Part 2: Ranger – Rehabilitation

Define the geomorphic characteristics of Gulungul Creek catchment

D Moliere, K Turner, K Evans & M Saynor

Background

This project has three aims: (1) to develop reliable impact assessment methods for quantifying the loads of fine suspended sediment transported in Gulungul Creek during rainfall/runoff events; (2) to characterise the channel stability of the creek; and (3) to characterise the bedload movement in the creek upstream and downstream of Ranger. This project was last reported at ARRTC 24 and in Moliere et al (2009). Studies such as these in Gulungul Creek are assuming greater importance for baseline characterisation given the current (tailings dam lift) and proposed expansion of mine infrastructure by ERA into the catchment.

Progress

Suspended sediment loads for event data collected between 2003 and 2008 are shown in Moliere and Evans (2008).

Data for the 2008–09 wet season (rainfall, discharge, turbidity and electrical conductivity (EC)) were collected at the upstream (GCUS) and downstream (GCDS) monitoring stations in Gulungul Creek (see Map 2 for locations). Discharge and rainfall data for the upstream and downstream sites are shown in Figure 1 and Figure 2, respectively, and the EC and turbidity data for both sites are shown in Figure 3.

Rainfall and runoff data were also collected at the mid Gulungul Gauging Station (GS210012), which is located between GCUS and GCDS (Map2) . Bed material was collected from stream cross-sections as they were being surveyed during the 2008 dry season. Analysis and reporting of the bed material data have been delayed due to staff changes and competing priorities.

During the 2009 dry season, work was carried out at each of the Gulungul Creek continuous monitoring stations to improve the ease of use and security of the installed instrumentation. This work included installation of cable connectors and conduits for each of the sensors so that they can easily be removed for maintenance or calibration and to reduce possible damage due to passing debris. New sensor mounts were also constructed to provide a more secure location for mounting the various sensors at each site.

A thorough assessment has also been made of the performance of the backup EC sensors currently used at the Gulungul Creek sites. All of SSD’s continuous monitoring sites in Gulungul and Magela Creeks have backup sensors in place to provide redundancy for these critical data. In Magela Creek the backup for the upstream and downstream sites is provided by a second multiparameter sonde. However, in Gulungul Creek the backup data are provided by individual probes connected to a central datalogger. This situation is the result of the historical evolution of the continuous monitoring systems in this creek.

It was found that the backup sensors have relatively low measurement repeatability, and measurement error of ±10% for EC values ≤ 500 μS/cm. The sensors are also only capable of outputting data at 1 μS/cm resolution, which is low compared with other EC sensors utilised by SSD which are capable of outputting data at 0.01 μS/cm resolution. These backup EC sensors have not been used as a primary source of EC data since 2007, when minisondes were installed as the primary sensors at GCUS and GCDS. The current backup EC sensors will be phased out of SSD's continuous monitoring program over the next two years, and replaced with probes with higher performance characteristics.

