EMCL for ^{238}U in secular equilibrium with progeny calculated from the tier 3 analysis. The value was 130 Bq kg^{-1} and comes from olive python as the limiting organism. The above baseline soil ^{238}U activity concentration from the Ranger rehabilitation strategy could be compared to this EMCL to establish the general ecosystem risk context in a similar way to doing a tier 1 assessment in the ERICA Tool, except that the EMCL in this case has been calculated from local organism data. The ERICA Tool does not allow the user to enter any local organism data during a tier 1 assessment – the user is restricted to use the default reference organisms and default CRs within the tool. The assessment should progress to the tier 2 level if the comparison of above baseline soil ^{238}U activity concentration to the EMCL gives a quotient greater than one.

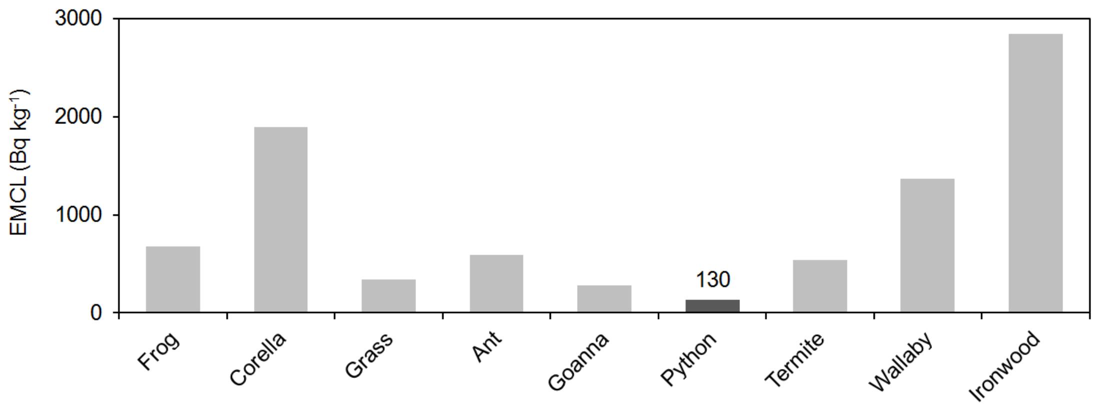

Figure 1 Highly conservative general ecosystem EMCL for ^{238}U in secular equilibrium with progeny from the ERICA Tool tier 3 analysis.

Figure 2 shows the typical EMCL for ^{238}U in secular equilibrium with progeny for each organism calculated from the tier 2 analysis. The values ranged from 350 Bq kg^{-1} for olive python to 6500 Bq kg^{-1} for ironwood. The above baseline soil ^{238}U activity concentration from the Ranger rehabilitation strategy could be compared to these EMCLs to establish the risk context for each organism in a similar way to doing a tier 2 assessment in the ERICA Tool. The assessment of doses to the environment should progress to the tier 3 level for those organisms where the comparison gives a quotient greater than one.

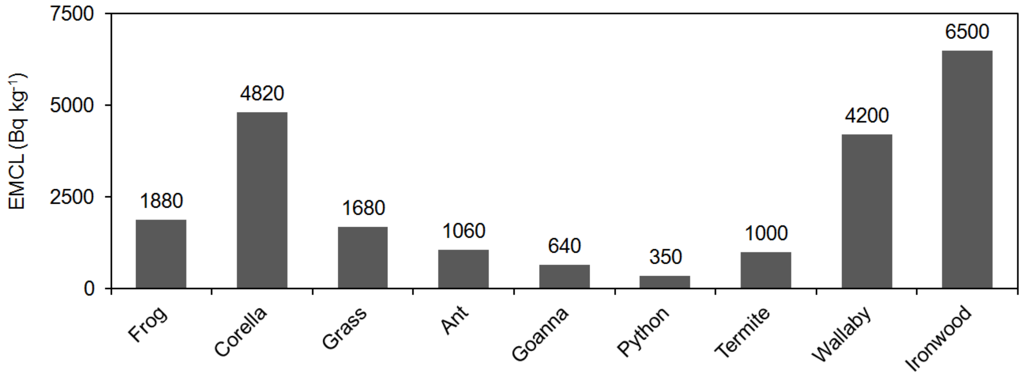

Figure 2 Individual organism EMCLs for 238U in secular equilibrium with progeny from the ERICA Tool tier 2 analysis.

Figure 3 shows the EMCL for each individual radionuclide-organism combination and the relative contribution of the internal and external exposure pathways. The most important radionuclide for animals was ^{226}Ra – the proportion of the EMCL related to this radionuclide was 79% on average and ranged from 60% for sand goanna to 97% for green tree frog. The most important radionuclide for plants was ^{210}Po – the proportion of the EMCL related to this radionuclide was 66% for ironwood and 89% for spear grass. The internal exposure pathway was much more important than the external exposure pathway for all organisms – the proportion of the EMCL related to internal exposure was 92% on average and ranged from 82% for ironwood to 99% for olive python. The dominance of the internal exposure pathway suggests that the most critical parameter in environmental radiological assessment is likely to be soil-to-organism CRs and that research should focus on characterisation of this parameter, especially for ^{226}Ra in animals and ^{210}Po in plants.

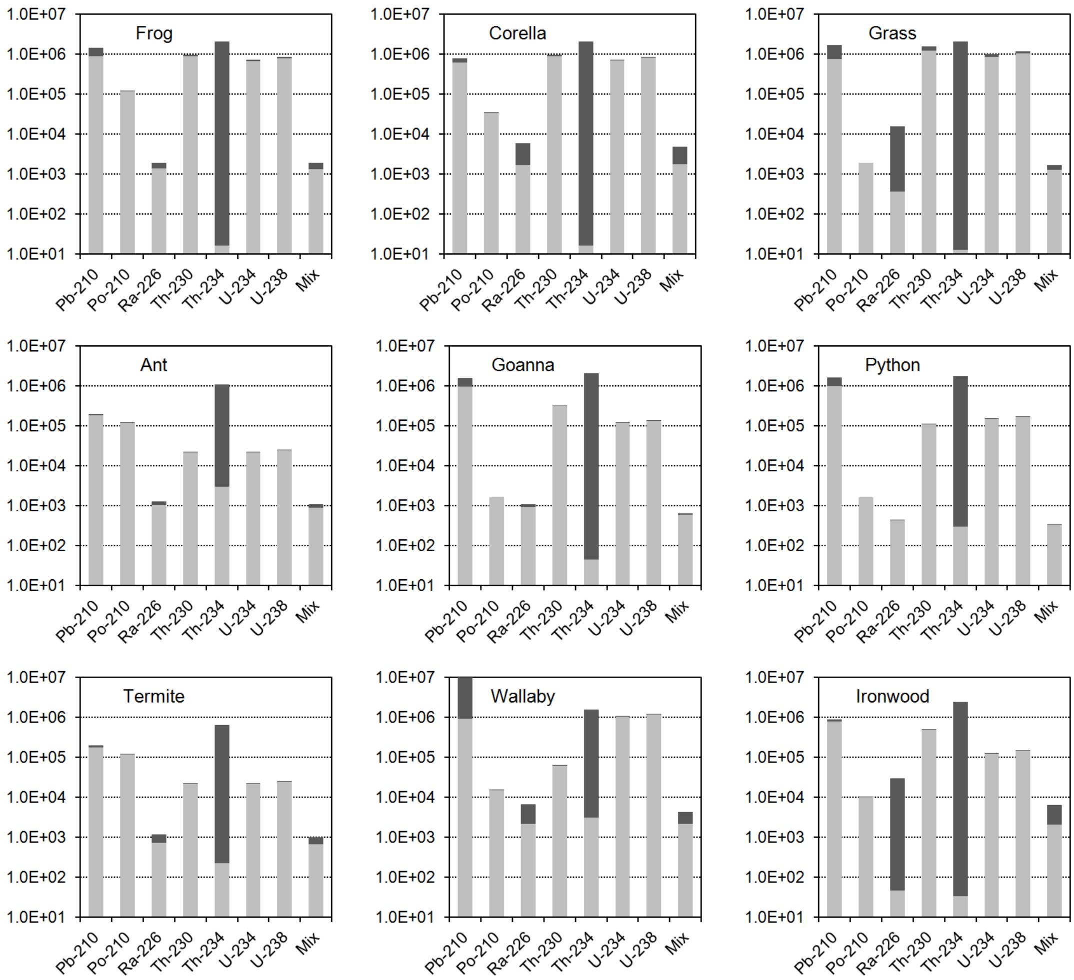

Figure 3 Organism EMCLs (Bq kg^{-1}) for individual radionuclides and ^{238}U series secular equilibrium mix ('Mix'). The light grey part of each column indicates the proportion of the EMCL related to internal exposure and the dark grey part of each column indicates the proportion of the EMCL related to external exposure on a linear scale relative to the total height of the column.

Steps for completion

This calculation of EMCLs has not considered environmental exposures to wildlife from radon progeny in air, as the ERICA Tool is not capable of assessing such exposures. Therefore a step for the future is to investigate the magnitude of radon progeny exposures to wildlife and revise the EMCLs to include this pathway. The approach developed by Vives i Batlle et al. (2012) is the most likely starting point to investigate the magnitude of radon progeny exposures.

Additional samples are being processed and analysed by the ***eriss*** Environmental Radioactivity programme to increase the base data from which organism-to-soil CRs can be derived. In particular, several tissues of wallaby and goanna are currently being analysed for ^{226}Ra. It is expected that this analysis will enable the derivation of wholebody to tissue factors for these organisms for ^{226}Ra and thereby reduce reliance on the generic factors from Yankovich et al. (2010).

A sampling campaign to collect green ants and termites is planned for 2014–15 so that future EMCL calculations for these organisms can be based on local data. Figure 2 suggests that green ants and termites have the next lowest EMCLs after olive python and goanna. Local data for these organisms should provide a better indication of where they fit in the EMCL context for Ranger rehabilitation.

Use of the 10 μGy h^{-1} screening dose rate from the ERICA Tool as the basis for deriving EMCLs needs further consideration. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR 2011) has reviewed the effects data, including the outcomes of the ERICA project, and has concluded that for terrestrial organisms chronic dose rates of less than 100 μGy h^{-1} to the most exposed individuals in a population would be unlikely to have significant effects on most terrestrial organism populations. The International Commission on Radiological Protection (ICRP 2008) has also reviewed the effects data and has suggested that for the most sensitive terrestrial organisms dose rates in the range from 4 to 40 μGy h^{-1} may cause effects to some exposed individuals, but are unlikely to impact whole populations. The question for stakeholders to consider is which dose rate level to use as the basis for deriving soil EMCLs for wildlife-based protection in the Ranger rehabilitation context?

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Key Knowledge Need 2.6: Monitoring

Developing monitoring methods using an Unmanned Aerial System (UAS): Jabiluka revegetation

T Whiteside, R Bartolo & S Grant

Introduction

The Revegetation and Landscape Ecology (RLE) group in ***eriss*** is designing and establishing monitoring programmes for the impending closure of the Ranger minesite. Monitoring requires the assessment of numerous biophysical indicators within the landscape (Ruiz-Jaen & Mitchell Aide 2005) and may include parameters such as: tree and canopy density; ground cover (vegetative and bare ground); tree height; vegetation rigour and health; hydrological change; and erosion. The impact of landscape disturbances such as fire, weeds and cyclones may also need to be measured and accounted for. The biophysical indicators that will be monitored will be selected by the Flora and Fauna and the Landform Technical Working Groups of the Closure Criteria Working Group. For effective monitoring, data and information on these indicators are required at suitable temporal scales (frequency) and within an appropriate spatial sampling resolution and design. Frequent monitoring is particularly critical in the early stages of revegetation, where an understanding of the dynamics of survivability of sensitive life stages may be required by identifying the condition of individual plantings. Recent technological advances have led to UAS (unmanned aerial systems) having lower costs and increased reliability. Sensors onboard UAS capture very high spatial resolution data that not only include imagery of sub-decimetre (< 5 cm) ground sample distance (GSD) but are also the basis for very high resolution, photogrammetric products including 3D point cloud data, digital surface models (DSMs) and digital elevation models (DEMs). Advances in photogrammetric image processing, using multi-view stereopsis (Harwin & Lucieer 2012), have also meant that low cost, lightweight cameras can now produce imagery of a comparable quality as traditional aerial photography. Data of this spatial scale will greatly assist the monitoring of mine closure activities. Advantages over field-based data collection are the availability of complete data coverage in a much shorter amount of time. Advantages over satellite imagery and traditional aerial photography include the higher spatial resolution, lower costs for high temporal resolution sensing and the ability to operate below cloud. UAS have the added flexibility to fly when required with short deployment and data capture times, enabling rapid response to events. In addition, data products can be produced almost immediately after capture. In a minesite rehabilitation context, data captured by UAS for monitoring will be of a relevant spatial and temporal scale, with particular emphasis on monitoring surface conditions, landform changes and vegetation growth.

Rehabilitation work on the Jabiluka minesite has been undertaken by ERA, including the planting of over 3500 trees and shrubs in October 2013. The newly-planted site now provides an ideal base for conducting research and testing methodologies for UAS based monitoring. The main aim of this project is to develop effective techniques for monitoring the rehabilitation and revegetation of minesites, in particular the Jabiluka mine area, using UAS technology. The data from the UAS will be used to measure the success of the early stages of revegetation of the site.

The aims of the project are to:

1. Capture data over the Jabiluka site at approximately monthly intervals.
2. Establish and document formalised procedures for data processing and analysis including the extraction of relevant information, such as landscape metrics, for monitoring rehabilitation.
3. Use UAS data to quantify over time, the revegetation success of the planting areas, revegetation success of species planted and the dominant volunteer species on the site.
4. Report on the findings relating to the success of revegetation in the first 12–18 months of revegetation.

This paper reports on the initial data captures and preliminary data processing and analysis.