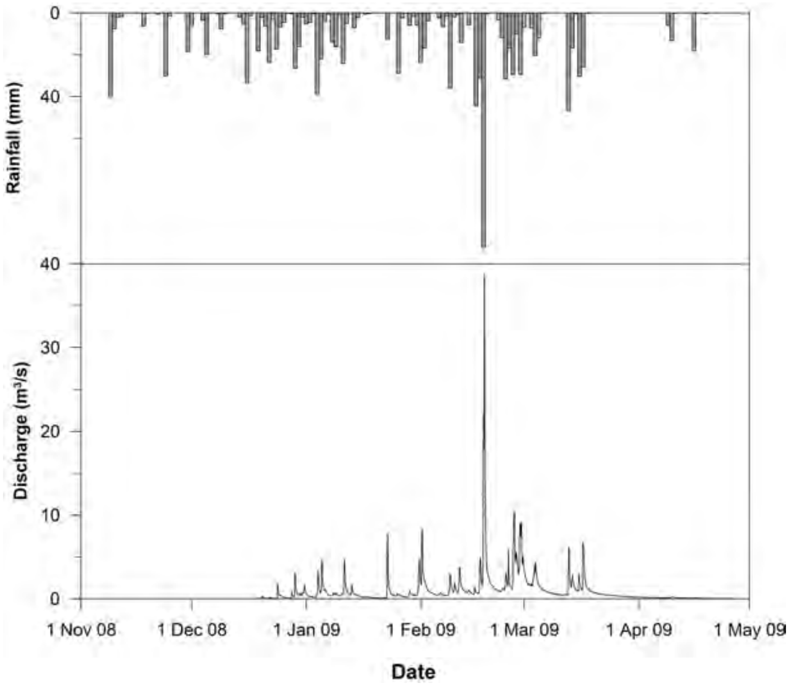

Figure 1 Discharge and rainfall data for GCUS during the 2008–09 wet season

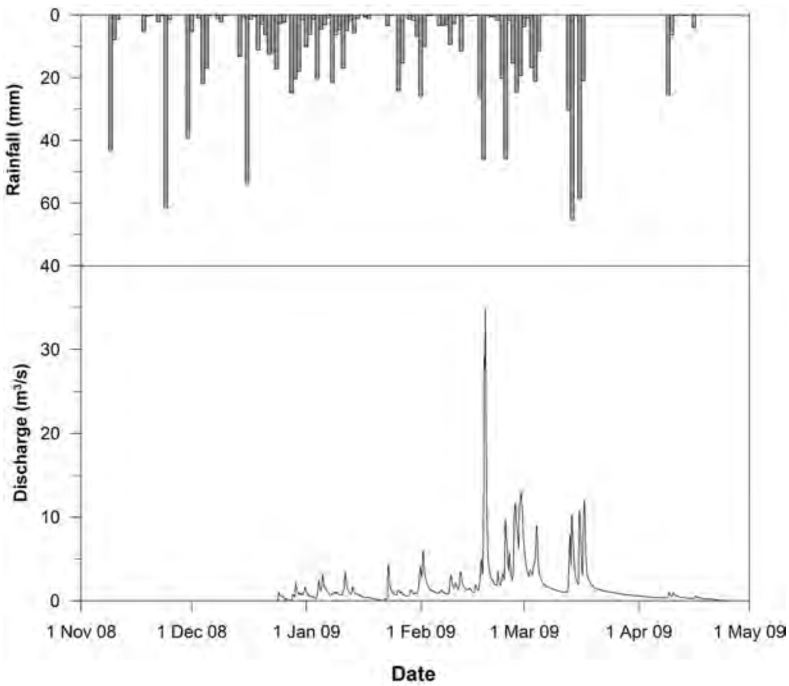

Figure 2 Discharge and rainfall data for GCDS during the 2008–09 wet season

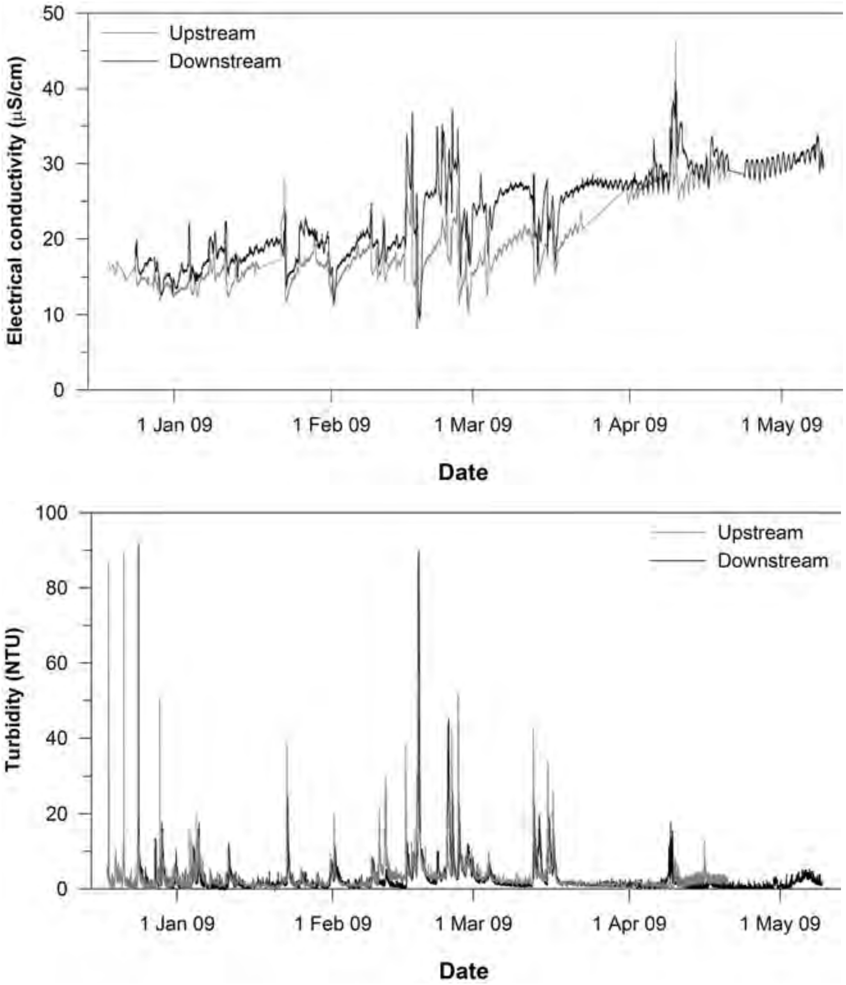

Figure 3 Electrical conductivity and turbidity data for GCUS and GCDS during the 2008-09 wet season

Future work

Rainfall, discharge, turbidity and EC data will be collected for the 2009–10 wet season. cross sections are surveyed annually during the dry season to determine channel changes within the creek. Additionally, continuous EC data will be collected at GS210012 to be able to more clearly identify any inputs contained in runoff from the waste-rock-clad walls of the Ranger tailings dam. Analyses and reporting of these data will progress as new staff are trained in the process.

Reference

Moliere DR, Evans KG & Saynor MJ 2009. Development of catchment geomorphic characteristics of Gulungul Creek – monitoring results. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 166–169.

Moliere DR & Evans KG 2010. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research*, DOI: 10.1111/j.1745-5871.2010.00641.x

Revegetation trial and demonstration landform – erosion and chemistry studies

M Saynor, K Turner, R Houghton & K Evans

Introduction

A trial landform was constructed during late 2008 and early 2009 by Energy Resources of Australia Ltd (ERA) adjacent to the north-western wall of the tailings storage facility (TSF) at Ranger mine. The trial landform will be used to test landform design and revegetation strategies to be used once mining and milling have ceased. The trial landform is an extension of the topography of the TSF, projecting out from the TSF wall in a north-west direction (Figure 1). It is a rectangular shape of approximately 200 m x 400 m (8 ha) in footprint area.

Figure 1 Location of the elevated trial landform (bottom right of photograph) at Ranger mine

The landform was designed to test the performance (in terms of geomorphic stability and suitability for plant growth medium) of two types of potential final cover layers:

1 Waste rock alone

2 Waste rock blended with approximately 30% v/v fine-grained weathered horizon material (laterite)

For cover layer 2 from above, two thicknesses (2 m and 5 m) of the mixed laterite and waste rock cover type are being evaluated. It is anticipated that whilst the different thicknesses are not likely to exhibit any material difference in erosion properties, they may differ in their long term ability to sustain mature, deeply rooted vegetation. However, it is the erosion potential and emanation of solutes that are the focus of the work described here.

The landform is divided into six plots (Figure 2), each of which is being used to test different vegetation planting methods and substrate types, as follows:

1 Tube stock planted in waste rock mixed with laterite material to a depth of 2 m;

2 Tube stock planted in waste rock mixed with laterite material to a depth of 5 m;

3 Direct seeding in waste rock mixed with laterite to a depth of 2 m;

4 Direct seeding in waste rock mixed with laterite material to a depth of 5 m;

5 Direct seeding in waste rock material;

6 Tube stock planted in waste rock material.

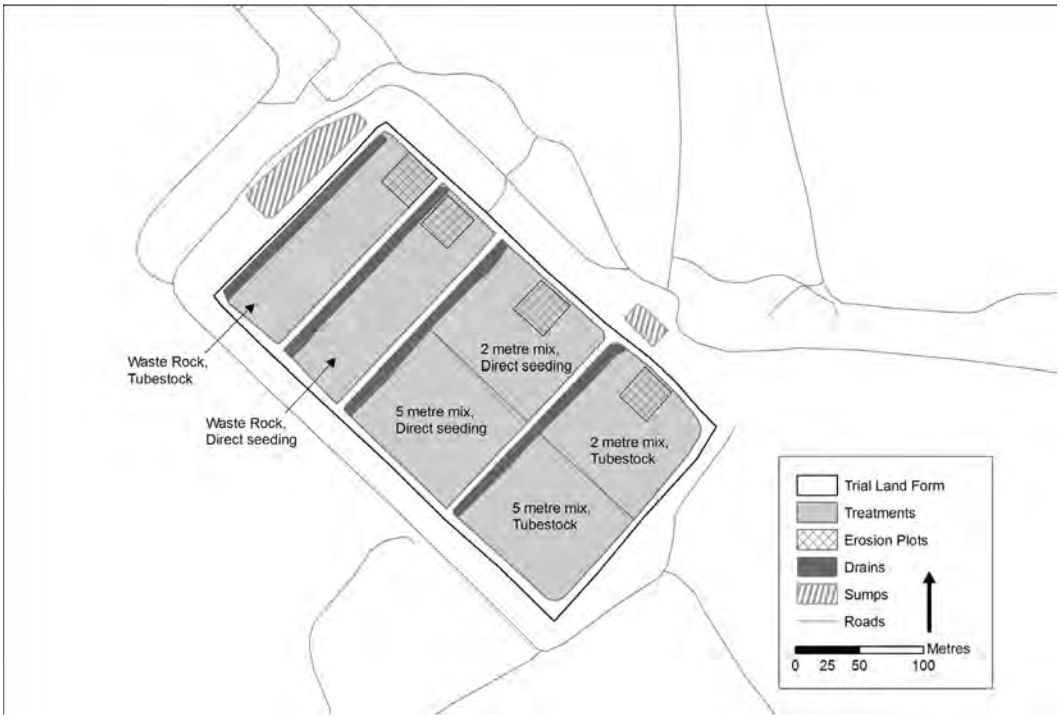

Figure 2 Layout of the plots on the trial landform

Runoff, sediment concentration and water quality (turbidity and electrical conductivity) will be measured in ensuing wet seasons from four 900 m^2 erosion plots that were constructed on the landform between February and July 2009. The plot locations are shown in Figure 2. These locations allow comparison of erosion products and water quality in runoff from a mixed waste rock and laterite substrate vegetated by direct seeding and tube stock and a waste rock substrate vegetated by direct seeding and tube stock.

Plot construction

Each of the 30 m x 30 m erosion plots are physically isolated from runoff from the rest of the trial landform surface area by damp-proof course borders held in place by concrete mortar (Figure 3). Half-section 300 mm diameter U-PVC stormwater pipes (Figure 3) have been placed at the down-slope ends of the plots to capture runoff and channel it through rectangular, broad-crested (RBC) flumes (Figure 4) where rainfall event discharge will be measured. Transported bed sediment will be trapped in a reservoir constructed upstream of the inlet to the flume (Figure 4). Plot construction is described in detail in Saynor et al (2009).

Each flume will be instrumented with a pressure transducer and shaft encoder to measure stage height, a turbidity probe, an automatic water sampler and a data logger. Continuous electrical conductivity data will be measured in the half-section stormwater pipes and in the flumes to provide a measure of the concentrations of dissolved salts in the runoff. A rain gauge will be installed near each flume to record the rainfall and rainfall intensity at each of the plots. All the he data will be downloaded once a day via mobile phone telemetry and then stored in the hydrological database, Hydstra. Decisions on how often the plots will be visited to clear bedload and to collect water samples will be made after the plots are in place and there has been an opportunity to observe erosion rates and discharge relative to size of rainfall event. The construction and instrumentation of the erosion plots was completed early in Q4 2009, in advance of the 2009–10 wet season.

Figure 3 Plastic half pipe trough and boundary

Figure 4 Reservoir and flume at the outlet of the erosion plot

Future work

For the 2009–10 wet season, the resources of the HGP program, with assistance from EWLS, will focus on ensuring the monitoring systems are functioning correctly and on successfully obtaining the first year of data from the plots and reporting these data.

Wet season suspended and bed sediment loads, EC (continuous) and water chemistry data (grab samples) will continue to be collected for several years to track the evolution of the landform surface and any mitigative effect of increasing coverage by vegetation.

Solute generation will be investigated using the continuous electrical conductivity record in conjunction with the major ion composition measured in collected water samples. It is hoped that significant relationships will be found between EC and major-ion solute composition such that continuous EC will be able to be used to infer solute export in the 2009-10 and subsequent wet seasons.

References

Saynor MJ, Evans KG & Lu P 2009. Erosion studies of the Ranger revegetation trial plot area. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 125–195.

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

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

Introduction

The majority of this project, as it relates to Ranger landform evolution simulations, was reported at the 22nd ARRTC meeting in October 2008. It was also reported in Evans et al (2009).