Methodology

We have been conducting periodic flights of the rehabilitated Jabiluka site using the Swampfox UAS. The platform is a remotely-piloted, fixed-wing aircraft propelled by an electric motor capable of autonomous flight and controlled via a ground control station (Figure 1a). The main payload for the Swampfox is a dual DSLR camera setup: one camera captures standard colour imagery (Red, Green and Blue bands: RGB) while the other is converted to capture near infrared (NIR) imagery (Figure 1b). The NIR imagery (> 720 nm) is important for assisting with the monitoring of plant vigour.

(a)

(b)

Figure 1 The Swampfox UAS (a) and dual camera payload with modified NIR camera at the top (b).

Flights are conducted at a height of approximately 120 m above the surface, and in adjacent and overlapping transects over the study area to ensure at least 60% side overlap and 90% forward overlap between photos to enable photogrammetric processing. The cameras are triggered automatically, capturing an image every 1.5 seconds.

To date, imagery has been captured over the Jabiluka site for three dates: 28 April, 13 June and 23 September 2014. Ground control points (GCPs) were placed on the site in June using a differential GPS (Global Positioning System) receiver to facilitate accurate geometric correction of the imagery. The accurate alignment of imagery is necessary for fine-scale change analysis. An on-ground survey along a transect placed across the study site was undertaken in September, recording species and life state (alive or dead) to assist in validation of the image analysis.

The hundreds of images captured each flight are processed using photogrammetry software (Pix4D Mapper) using the GCPs, the GPS and attitude data from the UAS flight log, together with the overlapping photos, to create geo-referenced orthomosaics in true colour RGB and NIR, 3D point clouds and digital surface models .

For each date, the NIR mosaic was merged with the RGB mosaic to create a 4-band multispectral image. To assess information potentially useful for monitoring vegetation condition, the Normalised Difference Vegetation Index (NDVI; Rouse et al. 1973) was then calculated for each image. The NDVI is the normalised ratio between the NIR and Red bands of the imagery (equation 1). The NDVI is a well-recognised indicator of canopy biophysical properties such as vegetation fraction, leaf area index, fraction of absorbed, photosynthetically-active radiation, and net primary production (Gitelson 2004).

$$NDVI = {}^{NIR - Red}/_{NIR + Red} \quad (1)$$

Using a subsection of the Jabiluka planting designated as area D as a test site, binary classification (i.e. two classes: above or below a NDVI threshold) of the NDVI imagery was undertaken for each date to look at the effectiveness of the measure at discriminating live plants.

As a validation measure, a visual assessment of the plants (live or dead) was undertaken using the imagery from each date. Within a GIS, each image was visually assessed and the location of plantings was manually determined. From the imagery, the locations of some plantings were unable to be located due to the movement of sand downslope since planting, with some sites being eroded and some sites being covered. The plant state (alive or dead) was recorded at each planting location visible for each date. Volunteers (plants growing on site that were not planted) were also recorded. These validation data were then compared to the NDVI classifications and accuracy assessed.

Results and Progress to date

The initial results from this preliminary data processing show considerable potential of UAS in assessing revegetation efforts. From Figure 2, the drying out of the area can be readily observed over the three dates that span late wet to mid dry season. After an analysis of the NDVI imagery, it was observed that pixels with a value greater than 0.25 belonged to actively photosynthesising vegetation across the three dates. The classification based on this threshold was able to show changes in plant cover on the site for each of the three dates (Figure 3d-f), clearly distinguishing tree mortality and volunteers. By comparing this classification across the dates, survivability of the plantings can be estimated by determining NDVI persistence or reduction through time. In addition, the number of new volunteers between dates can be estimated by looking at new areas of high NDVI that appear at consecutive sampling occasions. This is verified by the validation assessment (Figure 3g-i) where, for April, there was 96.3% accuracy between live plants visually observed and an NDVI value > 0.25. For June, there was a 94.9% accuracy and for September 89.4%. Within the subsection (Figure 3), 109 plants were identified as being alive in April. Of these, 39 had died by June. Conversely, between April and June, 8 volunteer plants were detected in the subsection. Between June and September, 26 plants were observed to have died while a further 14 volunteer plants appeared. This was substantiated by the field-based transect data collected in

September. The transect data showed that most of the volunteers to the site comprised *Acacia* spp.

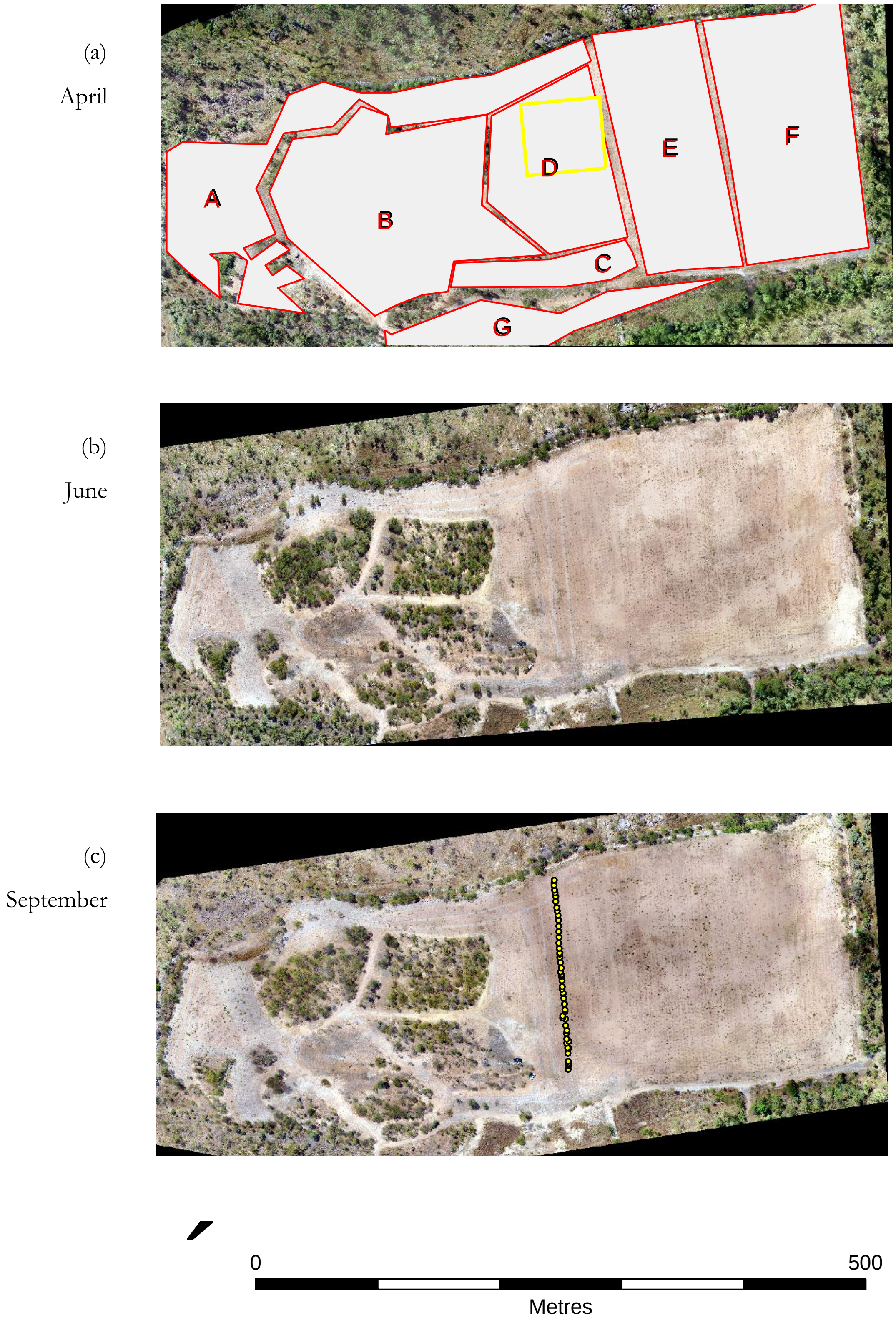

Figure 2 RGB mosaics over Jabiluka for each date: (a) 28 April 2014, (b) 13 June 2014, and (c) 23 September 2014. Red polygons in image (a) are ERA planting areas. The yellow polygon is the subsection for this study. The yellow points in image (c) are records along the field transect undertaken in September.

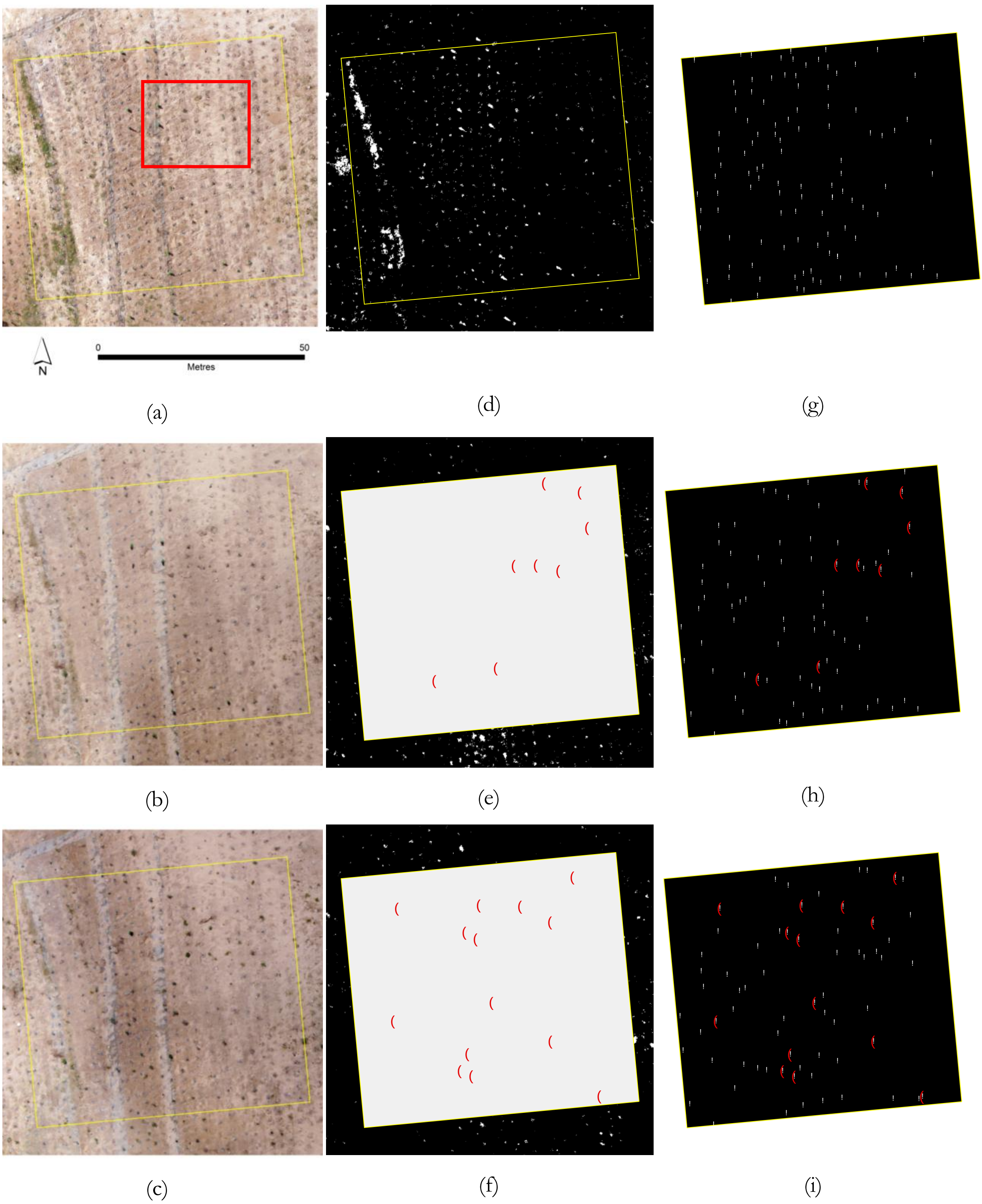

Figure 3 Subsection of the Jabiluka site. Images in the left hand column are the RGB subsets for April (a), June (b) and September (c).The middle column shows classified NDVI (bright areas are over 0.25) for April (d), June (e) and September (f). The right column is the manual delineation of plants for April (g), June (h) and September (i). Red circles indicate volunteer plants not present in the image from the previous date. The red rectangle in image (a) is a zoom-in section for Figure 4.

(a)
April

(b)
June

(c)
September

Figure 4 'Zoomed in' area of the red rectangle shown in Figure 3. (a) April, (b) June and (c) September. Yellow points in image (c) show the plant status along the field transect.

Figure 4 shows the fate of four *Grevillea pteridifolia* plantings within the subsection. In the April image (Figure 4a), all four are alive. In the June image (Figure 4b), it can be observed that the bottom left plant has died, while in the September image (Figure 4c) only the top left plant has survived. The transect data confirm two of the plants' death. Also, a number of volunteers can clearly be seen along the right hand edge of the September image.

Steps for completion

An immediate priority will be to compare survivability data collected on the ground by ERA in June with an analysis of UAS imagery also captured in June. Routine flights will continue to be undertaken over the wet season in order to quantify the impacts of wet season run-off on revegetation success. Flights will also be undertaken on a monthly basis during the 2015 dry season. Methods of species identification will also be investigated using spectral, textural and structural information. Methods for quantitative analysis and scales for reporting results will be assessed before a recommendation is made on sampling frequency beyond the 2015 dry season and how results for revegetation should be reported. A component of the quantitative analysis will be to investigate the utility of information from the UAS imagery-derived DSM for plants identifying plant height and structure.

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Remote sensing analysis of the inter-annual variation of Magela Creek floodplain vegetation: 2010–2013

T Whiteside & R Bartolo

Introduction

This paper presents work undertaken to map the inter-annual variation in floodplain vegetation in the Magela Creek floodplain, a large ecologically significant wetland situated downstream of Ranger uranium mine. The data used were captured by the satellite-based WorldView-2 (WV-2) sensor in the early dry season from 2010 to 2013. The project investigated the application of a transferrable GEOBIA rule set to four dates of high spatial resolution multispectral satellite data to map the floodplain vegetation for each year and undertake change detection analysis between these dates. This time series mapping of floodplain vegetation provides a contemporary baseline of annual vegetation dynamics on the floodplain to assist with future monitoring and analyses of off-site change during and after the rehabilitation of Ranger mine.

The aims of this project were to:

1 Produce high resolution maps of the vegetation communities of the Magela Creek floodplain for 2010–13 (inclusive) to be used as a baseline for monitoring potential impacts arising from rehabilitation of the Ranger minesite.

2 Map and analyse inter-annual change within vegetation communities on the floodplain.

The vegetation communities within the floodplain are seasonally and annually dynamic (Finlayson et al. 1989). While there has been little research on the drivers behind these dynamics, drivers have been hypothesized to include water depth and duration of inundation (Sanderson et al. 1983, Finlayson et al. 1989). Accurate mapping of the vegetation communities within the floodplain at an appropriate spatial and temporal scale will provide information that may be used to determine these drivers in conjunction with other relevant data, such as hydrology. Mapping will also inform ecological risk assessment for floodplain management. Current ecological risks for the floodplain are identified as weeds, feral animals and unmanaged wildfire. Additionally, given that the Magela Creek floodplain is a down-stream receiving environment for the Ranger minesite, off-site monitoring of this area will become increasingly important in the years following decommissioning and rehabilitation of the minesite.

Method

The image data sets for this project consisted of four sets of WV-2 multispectral data captured in the early dry season of 2010 (11 May), 2011 (14 May), 2012 (9 June) and 2013 (2 June) at approximately 11:20 ACST (±6 min). The sensors onboard the WV-2 satellite capture data in 11-bit format and after geometric correction, the image pixels represent 2 m ground sample distance (GSD) at nadir. The multispectral data consist of 8 spectral bands (coastal, blue, green, yellow, red, red edge, NIR1 and NIR2).

Figure 1 shows the workflow for processing and analysing the data. The steps include the application of masks to delineate the floodplain; identify and eliminate cloud and cloud shadow; extract open water objects; and segment and classify floodplain objects.

Image pre-processing

The 2010 imagery was geometrically ortho-rectified to sub-pixel accuracy (RMSE = 1.82 m) using field based ground control points (GCPs) recorded using differential GPS (Whiteside et al. 2013). The 2011, 2012 and 2013 imagery were then geometrically registered to the 2010 imagery. All images were radiometrically calibrated to surface reflectance using the FLAASH atmospheric correction algorithm (Whiteside et al. 2013).

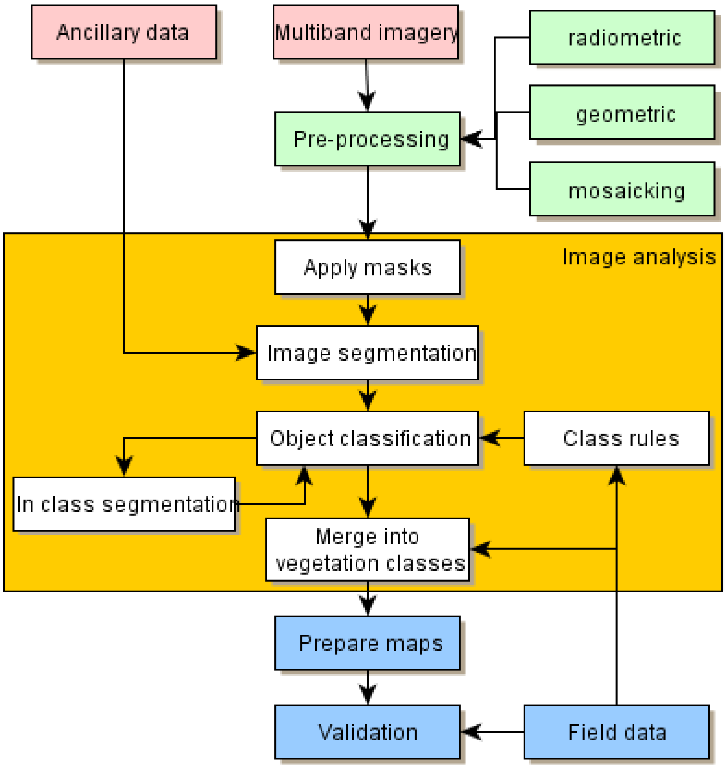

Figure 1 The image analysis approach used for wetland vegetation mapping for all four dates.

Image mosaicking and sub-setting

Prior to creating the image mosaics for each year, null pixels (edge pixels that are not part of the imagery) were converted from the value zero to -1500. If the default value of zero for null pixels was retained, the pixels within the reflectance images with an actual value of zero in some bands (some pixels within water and shadows) would result in null sections within the image during the mosaic process. Any null pixels in the final mosaics were given the value -1500 making it easy to eliminate these pixels from the analysis. All images were then subset to the area common to all dates which is the southern portion of the floodplain.

Ancillary data

A thematic layer delineating the floodplain boundary was included to confine analysis to within the floodplain. The layer was created using a binary classification of SRTM (Shuttle Radar Topography Mission) and aerial photo digital elevation models at a threshold of 6 m. All areas below 6 m were assigned as floodplain and all areas above were assigned as non-floodplain. It was necessary to conduct manual adjustment of the boundary based upon a visual assessment of the boundary using the WV-2 data.

A canopy height model (CHM) was incorporated to differentiate between treed land cover (Melaleuca forest) and spectrally similar non-treed land cover. The CHM was derived from a LiDAR capture conducted within Kakadu National Park between 22 October and 16 November 2011 and had a resolution of 2 m GSD. A brief description of how a CHM was derived can be found in Whiteside and Bartolo (2014).

Classification of floodplain vegetation

The floodplain regions in the imagery for each year were initially segmented using the multi-resolution segmentation algorithm (Baatz & Schäpe 2000) with following parameters: scale = 200; shape = 0.3; and compactness = 0.7.

Classification of treed vegetation classes

The CHM was used to distinguish floodplain objects that contained trees from objects with no trees. The first step identified objects potentially containing trees by using a mean height threshold of 0.8 m derived from the CHM. The objects with a mean height above 0.8 m were classified as potentially treed. Within these potentially treed objects, sub-objects representing trees, or clusters, were created using a threshold segmentation algorithm with trees assumed as being the areas within the CHM above 4 m. The potentially treed objects were then assigned to a class based on the proportion of tree sub-objects per potentially treed object: Open forest was greater than 50% proportional cover; Woodland was between 10 and 50% proportional cover; and Open woodland was less than 10% proportional cover.

Classification of non-treed vegetation classes

The classification of the non-treed image objects into the vegetation community classes was implemented using a set of rules that followed a decision or classification tree (CT) model. The method used in this project differs slightly from CT methods that use a quantitative statistical approach to derive the thresholds and structure of the tree. The process was developed using the spectral information extracted for objects of known vegetation types found within the imagery. The reason for using a knowledge-based classification tree over a statistical-based classification tree was twofold. Firstly, the amount of field-based reference data for the first two years was limited (128 and 68 samples respectively), so it was decided to use all reference data for accuracy assessment and none as training samples for classification. Secondly, the intent was to develop a transferrable methodology for imagery from the same time of year in future years focussing on key spectral indices with only minor changes required for spectral variations between images. Using statistical methods would have produced very different classification tree structures and potentially used different feature variables and thresholds for each image.

The objects representing the non-treed floodplain vegetation community classes were dichotomously classified using a ‘decision tree’ based upon thresholds placed upon four spectral indices (Table 1). Partitioning continued until satisfactory class separation was achieved. The final classes where then assigned into vegetation community classes based upon expert knowledge. Thresholds were determined based upon on expert knowledge of the observed surface spectral variations in the image and through a series of trial and error tests, and were considered optimal for splitting groups of objects. The four spectral indices used were: (i) the forest discrimination index (FDI) after Bunting and Lucas (2006); (ii) the NDVI or Normalised Difference Vegetation Index (Rouse et al. 1973); (iii) the Enhanced Vegetation Index (EVI) (Huete et al. 2002); and (iv) the Lily index (LI), which was specifically created for this project and named initially for ability to delineate *Nelumbo* cover.

Classification validation

Due to the inter-annual variability of the floodplain vegetation, no reference data set from a single date would be suitable to assess the accuracy of all classifications. Therefore, reference data for each year were collected concurrent with the image capture dates. Due to inaccessibility and local hazards, field data are impractical to collect using ground-based methods. In 2010, the field-based reference data were collected using an airboat and low-level helicopter survey. For 2011–13, only helicopter surveys were used, although methods of data collected were modified each year. The classes observed in the compiled reference data were compared to the classified imagery and accuracy measures for each year were calculated using confusion matrices (Congalton & Green 2009).

Table 1 Spectral indices used in this project.

Spectral Index	Algorithm1
Normalized Difference Vegetation Index	$NDVI = \frac{(NIR2 - R)}{(NIR2 + R)}$
Enhanced Vegetation Index	$EVI^2 = \frac{G \times (NIR2 - R)}{(NIR2 + (C1 \times R) + (C2 \times B) + L}$
Forest Discrimination Index	$FDI = NIR2 - (RE + B)$
Lily Index	$LI = (NIR2 + RE) - B$

[1]NIR2, RE, R and B are the near infrared 2 (860–1040 nm), red edge (705-745), red (625-690 nm) and blue (450-510 nm) bands respectively; [2]Within EVI, G = 2.5, C1 = 6, C2 = 7.5 and L = 1

User's and Producer's accuracies (Story & Congalton 1986) were calculated for each class along with the overall classification accuracy. The Producer's accuracy is a measure of omission error and termed as such because the producer of the classified image/map is interested in how well the area under study can be mapped. The User's accuracy is a measure of commission error and termed as such because the user is interested in the reliability of the map (how well the map represents what is really on the ground).

Change analysis

Change detection was undertaken using an object-based process whereby the classified objects were compared to identify changes that occurred between two dates. The change detection was conducted by using a GIS operation of overlaying the layer of classified objects from one year (Date 2 in Figure 2) onto the objects from the previous year (Date 1 in Figure 2). These objects were categorised into two broad groups: (i) 'No change', where the object is the same class for both dates; and (ii) 'Change', where the object is a different class in date 2 compared to date 1 (Figure 3a). The objects that were classified as 'change' were described further using From/To classes (Figure 3b).

From the analyses, three change maps were created 2010–11, 2011–12 and 2012–13. The maps included polygons belonging to consistent classes and polygons belonging to changed classes (From/To classes). For each of the change maps there were potentially 144 From/To classes.

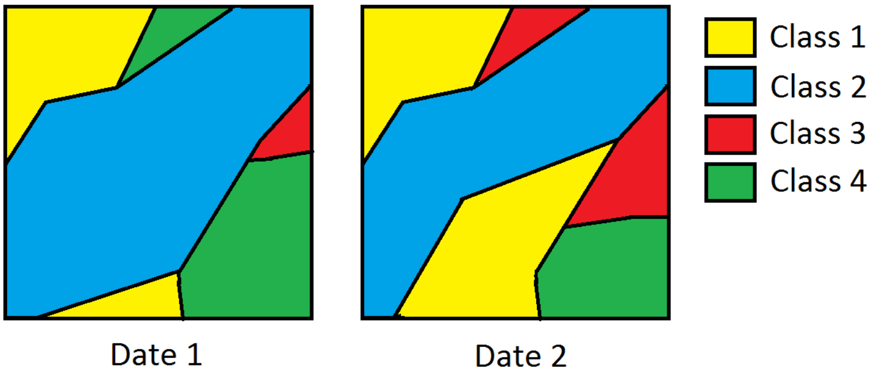

Figure 2 Classified objects for two dates.

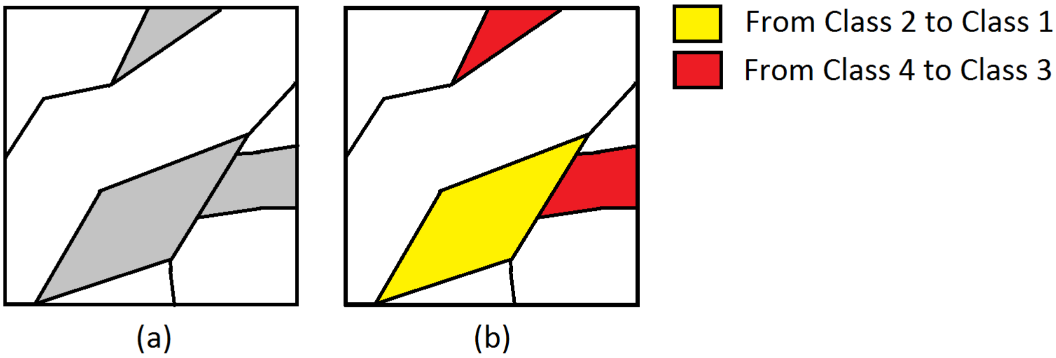

Figure 3 (a) 'Change' (grey) and 'No change' (white) objects based on the overlay of classified objects from the two dates shown in Figure , and (b) From/To classes for objects that changed class between the two dates.

Results and Discussion

Annual vegetation maps

Each of the final vegetation community maps (for 2010, 2011, 2012 and 2013) consisted of an Open water class and 11 vegetation classes labelled based on the dominant Genera for the community. These classes were consistent with those identified in previous research (Finlayson et al. 1989, Boyden et al. 2013): *Eleocharis* sedgeland, *Hymenachne* grassland, *Leersia* grassland, *Melaleuca* open forest, *Melaleuca* woodland, *Nelumbo* herbland, *Oryza* grassland, Para grass (*Urochloa mutica*), *Pseudoraphis* grassland, *Pseudoraphis*/ *Hymenachne* grassland and *Salvinia* floating mats. The 2010 map also has cloud and cloud shadow classes, whereas there was no cloud present in the other years. Areas in hectares for each of the classes for each year are provided in Table 2.

Table 2 Areas (ha) of each vegetation class for each year.

Class	2010	2011	2012	2013
Eleocharis	0	90	0	0
Hymenachne	557	543	348	1232
Leersia	33	177	40	0
Melaleuca open forest	336	249	205	403
Melaleuca woodland	1560	1262	1345	1544
Nelumbo	26	63	144	6
Oryza	396	330	288	1031
Para grass	465	258	499	289
*Pseudoraphi*s	662	460	117	98
Pseudoraphis/Hymenachne	627	92	1176	421
Salvinia	33	80	65	0
Water	382	1686	1007	500
Cloud	186	-	-	-
Cloud shadow	6	-	-	-
Total	**5268**	**5288**	**5234**	**5523**

Accuracy assessment

User and Producer accuracies for each year are shown in Table 3. Overall accuracies for 2010, 2011, 2012 and 2013 were 74.6%, 74.8%, 74.1% and 74.2% respectively. Most of the error is associated with confusion between the classes that were spectrally similar.

Change detection maps

The change detection maps show that: between 2010 and 2011, there was 3877 ha detected as changed cover class, whereas 1236 ha was detected as not changed; between 2011 and 2012, 3591 ha was detected as changed, and 1686 ha detected as not changed; and between 2012 and 2013, there was 3788 ha detected as changed, and 1487 ha detected as not changed (Figure 4). Table 4 is a matrix showing the change in area between classes for the 2012–2013 change map. From the change map and matrices for all three change epochs, the most noticeable change is in the areal extent of open water.

Table 3 User (UA) and Producer (PA) accuracies for each class calculated from confusion matrices for the accuracy assessment.

	2010		2011		2012		2013	
Class	UA	PA	UA	PA	UA	PA	UA	PA
Eleocharis	91.7	61.1	40.0	66.7	100.0	33.3	100.0	75.0
Hymenachne	62.5	75.0	72.7	66.7	66.7	85.7	60.0	60.0
Leersia	100.0	57.1	75.0	75.0	100.0	33.3	100.0	60.0
Melaleuca open forest	75.0	100.0	60.0	100.0	100.0	100.0	100.0	100.0
Melaleuca woodland	50.0	83.3	86.4	73.1	80.0	66.7	88.2	88.2
Nelumbo	71.4	100.0	50.0	100.0	66.7	66.7	83.3	83.3
Oryza	75.0	85.7	60.0	50.0	40.0	40.0	66.7	76.9
Para grass	87.5	87.5	100.0	45.5	66.7	46.2	90.9	52.6
Pseudoraphis	72.2	81.3	28.6	80.0	83.3	62.5	66.7	60.0
Pseudoraphis/Hymenachne	62.5	71.4	83.3	62.5	60.7	100.0	44.7	68.0
Salvinia	80.0	72.7	100.0	60.0	66.7	66.7	77.8	87.5
Water	100.0	64.3	93.1	100.0	95.0	100.0	92.3	80.0

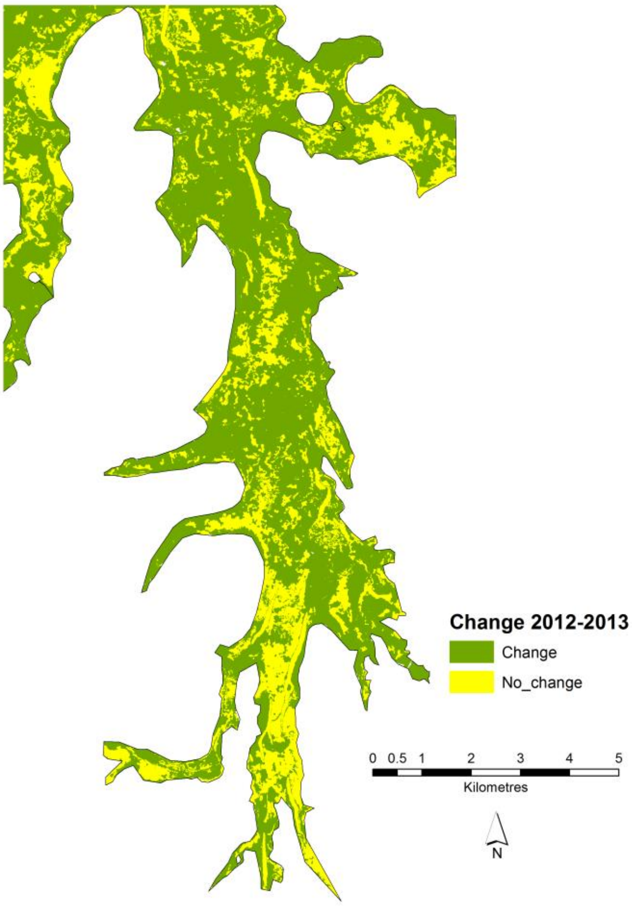

Figure 4 Change detection map for 2012–2013.

There was an increase in the area of surface water from 2010 to 2011, and again for 2011 to 2012 and a reduction between 2012 and 2013. This corresponds with the available rainfall (Figure 5) and surface water discharge data for the region over that period.

Results of the change analysis indicate that there are large areas that change composition each year. Figure 6 shows the change in open water on the floodplain between 2010 and 2011. Areas of open water that were persistent between images include the permanent billabongs on the floodplain. The area of vegetation in 2010 that became water in 2011 was over 1300 ha, whereas the area of open water that became vegetation over the same period was only 106 ha (Table 5). The change in open water on the floodplain between 2011 and 2012 is also shown in Figure 6. The area of open water that was persistent between the images had increased to over 800 ha. The area of vegetation in 2011 that became water in 2012 was over 187 ha, whereas the area of open water that became vegetation over the same period was 880 ha. When examining the change in open water extent on the floodplain between 2012 and 2013, the area of open water that was persistent between the images had decreased to 370 ha. The area of vegetation in 2012 that became water in 2013 was over 130 ha, whereas the area of open water that became vegetation over the same period was 687 ha.

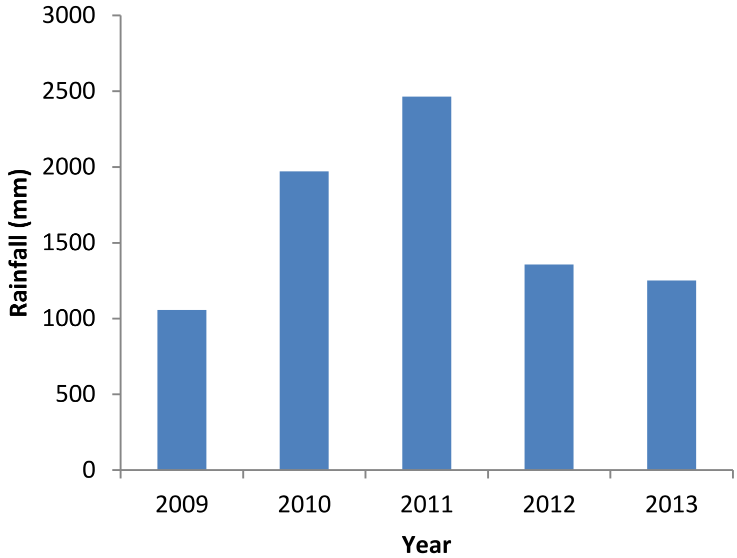

Figure 5 Total annual rainfall for Jabiru airport 2009–2013 (source: www.bom.gov.au)

Table 4 Matrix showing areas (ha) of change in From/To classes between 2012 and 2013. Shaded cells are areas where classes did not change between the dates.

2012 class	2013 class												
	Eleocharis	*Hymenachne*	*Leersia*	MOF	MW	*Nelumbo*	*Oryza*	Para	*Pseudoraphis*	*Pseudoraphis/Hymenachne*	*Salvinia*	Water	Total
Eleocharis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hymenachne	0.0	92.0	0.0	11.6	130.0	0.0	73.9	7.8	1.3	24.2	0.0	7.8	348.6
Leersia	0.0	8.7	0.0	0.5	20.2	0.0	4.2	0.6	0.0	4.0	0.0	1.8	40.0
MOF	0.0	52.1	0.0	3.7	40.5	0.0	69.2	14.4	1.2	19.5	0.0	4.3	204.9
MW	0.0	259.4	0.0	106.4	692.3	0.0	136.0	71.9	13.9	37.7	0.0	23.8	1341.4
Nelumbo	0.0	36.1	0.0	1.3	53.3	0.0	32.0	3.2	0.4	0.0	0.0	4.4	130.7
Oryza	0.0	0.0	0.0	0.0	0.0	107.7	135.4	14.6	1.2	23.9	0.0	5.2	288.1
Para	0.0	109.4	0.0	22.4	174.2	0.0	89.1	59.2	1.8	33.2	0.0	7.8	497.2
Pseudoraphis	0.0	22.8	0.0	5.0	34.1	0.3	34.9	2.9	1.3	12.3	0.0	3.6	117.2
Pseudoraphis/Hymenachne	0.0	402.8	0.0	14.5	191.0	0.5	273.1	59.0	26.7	133.3	0.0	71.0	1171.9
Salvinia	0.0	8.1	0.0	0.7	34.9	0.0	9.9	9.4	0.0	1.8	0.0	0.7	65.5
Water	0.0	174.2	0.0	7.0	124.7	4.9	166.9	43.5	49.7	115.8	0.0	369.9	1056.7
Total	0	1166	0	173	1495	113	1025	287	98	406	0	500	5262.2

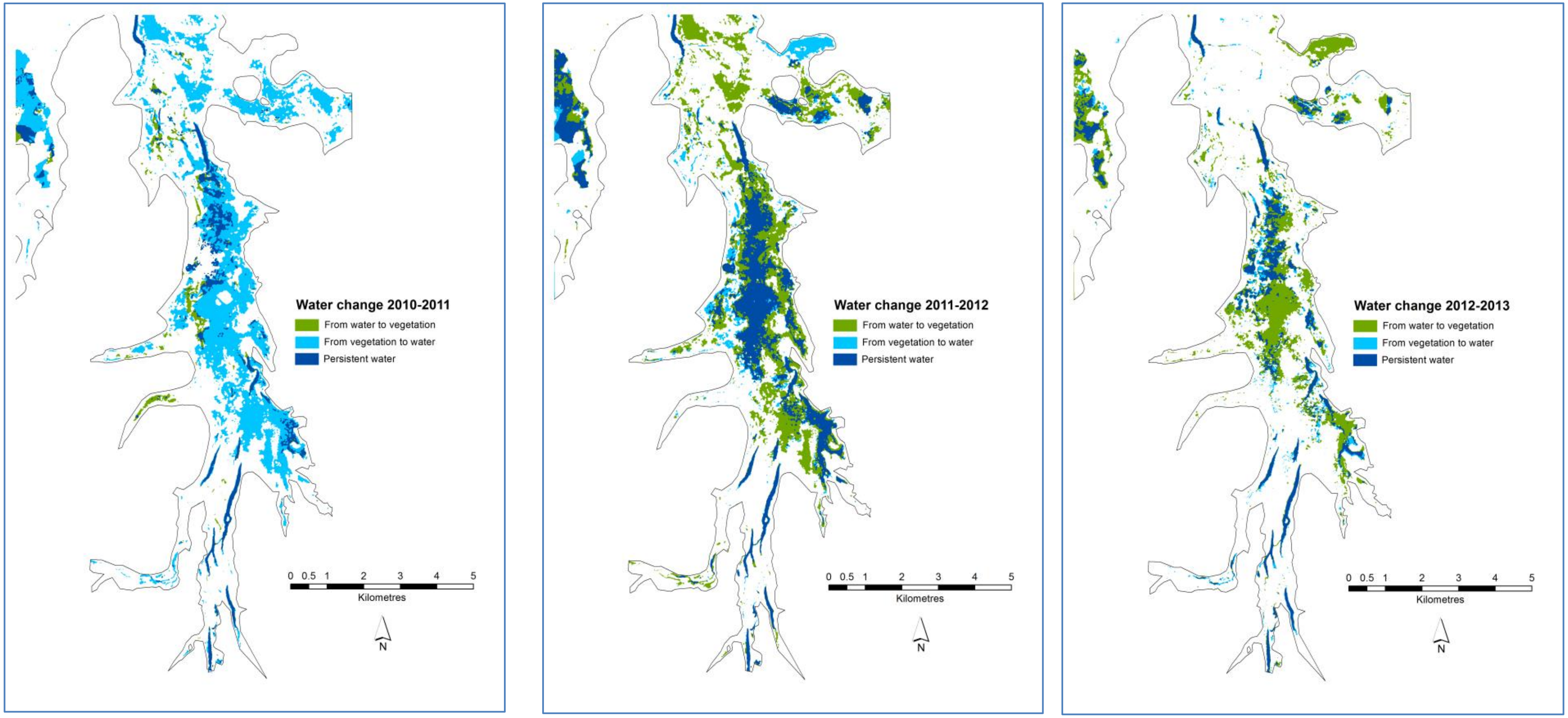

Figure 6 Change in open water for the three change periods (2010–2011, 2011–2012 and 2012–2013) Persistent water is open water detected in both the first and second year of each map.

Table 5 Areas (ha) of water-related change between dates. Persistent water is open water detected in both years for each period.

	Area (ha)		
Condition	2010–2011	2011–2012	2012–2013
Persistent water	284	802	370
From vegetation to water	1308	187	130
From water to vegetation	106	880	687

Summary and further work

This study shows that WV-2 imagery can be used for change detection analysis of a spatially and temporally dynamic wetland such as the Magela Creek floodplain. Even though the imagery is from the same time of year for all years there are a number of issues to consider when using the data. Extrinsic factors such as atmospheric effects, the sensor view angle and the influence of sub-canopy water all influence the spectral response and need to be considered. Intrinsic factors such as the variations in plant phenology due to variations in water availability are another consideration.

The data analysis shows that vegetation and water cover on the floodplain does vary dramatically from year to year. This variation is most likely related to differences in floodplain water depth and extent associated with rainfall and discharge in the catchment. This study does not show the intra-annual change that occurs on the floodplain that is related to the fluctuation of water level and availability throughout the seasons. Future monitoring needs to recognise the dynamics of the system and any assessments of environmental impacts in the region need to acknowledge the variability and that recorded change may not be the result of any degradation of the system.

We now have an operational procedure to monitor change in the downstream off-site wetland environment using high resolution satellite imagery. This will become increasingly important in the years following rehabilitation of Ranger minesite. Further work will involve publication of the method and a more detailed analysis of changes, particularly when a vegetation community changes to another vegetation community. Drivers and causes of change need to be investigated so that we have an adequate baseline understanding of natural drivers compared with potential drivers from minesite rehabilitation. Imagery from June 2014 will also be processed and added to the change series.

With the launch of the WorldView-3 (WV-3) satellite in August 2014, we will need to look at transferring and adjusting our method to WV-3 imagery, which provides increased spatial and spectral resolution. Spatial resolution for the Panchromatic band is now 31 cm and 1.24 m for the 8 multispectral bands. There are also 8 Shortwave Infrared (SWIR) bands (1195 nm - 2365 nm) with a spatial resolution of 3.7 m on board WV-3, which will enable us to 'see' through smoke. Currently we are restricted to a very small 'optically clear' capture window for imagery, when there are no wet season clouds and prior to the commencement of dry season fires. The SWIR bands will enable the capture window to extend further into the dry season. More importantly, the SWIR bands will increase the amount of information we can use to classify vegetation. We plan to acquire a WV-3 scene for May-June 2015.

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Key Knowledge Need 2.7: Risk Assessment

Rehabilitation and closure ecological risk assessment for the Ranger uranium mine: Identification and incorporation of key ecological processes

R Bartolo, A George, R van Dam, A Harford & C Humphrey

Introduction

Energy Resources Australia Pty Ltd (ERA) is required to rehabilitate Ranger uranium mine by 2026 and thus a large number of research and assessment projects are underway by both ***eriss*** and ERA to ensure the necessary and appropriate knowledge is available to inform the rehabilitation and closure strategy. ***eriss*** and ERA are collaborating on an ecological risk assessment for the rehabilitation and closure of the Ranger uranium mine. The rehabilitation ecological risk assessment provides a structured and comprehensive framework for confirming that the key issues related to ensuring the protection of the off-site environment and successful rehabilitation of the on-site environment are identified.

The ecological risk assessment has been broken into the following three phases: (1) Problem Formulation; (2) Risk Analysis; and (3) Interpretation of results (Figure 1). The conceptual models produced during the Problem Formulation phase and reported at ARRTC 32 were finalised in late 2013 and have been published in Internal Report 624 that is available on SS's website. The conceptual models were designed for the assets or components (e.g. biodiversity) which are sustained by ecological processes. Ecological processes may be a suitable assessment endpoint, but their value is derived from the direct assessment and measure of key ecological components. However, ecological processes required further definition, delineation and prioritisation to be successfully incorporated into the current ecological risk assessment.

The importance of ecological processes

The maintenance of ecological processes is specifically referenced in the Environmental Requirements (ERs) as a primary environmental objective for protection of the environment. Ecological processes and functions were identified in the conceptual models for aquatic ecosystems during the initial workshop to develop these models. However, the link between these assessment endpoints and the measurement endpoints that account for ecosystem functions were not clear in the final conceptual models. In order to incorporate ecological processes and tangible measurement endpoints in the conceptual models, further work has focused on defining and understanding the importance of ecological processes for the on-site and off-site environment. The aim of this work is to determine whether the existing conceptual models sufficiently address ecological processes, or if ecological processes require further articulation and characterisation.

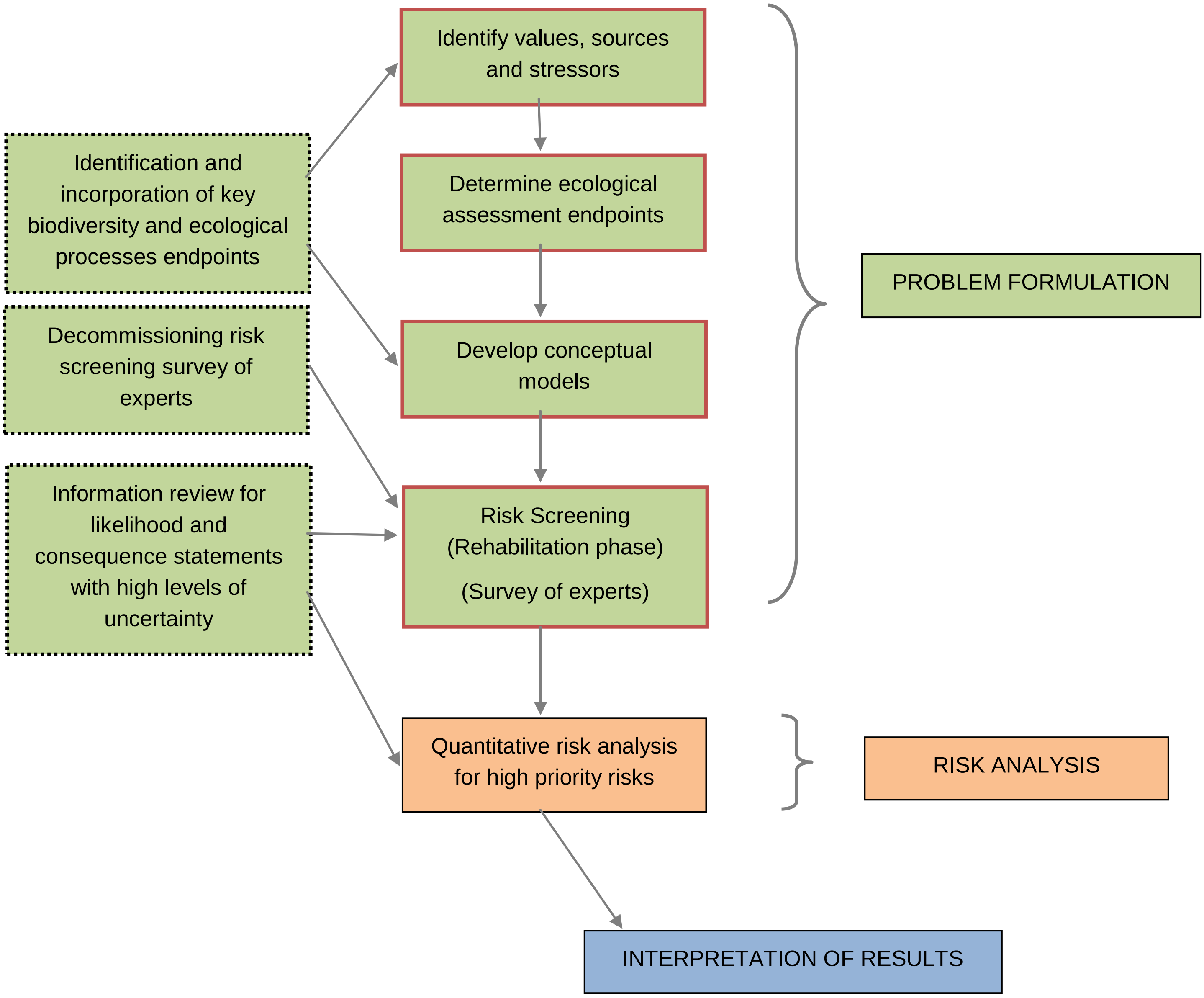

Figure 1 The phases of the ecological risk assessment showing progress to date. Those boxes outlined in red have been completed, whilst those boxes with dashed outlines are in progress.

Candidate abiotic and biotic processes were initially compiled and defined for the on-site and off-site environment (Table 1). Following this, their importance in habitats was ranked overall, and then separately for the dry season and wet season, during a workshop held in February 2014. The ranking of importance of the process to each habitat was assigned to a three point scale based on activity.

Table 1 Abiotic and biotic processes for the on-site and off-site environment identified for the initial processes ranking.

Abiotic	Biotic
Formation of habitat	Movement of organisms: Recruitment, regeneration and dispersal
Chemical processes	Primary Productivity: Phytoplankton and Macrophytes
Hydrological processes	Predation, herbivory, competition, parasitism, mutualism
Natural disturbance – Fire, cyclone, drought and flood	
Geomorphic processes	

During a workshop in late September 2014, the key stressors were assigned across the habitats and ecological processes, and an assessment was made as to whether the stressors will vary between the dry and wet season. The workshop participants also reviewed a list of additional ecological processes for aquatic and terrestrial ecosystems, which were identified from literature searches subsequent to initial compilation and assessment. The additional processes were reviewed and, where possible, reconciled (e.g. merged) with the previous processes. An additional requirement of the agreed, final list of processes is that they represent the full spectrum of spatial and temporal scales, and thereby provide a landscape approach to ecosystem rehabilitation. Although the processes are divided into abiotic and biotic processes, it is well recognised that ecological processes are inherently interconnected and that often a process can be interpreted as either abiotic or biotic, or both (e.g. nutrient dynamics).

Once the revised list of ecological processes has been agreed to, methods for monitoring these processes will be investigated and reported. This will include an evaluation of the extent to which existing monitoring programmes act as surrogates for the key ecological processes.

Risk screening

The results of the risk screening are currently under final review. Information has recently been collated on those responses for likelihood and consequence statements that had high levels of uncertainty. This information is being summarised and used to populate a literature database. A follow-up survey is currently being underway to obtain scores for consequence and likelihood statements focused on the decommissioning phase. These scores will be analysed and reported in the same manner as for the post-decommissioning timeframe.

Further work

Once the risk screening work is finalized and completed, we will commence the risk analysis phase. A review of quantitative risk analysis methods will be completed prior to determining those risks that require a quantitative assessment. The details and results of this work will be published in the per reviewed literature.

Part 3: Jabiluka

Surface Water Quality Monitoring: Jabiluka

A Sinclair

Flow was first recorded at the Ngarradj (Swift Creek) monitoring station on 27 November 2013 and was very low at the start of the wet season with the multiprobes only submerged for short periods of time when the water level was sufficiently high. Increased flow from 9 January 2014 resulted in the multiprobes being fully submerged. EC decreased and remained below 20 µS cm^{-1} (Figure 1).

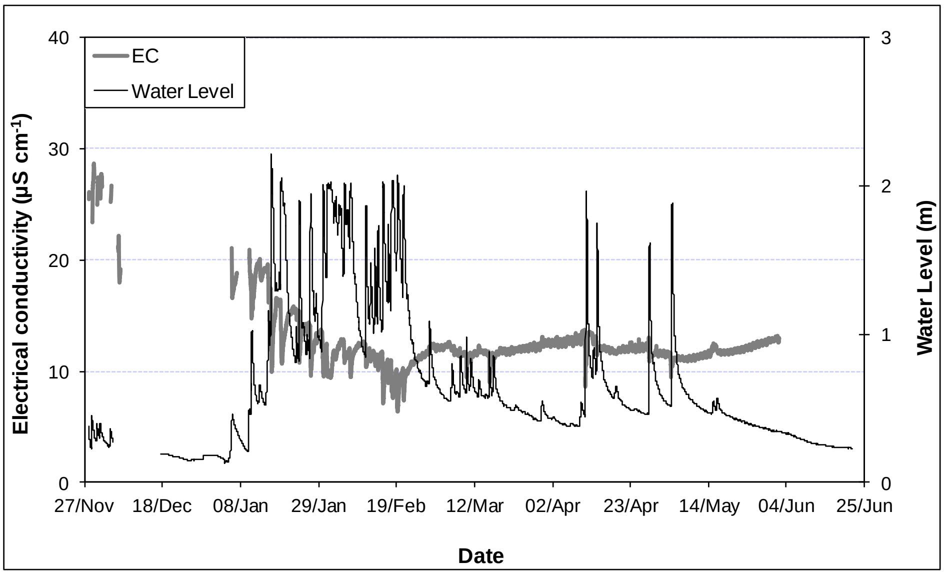

Figure 1 Ngarradj (Swift Creek) continuous EC monitoring data during the 2013–14 wet season.

On 26 January 2014 a turbidity event of occurred during a 90 mm rainfall event at the site (Figure 2). Trigger values for turbidity within Ngarradj (Swift Creek) were not included within the Water Quality Objectives as determined by the Jabiluka Minesite Technical Committee on 21 September 2001. However, baseline values for the physical and chemical characteristics of streams within the Jabiluka lease were established by Cusbert et al. in 1998[1] including 'low risk trigger value' ranges for ecosystem protection. The trigger value range proposed for turbidity was 4.0–105, with non-compliance being turbidity events due to mining activity < 4.0 NTU or > 105 NTU. Thus, the turbidity event of 26 January 2014 of 56.5 NTU falls in the middle of this 'acceptable' range. There have previously been 20 turbidity events > 50 NTU at Ngarradj since continuous monitoring of turbidity began in the 2003–04 wet season.

Another turbidity event occurred on 10 April 2014 during a 90 mm rainfall event at the site and peaked at 112 NTU. This peak was above the upper guideline level for the 'acceptable' range (105 NTU) for less than 10 minutes. There have previously been 3 turbidity events > 105 NTU at Ngarradj since continuous monitoring of turbidity began in the 2003–04 wet season.

[1] Cusbert P, Klessa D, leGras C, Moliere D & Rusten K 1998. Baseline values for physical and chemical indicators in streams of the Jabiluka lease area. Part 1. Interim findings. Internal Report 300. September, Supervising Scientist, Darwin.

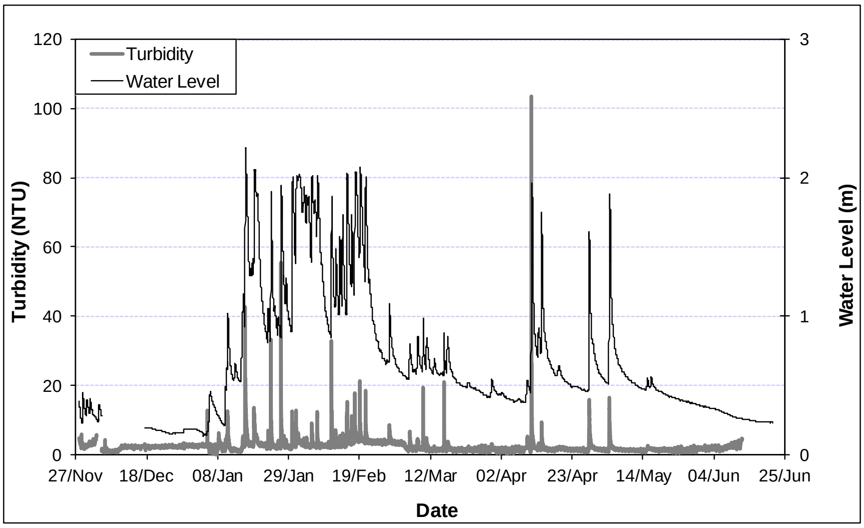

Figure 2 Ngarradj (Swift Creek) continuous turbidity monitoring data during the 2013–14 wet season.

During late February and March 2014 the water levels within Ngarradj decreased leading to gradually increasing EC levels, fluctuating with each rainfall event. Conductivity stabilised in Ngarradj through April and May with the creek entering its recessional flow period.

Continuous monitoring of Ngarradj continued until 12 June 2014 when the multiprobes were out of the water and could not be lowered any further. Cease to flow was agreed by stakeholders on 13 June 2014.

Overall, once flow commenced in the creek, the water quality measured in Ngarradj for the 2013–14 wet season was comparable with previous wet seasons (Figure 3).

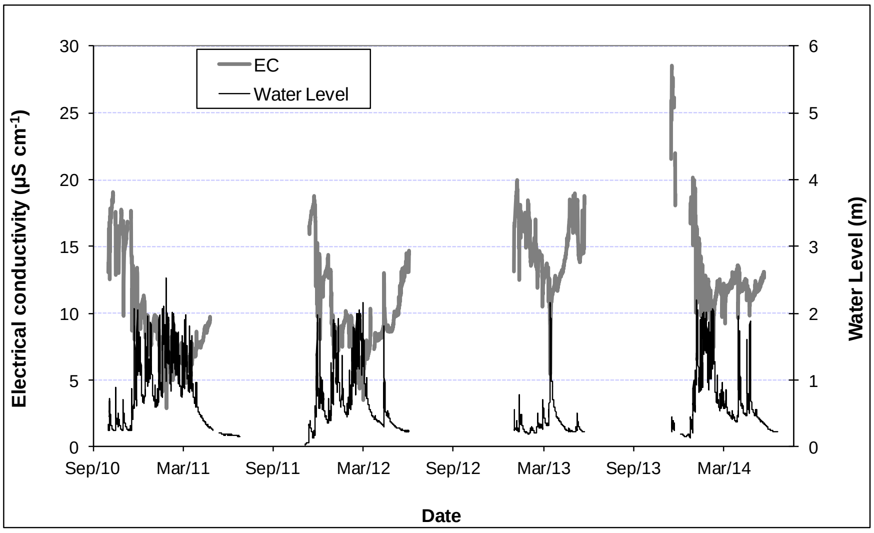

Figure 3 Continuous electrical conductivity and water level (lower trace) in Ngarradj (Swift Creek) for each wet season between September 2010 and June 2014 (values averaged over a 90 minute period of measurement).

Sediment movement off the rehabilitated Jabiluka minesite

M Saynor and W Erskine[1,2]

Background

The Jabiluka mine site remains in long-term care and maintenance. The removal of the Interim Water Management pond (IWMP) was completed in October 2013. As of February 2014, 36000 individual tube stock had been planted within the Jabiluka mine site footprint with survival of 48% noted during the June 2014 Routine Periodic Inspection (RPI) (Supervising Scientist 2014).

Jabiluka mine site RPI visit 20 February 2014

An RPI was completed on 20 February 2014 to look at the wet season impacts on the rehabilitation works at the Jabiluka mine site. ***eriss*** staff (Mike Saynor and Wayne Erskine) joined the RPI team at the Jabiluka mine site. On the approach to land at the site, water could be seen on the surface of the lower half of the slope (previously where the IWMP had been located)(Figure 1). The water was not associated with any visible drainage depressions and was general seepage in the area. The flow of water would not impact on the erosion rates of sediment but could potentially waterlog/drown the tube stock. Also on the approach by helicopter to the Jabiluka mine site areas of sand build up could be seen at locations across the site.

Figure 1 Aerial view looking east over the rehabilitated IWMP. Water on the lower half of the slopes suggest a seep where water is coming out of the ground. Image taken 20-2-2014.

[1] Conjoint Professor - School of Environmental and Life Sciences, The University of Newcastle – Ourimbah Campus

[2] Professorial fellow - Research Institute for the Environment and Livelihoods, Charles Darwin University

Figure 2 Jabiluka mine site April 2014. Letters refer to photo locations in the document.

KKN 3.1.1 Monitoring during the care and maintenance phase

Ground inspections at the Jabiluka mine site showed many areas where sand had been eroded from steeper slopes and deposited on the more gentle slopes below. Figure 2 is an oblique aerial image taken by Unmanned Aerial Vehicle (UAV) in April 2014. Annotated letters on Figure 2 show the location of the following images.

Figure 3 shows sand deposited below the steeper area at the north western end of the steeper slope (Site A on Figure 2). Figure 4 also shows sand deposited on the steeper slope at Site B, but in the jute cloth which was placed as a crude erosion control measure.

Figure 3 Sediment deposited below the steeper slope in the north western area of the Jabiluka mine site area. Image taken 20-2-2014.

Erosion at the site was generally from the steeper slopes, transported down the slope and then deposited on the less steep slopes, this occurred at the following areas shown on Figure 2, C, D, E & F.

Pedestal erosion was present at the site, either larger rocks protecting sediment while the sediment adjacent is eroded (Figure 5) or protective matting around the tube stock (Figure 5) acting as a surface protection.

Figure 4 Sand sediment on the steeper slope, some caught in the jute, some being transport downslope across the jute. Image taken 20-2-2014.

Figure 5 Pedestals cause by large rocks protecting the sediment underneath whilst adjacent sediment is eroded. Image taken 20-2-2014

Figure 6 Protective matting has prevented the erosion of sediment underneath it, whilst surrounding sediment has been transported away. Image taken 20-2-2014

Figures 7 and 8 show sand that has been deposited in the area of Site F (Figure 2). Sand has also been deposited further down the slope at Sites E & D (Figure 2).

Figure 7 Sand deposited on the lower slope which has less angle. Image taken 20-2-2014 looking south toward Site F on Figure 2.

Figure 8 Sand deposited on the lower slope which has less angle. Image taken 20-2-2014 looking east in the vicinity of Site E on Figure 2.

Sand has not only been eroded, transported and deposited on the Jabiluka mine site but has been transported into the two mine site tributaries, Tributary North and Tributary Central. Figure 9 shows sand that has been deposited near to Tributary North (Site G on Figure 2) and Figure 10 shows the creek at the same location. Sand can be seen in the creek bed amongst the rocks that have been used to line the part of the creek that was realigned during construction of the Jabiluka mine site. Figure 11 shows another part of Tributary North with large amounts of sand deposited amongst the rocks, almost covering and swamping them.

After the visit on 20 February 2014, ***eriss*** staff were concerned that there would be large amounts of sand transported downstream by the two mine site tributaries, from the Jabiluka mine site. A dry season visit was undertaken to determine how far downstream the sediment had moved.

Dry Season field trip to Jabiluka mine site

During the week beginning 14 July 2014, ***eriss*** staff (Mike Saynor, Wayne Erskine and Richard Houghton) visited the two mine site tributaries at Jabiluka mine site. A gauging station is located on each of the mine site tributaries just downstream of the Jabiluka mine site. Both of these gauging stations have a concrete weir structure (to enable the station be to rated to calculate discharge) with an upstream weir pool (an area of the creek bed upstream of the weir is also concreted). ***eriss*** staff thought that these weir pools would be full of sand and that the sand would have been transported further downstream.

Both of these gauging stations were visited and neither of the weir pools had any sand deposited in the weir pool. On Tributary North the sand was located approximately 30 m upstream of the weir pool (Figures 12 & 13). Tributary Central was not investigated to see where the sand was due to time of day, however it had not yet reached the weir pool at the gauging station.

Figure 9 Sand deposited on the area just upslope of the bank of Tributary North at Site G on Figure 2. Image taken 20-2-2014.

Figure 10 Sand deposited in the creek down from Figure 9. Image taken 20-2-2014.

Figure 11 Sand deposited in Tributary North. Image taken 20-2-2014.

Figure 12 Tributary North looking downstream towards the gauging station. Image taken 17-7-14.

Figure 13 Tributary North looking upstream at the front of the sand slug. The open spaces of the rehabilitated area can be seen through trees at the top left of the image. Image taken 17-7-14.

Figure 14 shows the sand that has been deposited towards the bottom of the Jabiluka mine site at Site D. A mass of sand can be seen as well as tress and low ground cover shrubs.

Figure 14 Looking west up the Jabiluka mine site from the fenceline over Site D (Figure 2). Areas of sand have been deposited towards the bottom of the slope. Image taken 17-7-14.

Conclusions and Recommendation

There has been a large amount of erosion and sediment movement of sand on the rehabilitated Jabiluka mine site during the 2014–2015 wet season. Most of the erosion was of sand which was eroded off the steeper slopes at the site. The eroded material was deposited on the lower (less angled) slopes, and some of the sand was transported into the two mine site tributaries. The sand had not moved as far downstream as was expected. On Tributary North the sand front was approximately 30 m upstream of the Tributary North Gauging Station. Given the nature of the sandstone parent material and the type of sediment that is being transported (medium to coarse sand) there would not be much fine sediment available to be transported.

A turbidity probe has been in intermittent operation at the Swift Creek Gauging station (approximately 1.2 km downstream of the Jabiluka mine site) since 2004. A cursory look at the turbidity data does not suggest elevated turbidity values compared to previous wet seasons. The largest turbidity value of 112 NTU's occurred on 10/4/14 and was associated with a storm event of 62 mm. Several other turbidity events of this magnitude have occurred in previous years.

It is recommended that the two mine site tributaries be inspected during the 2015 dry season to see how far the sand has moved during the 2014–2015 wet season. Sediment samples should also be taken from the inchannel sediment and the depositional areas on the rehabilitated Jabiluka mine site for particle size comparisons.

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Appendix 1

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Appendix 2
List of inactive, delayed or other projects for which summaries are not provided in this report

KKN / Project	Comments
KKN 1.2.4 Ecotoxicology	
The direct effects of suspended sediment on tropical freshwater biota	Project suspended due to higher priorities and resource constraints.
Re-analysis of existing uranium freshwater chronic toxicity data to calculate low effect concentrations	Revision of site-specific uranium trigger values delayed due to need to conduct further chronic toxicity testing for the snail, *Amerianna cumingi*. Completion will be in early 2015.
KKN 2.6.1-Monitoring of the rehabilitated landform	
Development of a spectral library for minesite rehabilitation assessment – vegetation components	All work on this project has been finalised and an internal report is currently in publication.
Development and implementation of a remote sensing framework for environmental monitoring within the Alligator Rivers Region (focus on the Magela Floodplain)	All work on this project has been finalised and an internal report is currently in publication.
Geological province of fine suspended sediment within the Magela Creek catchment	Project suspended due to higher priorities and resource constraints.
Analysis of landscape change on the Ranger site pre-mine using historical aerial photography	Data has been prepared however analysis has yet to commence.
Development of a method for continuous monitoring of vegetation regrowth on a rehabilitated minesite using a simple LED spectroradiometer	Project was presented in ARRTC31, sensors have been deployed on the TLF and experiments are currently being conducted.
KKN 2.6.2 Off-site monitoring during and following rehabilitation	
Assessment of the significance of extreme events in the Alligator Rivers Region – impact of Cyclone Monica on Gulungul Creek catchment, Ranger mine site and Nabarlek area.	Project suspended due to higher priorities and resource constraints.
KKN 5.1.1 Develop a landscape-scale ecological risk assessment framework for the Magela catchment	
Use of LiDAR and other Digital Elevation Data to develop a landscape model of the Magela catchment	Project no longer required. A product was developed by CSIRO through the NERP Northern Australia hub. We have the product but are yet to validate/use it.
KKN 5.2.1 Assessment of past mining and milling sites in the South Alligator River valley	
Radiological monitoring and assessment of the El Sherana containment	Biennial radiological monitoring is required. Results from September 2013 reported in last ARRTC report. Measurements planned for early dry season 2015.
KKN 5.1.1 Develop a landscape-scale ecological risk assessment framework for the Magela catchment	
Use of LiDAR and other Digital Elevation Data to develop a landscape model of the Magela catchment	Project no longer required. A product was developed by CSIRO through the NERP Northern Australia hub. We have the product but are yet to validate/use it.
KKN 2.5.3 Establishment and sustainability of ecosystems on mine landform	
Aquatic ecosystem establishment	Limited progress on the literature review and aquatic vegetation analyses was possible in 2013–14 due to higher priorities and resource constraints. Progress and revised research proposals arising from ERISS-ERA consultations were presented to ARRTC33. New focus and direction has commenced on aquatic vegetation communities and will be reported on in the next annual research summary (2014-15).

Appendix 3
Alligator Rivers Region Technical Committee Key Knowledge Needs 2008–2010: Uranium mining in the Alligator Rivers Region

The Alligator Rivers Region Technical Committee (ARRTC) last revised the KKNs in 2007-2008, with the resultant (2008-2010) KKNs reproduced below. Although the 2008–2010 KKNs are still being reported against, given their date range they are currently under review again. The revision is being largely informed by an ecological risk assessment of the rehabilitation and closure phases of Ranger, the screening phase of which will be completed by the end of 2014-2015. It is expected that a revised list of KKNs will be finalised by the end of 2015.

Overall objective

To undertake relevant research that will generate knowledge leading to improved management and protection of the ARR and monitoring that will be sufficiently sensitive to assess whether or not the environment is protected to the high standard demanded by the Australian Government and community.

Background

In assessing the Key Knowledge Needs for research and monitoring in the Alligator Rivers Region, ARRTC has taken into account current mining plans in the region and the standards for environmental protection and rehabilitation determined by the Australian Government. The assumptions made for uranium mining operations in the region are:

mining of uranium at Ranger is expected to cease in about 2012. This will be followed by milling until about 2020 and final rehabilitation expected to be completed by about 2026;

Nabarlek is decommissioned but has not reached a status where the NT Government will agree to issue a Revegetation Certificate to the mine operator. Assessment of the success of rehabilitation at Nabarlek is ongoing and may provide valuable data for consideration in the design and implementation of rehabilitation at Ranger;

Jabiluka will remain in a care and maintenance condition for some years. ERA, the project owner, has stated that further mining will not occur without the agreement of the traditional owners; and

grant of an exploration title at Koongarra is required under the terms of the Aboriginal Land Rights (Northern Territory) Act 1976 before the mining company can apply for a mining title. As such, any future activity at Koongarra is subject to the agreement of the traditional owners and the Northern Land Council.

This scenario is considered to be a reasonable basis on which to base plans for research and monitoring, but such plans may need to be amended if mining plans change in the future. ARRTC will ensure the research and monitoring strategy is flexible enough to accommodate any new knowledge needs.

The Australian Government has specified Primary and Secondary environmental objectives for mining at Ranger in the Ranger Environmental Requirements. Similar standards would be expected for any future mining development at Jabiluka or Koongarra.

Specifically, under the Ranger Environmental Requirements (ERs):

> The company must ensure that operations at Ranger are undertaken in such a way as to be consistent with the following primary environmental objectives:
>
> (a) maintain the attributes for which Kakadu National Park was inscribed on the World Heritage list;
>
> (b) maintain the ecosystem health of the wetlands listed under the Ramsar Convention on Wetlands (i.e. the wetlands within Stages I and II of Kakadu National Park);
>
> (c) protect the health of Aboriginals and other members of the regional community; and
>
> (d) maintain the natural biological diversity of aquatic and terrestrial ecosystems of the Alligator Rivers Region, including ecological processes.

With respect to rehabilitation at Ranger, the ERs state that:

> The company must rehabilitate the Ranger Project Area to establish an environment similar to the adjacent areas of Kakadu National Park such that, in the opinion of the Minister with the advice of the Supervising Scientist, the rehabilitated area could be incorporated into the Kakadu National Park.

The ERs go on to specify the major objectives of rehabilitation at Ranger as follows:

> (a) revegetation of the disturbed sites of the Ranger Project Area using local native plant species similar in density and abundance to those existing in adjacent areas of Kakadu National Park, to form an ecosystem the long term viability of which would not require a maintenance regime significantly different from that appropriate to adjacent areas of the park;
>
> (b) stable radiological conditions on areas impacted by mining so that, the health risk to members of the public, including traditional owners, is as low as reasonably achievable; members of the public do not receive a radiation dose which exceeds applicable limits recommended by the most recently published and relevant Australian standards, codes of practice, and guidelines; and there is a minimum of restrictions on the use of the area;
>
> (c) erosion characteristics which, as far as can reasonably be achieved, do not vary significantly from those of comparable landforms in surrounding undisturbed areas.

A secondary environmental objective applies to water quality and is linked to the primary ERs. This ER states:

> The company must not allow either surface or ground waters arising or discharging from the Ranger Project Area during its operation, or during or following rehabilitation, to compromise the achievement of the primary environmental objectives.

While there are many possible different structures that could be used to specify the Key Knowledge Needs, ARRTC has chosen to list the knowledge needs under the following headings:

- Ranger – current operations;
- Ranger – rehabilitation;
- Jabiluka;
- Nabarlek; and
- General Alligator Rivers Region.

'Key Knowledge Needs 2008–2010: Uranium mining in the Alligator Rivers Region' is based on and supersedes a predecessor document, 'Key Knowledge Needs 2004–2006: Uranium mining in the Alligator Rivers Region'. KKNs 2004–2006 remained the operative set during their review and the development of KKNs 2008–2010.

While some KKNs remain essentially unchanged, others contain revised elements or are new in their entirety. Care should be exercised if using KKN numbers alone as a reference because some continuing KKNs have changed numbers in the revised document.

1 Ranger – Current operations

1.1 Reassess existing threats

1.1.1 Surface water transport of radionuclides

Using existing data, assess the present and future risks of increased radiation doses to the Aboriginal population eating bush tucker potentially contaminated by the mining operations bearing in mind that the current Traditional Owners derive a significant proportion of their food from bush tucker.

1.1.2 Atmospheric transport of radionuclides

Using existing data and atmospheric transport models, review and summarise, within a risk framework, dose rates for members of the general public arising from operations at the Ranger mine.

1.2 Ongoing operational issues

1.2.1 Ecological risks via the surface water pathway

Off-site contamination during mine operation (and subsequent to decommissioning – refer KKN 2.6.1) should be placed in a risk-based context. A conceptual model of the introduction, movement and distribution of contaminants, and the resultant biotic exposure (human and non-human) has been developed, and the ecological risks (i.e. probability of occurrence x severity of consequence) of some of the contaminant/pathway sub-models have been estimated. This process should be completed for all the contaminant/pathway sub-models, noting, however, that the level of effort for each needs to be proportionate to the level of concern of the issue. It is critical that robust risk assessment methodologies are used, and that they explicitly incorporate uncertainty in both the assessment and subsequent decision making processes. Where ecological risk is significant, additional information may be required (e.g. mass-balance and concentration dynamics, consideration of possible interactive effects, field data). Further, knowledge gaps preventing reasonable estimation of potential risks (i.e. with unacceptable uncertainty) must be filled.

The Magela floodplain risk assessment framework developed to estimate and compare mining and non-mining impacts should be revisited periodically, and updated to the current risk profile. It should be revised in the event that either **(i)** the annual monitoring programme or other sources indicate that the inputs from mining have significantly increased relative to the situation in 2005, or **(ii)** an additional significant contaminant transport pathway from the minesite is identified, or **(iii)** there is a change in external stressors that could result in a significant increase in likelihood of impacts from the site.

1.2.2 Land irrigation

Investigations are required into the storage and transport of contaminants in the land irrigation areas particularly subsequent to decommissioning. Contaminants of interest/concern in addition to radionuclides are magnesium, sulfate and manganese. Results from these investigations should be sufficient to quantify the role of irrigation areas as part of satisfying KKN 1.2.1, and form the basis for risk management into the future.

1.2.3 Wetland filters

The key research issue associated with wetland filters in relation to ongoing operations is to determine whether their capacity to remove contaminants from the water column will continue to meet the needs of the water management system in order to ensure protection of the downstream environment. Aspects of contaminant removal capacity include (i) instantaneous rates of removal, (ii) temporal performance – including time to saturation, and (iii) behaviour under 'breakdown' conditions–including future stability after closure. Related to this is a reconciliation of the solute mass balance particularly for the Corridor Creek System (see KKN 1.2.5).

1.2.4 Ecotoxicology

Past laboratory studies provide a significant bank of knowledge regarding the toxicity of two of the major contaminants, uranium and magnesium, associated with uranium mining in the ARR. Further studies are scheduled to assess (i) the toxicity of manganese and, potentially, ammonia (in the event that permeate produced by process water treatment will contain potentially toxic ammonia concentrations), and (ii) the relationship between dissolved organic matter and uranium toxicity. This knowledge should continue to be synthesised and interpreted, within the existing risk assessment framework (refer KKN 1.2.1), as it comes to hand.

An additional issue that needs to be addressed is the direct and indirect effects on aquatic biota of sediment arising from the minesite. In the first instance, a conceptual model needs to be developed (building on the relevant components of the conceptual model developed under KKN 1.2.1) that describes the movement of sediment within the creek system, including the associated metal-sediment interactions and biological implications. Studies likely to arise from the outcomes of the conceptual model include:

- the effects of suspended sediment on aquatic biota;
- the relationship between suspended sediment and key metals, and how this affects their bioavailability and toxicity; and

- the effects of sediment-bound metals to benthic biota, including, initially, a review of existing information on uranium concentrations in sediments of waterbodies both on- and off the Ranger site, and uranium sediment toxicity to freshwater biota.

Whilst of relevance at present, the above issues will be of additional importance as Ranger progresses towards closure and rehabilitation (refer KKN 2.6.1). Finally, the need for studies to assess the toxicity of various mine waters (treated and untreated) in response to specific supervisory/regulatory or operational requirements is likely to continue.

1.2.5 Mass balances and annual load limits

With the expansion of land application areas and the increase in stockpile sheeting that has occurred in concert with the expansion of the footprints of the waste rock dumps and low grade ore stockpiles, it is becoming increasingly important to develop a solute mass balance for the site – such that the behaviour of major solute source terms and the spatial and temporal contribution of these sources to water quality in Magela Creek can be clearly understood. Validated grab sample and continuous data records are needed to construct a high reliability solute mass balance model.

Related to mass balance is the issue of specifying allowable annual load limits from the site – as part of the site's regulatory requirements. The technical basis for these load limits needs to be reviewed since they were originally developed decades ago. There has since been significantly increased knowledge of the environmental geochemistry of the site, a quantum increase in knowledge about ecotoxicological sensitivity of the aquatic systems and updated data on the diet profile of traditional owners.

1.3 Monitoring

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

Routine and project-based chemical, biological, radiological and sediment monitoring should continue, together with associated research of an investigative nature or necessary to refine existing, or develop new (promising) techniques and models. A review of current water quality objectives for Ranger should be conducted to determine if they are adequate for future water management options for the whole-of-site, including the closure and rehabilitation phase (KKN 2.2.1 and KKN 2.2.2).

ARRTC supports the design and implementation of a risk-based radiological monitoring programme based on a robust statistical analysis of the data collected over the life of Ranger necessary to provide assurance for Aboriginal people who source food items from the Magela Creek system downstream of Ranger.

2 Ranger – Rehabilitation

2.1 Reference state and baseline data

2.1.1 Defining the reference state and baseline data

There is a requirement to define the baseline data/reference state that existed at the Ranger site prior to development. This will inform the process of the development of closure criteria which is compatible with the Environmental Requirements. The knowledge need is to develop and perform analysis to generate agreed reference data that cover the range of pre-mining and operational periods.

2.2 Landform

2.2.1 Landform design

An initial design is required for the proposed final landform. This would be based upon the optimum mine plan from the operational point of view and it would take into account the broad closure criteria, engineering considerations and the specific criteria developed for guidance in the design of the landform. This initial landform would need to be optimised using the information obtained in detailed water quality, geomorphic, hydrological and radiological programmes listed below.

Current and trial landforms at Ranger and at other sites such as Nabarlek should be used to test the various models and predictions for water quality, geomorphic behaviour and radiological characteristics at Ranger. The detailed design for the final landform at Ranger should be determined taking into account the results of the above research programmes on surface and ground water, geomorphic modelling and radiological characteristics.

2.2.2 Development and agreement of closure criteria from the landform perspective

Closure criteria from the landform perspective need to be established at both the broad scale and the specific. At the broad scale, agreement is needed, particularly with the Traditional Owners and within the context of the objectives for rehabilitation incorporated within the ERs, on the general strategy to be adopted in constructing the final landform. These considerations would include issues such as maximum height of the landform, the maximum slope gradient (from the aesthetic perspective), and the presence or absence of lakes or open water. At the specific scale, some criteria could usefully be developed as guidance for the initial landform design such as slope length and angle (from the erosion perspective), the minimum cover required over low grade ore, and the minimum distance of low grade ore from batter slopes. Specific criteria are needed that will be used to assess the success of landform construction. These would include, for example, maximum radon exhalation and gamma dose rates, maximum sediment delivery rates, maximum constituent concentration rates in runoff and maximum settling rates over tailings repositories.

2.2.3 Water quality in seepage and runoff from the final landform

Existing water quality monitoring and research data on surface runoff and subsurface flow need to be analysed to develop models for the quality of water, and its time dependence, that would enter major drainage lines from the initial landform design. Options for adjusting the design to minimise solute concentrations and loads leaving the landform need to be assessed.

There is a need to develop and analyse conceptual models of mine-related turbidity and salinity impacts following closure. These models could be analysed in a variety of ways; as a precursor to the development of a quantitative model of potential turbidity and salinity impacts offsite cause by surface and subsurface water flow off the rehabilitated minesite. This analysis should explicitly acknowledge knowledge uncertainty (e.g. plausible alternative conceptual models) and variability (e.g. potential for Mg/Ca ratio variations in water flowing off the site) and explore the potential ramifications for the off-site impacts. (see also KKN 2.6.1)

2.2.4 Geomorphic behaviour and evolution of the landscape

The existing data set used in determination of the key parameters for geomorphological modelling of the proposed final landform should be reviewed after consideration of the near surface characteristics of the initial proposed landform. Further measurements of erosion characteristics should be carried out if considered necessary. The current site-specific landform evolution models should be applied to the initial proposed landform to develop predictions for long term erosion rates, incision and gullying rates, and sediment delivery rates to the surrounding catchments. Options for adjusting the design to minimise erosion of the landform need to be assessed. In addition, an assessment is needed of the geomorphic stability of the Ranger minesite with respect to the erosional effects of extreme events.

2.2.5 Radiological characteristics of the final landform

The characteristics of the final landform from the radiological exposure perspective need to be determined and methods need to be developed to minimise radiation exposure to ensure that restrictions on access to the land are minimised. Radon exhalation rates, gamma dose rates and radionuclide concentrations in dust need to be determined and models developed for both near-field and far-field exposure.

The use of potential analogue sites for establishing pre-mining radiological conditions at Ranger should be further investigated to provide information on parameters such as pre-mining gamma dose rates, radon exhalation rates, and levels of radioactivity in dust. This information is needed to enable estimates to be made of the likely change in radiation exposure when accessing the rehabilitated site compared to pre-mining conditions.

2.3 Groundwater dispersion

2.3.1 Containment of tailings and other mine wastes

The primary method for protection of the environment from dispersion of contaminants from tailings and other wastes will be containment. For this purpose, investigations are required on the hydrogeological integrity of the pits, the long-term geotechnical properties of tailings and waste rock fill in mine voids, tailings deposition and transfer (including TD to Pit #3) methods, geochemical and geotechnical assessment of potential barrier materials, and strategies and technologies to access and 'seal' the surface of the tailings mass, drain and dispose of tailings porewater, backfill and cap the remaining pit void.

2.3.2 Geochemical characterisation of source terms

Investigations are needed to characterise the source term for transport of contaminants from the tailings mass in groundwater. These will include determination of the permeability of the tailings and its variation through the tailings mass, strategies and technologies to enhance settled density and accelerate consolidation of tailings, and porewater concentrations of key constituents.

There is a specific need to address the existence of groundwater mounds under the tailings dam and waste rock stockpiles. Models are needed to predict the behaviour of groundwater and solute transport in the vicinity of these mounds and options developed for their remediation to ensure that on-site revegetation can be achieved and that off-site solute transport from the mounds will meet environmental protection objectives. Assessment is also needed of the effectiveness (cost and environmental significance) of paste and cementation technologies for increasing tailings density and reducing the solubility of chemical constituents in tailings.

2.3.3 Aquifer characterisation and whole-of-site model

The aquifers surrounding the tailings repositories (Pits 1 & 3) need to be characterised to enable modelling of the dispersion of contaminants from the repositories. This will involve geophysics surveys, geotechnical drilling and groundwater monitoring and investigations on the interactions between the deep and shallow aquifers.

2.3.4 Hydrological/hydrogeochemical modelling

Predictive hydrological/hydrogeological models need to be developed, tested and applied to assess the dispersion of contaminants from the tailings repositories over a period of 10 000 years. These models will be used to assess whether all relevant and appropriate factors have been considered in designing and constructing an in-pit tailings containment system that will prevent environmental detriment in the long term.

2.4 Water treatment

2.4.1 Active treatment technologies for specific mine waters

Substantial volumes of process water retained at Ranger in the tailings dam and Pit 1 must be disposed of by a combination of water treatment and evaporation during the mining and milling phases of the operation and during the rehabilitation phase. Research priorities include treatment technologies and enhanced evaporation technologies that can be implemented for very high salinity process water. A priority should be evaluation of the potential impact of treatment sludge and brine streams on long term tailings chemistry in the context of closure planning and potential post closure impacts on water quality.

2.4.2 Passive treatment of waters from the rehabilitated landform

Sentinel wetlands may form part of the final landform at Ranger. Research on wetland filters during the operational phase of mining will provide information relevant to this issue. Research is needed to establish the effect of wet-dry seasonal cycling on contaminant retention and release, since this aspect will influence design criteria and whether such wetlands should be maintained as ephemeral or perennial waterbodies There is also the need to assess the long-term behaviour of the physical and biotic components of the wetlands, their ecological health, and the extent of contaminant accumulation (both metals and radionuclides) in the context of potential human exposure routes.

2.5 Ecosystem establishment

2.5.1 Development and agreement of closure criteria from ecosystem establishment perspective

Closure criteria need to be established for a range of ecosystem components including surface water quality, flora and fauna. The environmental requirements provide some guidance but characterisation of the analogue ecosystems will be an important step in the process. Consultation on closure criteria with the traditional owners has commenced and it is important that this process continues as more definitive criteria are developed.

2.5.2 Characterisation of terrestrial and aquatic ecosystem types at analogue sites

Identification and characterisation of analogue ecosystems (target habitats) can assist in defining the rehabilitation objective and developing robust, measurable and ecologically-based closure criteria. The concept of using analogue ecosystems for this purpose has been accepted by ARRTC and the traditional owners. Substantial work has been undertaken on the Georgetown terrestrial analogue ecosystem while there is also a large body of information available on aquatic analogues, including streams and billabongs. Future work on the terrestrial analogue needs to address water and nutrient dynamics, while work on the aquatic analogue will include the development of strategies for restoration of degraded or removed natural waterbodies, Coonjimba and Djalkmara, on site.

2.5.3 Establishment and sustainability of ecosystems on mine landform

Research on how the landform, terrestrial and aquatic vegetation, fauna, fauna habitat, and surface hydrology pathways will be reconstructed to address the Environmental Requirements for rehabilitation of the disturbed areas at Ranger is essential. Trial rehabilitation research sites should be established that demonstrate an ability by the mine operator to be able to reconstruct terrestrial and aquatic ecosystems, even if this is at a relatively small scale. Rehabilitation establishment issues that need to be addressed include species selection; seed collection, germination and storage; direct seeding techniques; propagation of species for planting; fertiliser strategies and weathering properties of waste rock. Rehabilitation management issues requiring investigation include the stabilisation of the land surface to erosion by establishment of vegetation, return of fauna; the exclusion of weeds; fire management and the re-establishment of nutrient cycles. The sustainable establishment and efficiency of constructed wetland filters, reinstated waterbodies (e.g. Djalkmara Billabong) and reconstructed waterways also needs to be considered (see KKN 2.3.2).

2.5.4 Radiation exposure pathways associated with ecosystem re-establishment

Radionuclide uptake by terrestrial plants and animals on the rehabilitated ecosystem may have a profound influence on the potential utilisation of the land by the traditional owners. Significant work has been completed on aquatic pathways, particularly the role of freshwater mussels, and this now forms part of the annual monitoring programme. The focus is now on the terrestrial pathways and deriving concentration factors for bushtucker such as wallabies, fruits and yams. A project investigating the contemporary diet of traditional owners has commenced and needs to be completed. Models need to be developed that allow exposure pathways to be ranked for currently proposed and future identified land uses, so that identified potentially significant impacts via these pathways can be limited through appropriate design of the rehabilitation process.

2.6 Monitoring

2.6.1 Monitoring of the rehabilitated landform

A new management and monitoring regime for the rehabilitated Ranger landform needs to be developed and implemented. It needs to address all relevant aspects of the rehabilitated landform including ground and surface water quality, radiological issues, erosion, flora, fauna, weeds, and fire. The monitoring regime should address the key issues identified by the ecological risk assessment of the rehabilitation phase (KKN 2.7.1).

2.6.2 Off-site monitoring during and following rehabilitation

Building upon the programme developed and implemented for the operational phase of mining, a monitoring regime is also required to assess rehabilitation success with respect to protection of potentially impacted ecosystems and environmental values. This programme should address the dispersion of contaminants by surface water, ground water and via the atmosphere. The monitoring regime should address the key issues identified by the ecological risk assessment of the rehabilitation phase (KKN 2.7.1).

2.7 Risk assessment

2.7.1 Ecological risk assessments of the rehabilitation and post rehabilitation phases

In order to place potentially adverse on-site and off-site issues at Ranger during the rehabilitation phase within a risk management context, it is critical that a robust risk assessment framework be developed with stakeholders. The greatest risk is likely to occur in the transition to the rehabilitation phase, when active operational environmental management systems are being progressively replaced by passive management systems. A conceptual model of transport/exposure pathways should be developed for rehabilitation and post rehabilitation regimes and the model should recognise the potential that some environmental stressors from the minesite could affect the park and vice versa. Implicit in this process should be consideration of the effects of extreme events and climate change.

Conceptual modelling should be followed by a screening process to identify and prioritise key risks for further qualitative and/or quantitative assessments. The conceptual model should be linked to closure criteria and post-rehabilitation monitoring programmes, and be continually tested and improved. Where appropriate, risk assessments should be incorporated into decision making processes for the closure plan. Outputs and all uncertainties from this risk assessment process should be effectively communicated to stakeholders.

2.8 Stewardship

The concept of Stewardship (including ownership and caring for the land) is somewhat broader and applies to all phases of, in this case, uranium mining. In this context it is considered to be the post closure phase of management of the site, i.e. after relinquishment of the lease. If the rehabilitation phase is successful in meeting all objectives then this stewardship will effectively comprise an appropriate level of ongoing monitoring to confirm this. Should divergence from acceptable environmental outcomes be detected then some form of intervention is likely to be required. The nature, responsibility for, and duration of, the monitoring and any necessary intervention work remains to be determined.

3 Jabiluka

3.1 Monitoring

3.1.1 Monitoring during the care and maintenance phase

A monitoring regime for Jabiluka during the care and maintenance phase needs to be implemented and regularly reviewed. The monitoring programme (addressing chemical, biological, sedimentalogical and radiological issues) should be commensurate with the environmental risks posed by the site, but should also serve as a component of any programme to collect baseline data required before development such as meteorological and sediment load data.

3.2 Research

3.2.1 Research required prior to any development

A review of knowledge needs is required to assess minimum requirements in advance of any development. This review would include radiological data, the groundwater regime (permeabilities, aquifer connectivity etc), hydrometeorological data, waste rock erosion, assess site-specific ecotoxicology for uranium, additional baseline for flora and fauna surveys.

4 Nabarlek

4.1 Success of revegetation

4.1.1 Revegetation assessment

Several assessments of the revegetation at Nabarlek have been undertaken; the most recent being completed by ***eriss***. There is now general agreement that the rehabilitated areas require further work. Revised closure criteria are currently being developed through the minesite technical committee and these should be reviewed by relevant stakeholders, including ARRTC. The required works should then be completed on site with further monitoring leading to the relinquishment of the lease.

4.1.2 Development of revegetation monitoring method

A methodology and monitoring regime for the assessment of revegetation success at Nabarlek needs to be developed and implemented. Currently, resource intensive detailed vegetation and soil characterisation assessments along transects located randomly within characteristic areas of the rehabilitated landform are being undertaken. Whilst statistically valid, these assessments cover only a very small proportion of the site. Remote sensing (satellite) data are also being collected and the efficacy of remote sensing techniques for vegetation assessment in comparison to ground survey methods should continue. The outcomes of this research will be very relevant to Ranger.

4.2 Assessment of radiological, chemical and geomorphic success of rehabilitation

4.2.1 Overall assessment of rehabilitation success at Nabarlek

The current programme on erosion, surface water chemistry, groundwater chemistry and radiological issues should be continued to the extent required to carry out an overall assessment of the success of rehabilitation at Nabarlek. In particular, all significant radiological exposure pathways should be identified and a comprehensive radiation dose model developed. Additional monitoring of ground water plumes is required to allow assessment of potential future groundwater surface water interaction and possible environmental effects.

5 General Alligator Rivers Region

5.1 Landscape scale analysis of impact

5.1.1 Develop a landscape-scale ecological risk assessment framework for the Magela catchment that incorporates, and places into context, uranium mining activities and relevant regional landscape processes and threats, and that builds on previous work for the Magela floodplain

Ecological risks associated with uranium mining activities in the ARR, such as current operations (Ranger) and rehabilitation (Nabarlek, Jabiluka, future Ranger, South Alligator Valley), should be assessed within a landscape analysis framework to provide context in relation to more diffuse threats associated with large-scale ecological disturbances, such as invasive species, unmanaged fire, cyclones and climate change. Most key landscape processes occur at regional scales, however the focus will be on the Magela catchment encompassing the RPA. A conceptual model should first be developed to capture links and interactions between multiple risks and assets at multiple scales within the Magela catchment, with risks associated with Ranger mining activities made explicit. The spatially explicit Relative Risk Model will be used to prioritise multiple risks for further qualitative and/or quantitative assessments. The conceptual model and risk assessment framework should be continually tested and improved as part of Best Practice. Where appropriate, risk assessments should be incorporated into decision making processes using advanced risk assessment frameworks such as Bayesian Networks, and all uncertainties made explicit. This risk assessment process should integrate outputs from KKN 1.2.1 (risks from the surface water pathway – Ranger current operations) and the new KKN 2.6.1 (risks associated with rehabilitation) to provide a landscape-scale context for the rehabilitation of Ranger into Kakadu National Park, and should be communicated to stakeholders.

5.2 South Alligator River valley rehabilitation

5.2.1 Assessment of past mining and milling sites in the South Alligator River valley

SSD conducts regular assessments of the status of minesites in the SAR valley, provides advice to Parks Australia on technical issues associated with its rehabilitation programme and conducts a low level radiological monitoring programme. This work should continue.

5.3 Develop monitoring programme related to West Arnhem Land exploration activities

5.3.1 Baseline studies for biological assessment in West Arnhem Land

ARRTC believes there is a need to determine a baseline for (a) rare, threatened and endemic biota and (b) indicator species or groups such as macroinvertebrates in areas where advanced exploration or proposed mining projects are identified and in line with the current approvals process under the Aboriginal Land Rights Act.

5.4 Koongarra

5.4.1 Baseline monitoring programme for Koongarra

In line with the current approvals process under the Aboriginal Land Rights Act, a low level monitoring programme should be developed for Koongarra to provide baseline data in advance of any possible future development at the site. Data from this programme could also have some relevance as a control system for comparison to Ranger, Jabiluka and Nabarlek.