Progress/results

Little further work has been undertaken due to the temporary reassignment of staff to other duties. Collaborative research with Associate Professor Greg Hancock evaluating the Siberia and CAESAR models resulted in a paper presented at the 14th Australasian and Remote Sensing and Photogrammetry Conference in Darwin in October 2008 and now accepted for publication (Hancock et al in press). A paper titled 'Assessing the impact of extreme rainfall events on the geomorphic stability of a conceptual rehabilitated landform in the Northern Territory of Australia' was presented at the Fourth International Conference on Mine Closure in Perth in September 2009.

Steps for completion

The following steps are required to complete this project:

1 Develop model capability to incorporate spatial variability in surface material types and vegetation distribution. Little further simulation work can be done until this step is completed.

2 Systematically test the effects of Digital Elevation Model (DEM) resolution on model outputs.

3 Use the results obtained from the Ranger Trial Landform to calibrate/validate landform evolution model performance.

4 Define the duration and intensity characteristics for extreme rainfall events and apply these to simulations to conduct a risk assessment of landform stability.

5 Compare long-term erosion rates measured on natural undisturbed sites with CAESAR and Siberia simulations for those sites.

6 Assess and evaluate the importance of vegetation on landform stability.

While collaborative research programs have been developed between SSD and Drs Coulthard and Hancock, competing projects (spatial data management project, implementation of photo database Ranger landform trial) took priority in 2008–09. In 2009–10, Dr Coulthard is planning

1 School of Environmental & Life Sciences, The University of Newcastle, Callaghan, NSW 2308, Australia
2 Department of geography, University of Hull, Hull, HU6 7RX, UK

two visits to ***eriss*** to work on updating the CAESAR model to enable it to simulate variation in surface treatments on a landform ie rock mulch, vegetation and ripping; presently, CAESAR can only simulate erosion of landforms with uniform surface.

Some initial work has commenced on classifying extreme rainfall events using methods taken from the literature. The first method being assessed is that of Casas et al (2004). They used rainfall Intensity-Duration-Frequency curves (IDF curves) to analyse rainfall duration data and to select high intensity/low frequency return interval events based on the IDF curves. Cluster analysis was then used to classify the rainfall events into four groups: 1. very short duration events representing localised rainfall with a clear seasonal influence; 2. mesoscale duration where intense rainfall rate systems may develop; 3. synoptic rainfall events with intensities exceeding 5-year return period level for 12 to 24-hour time intervals; and 4. mid to large-scale meteorological processes showing high rates for large ranges of durations (20 min to 24 hr). Applying the Casas et al (2004) approach to the local (NT) situation, firstly, the Darwin continuous rainfall record (54 years) has been analysed in Hydstra and event duration data extracted, as well as high intensity/low frequency return interval events for each duration based on Darwin IDF curves. The next step will be to classify the events using cluster analysis. The most intense events for each class will be applied to a proposed Ranger rehabilitated landform using CAESAR to simulate erosion rates for each group of rainfall events to assess likely impacts which could occur during 54-year period for which continuous rainfall has been collected in Darwin. Depending on the results arising from use of the Darwin record, the much shorter Jabiru rainfall record may also be assessed.

Other methods of determining the magnitude of extreme events will also be investigated, including slackwater deposit studies (Erskine & Saynor 2000).

References

Casas, MC, Codina, B, Redaño, A & Lorente, J 2004. A methodology to classify extreme rainfall events in the western Mediterranean area. *Theoretical Applied Climatology* 77, 139–150.

Erskine WD & Saynor MJ 2000. *Assessment of the off-site geomorphic impacts of uranium mining on Magela Creek, Northern Territory, Australia*. Supervising Scientist Report 156, Supervising Scientist, Darwin NT.

Evans KG, Hancock GR, Lowry JBC & Coulthard TJ 2009. Assess the impact of extreme rainfall events on Ranger rehabilitated landform geomorphic stability using the CAESAR landscape evolution model. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 100–105.

Hancock GR, Lowry JBC, Coulthard TJ, Evans KG & Moliere DR 2009. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models. *Earth Surface Processes and Landforms* (in press)

Hancock GR, Lowry JBC, Coulthard TJ & Evans KG 2008. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models. In *14ARSPC: Proceedings of the 14th Australasian Remote Sensing and Photogrammetry Conference*, Darwin NT, 30 September – 2 October 2008. USB2.0

Lowry JBC, Evans KG, Coulthard TJ, Hancock GR & Moliere DR 2009. Assessing the impact of extreme rainfall events on the geomorphic stability of a conceptual rehabilitated landform in the Northern Territory of Australia. In *Proceedings of the Fourth International Conference on Mine Closure*, eds M Fourie & M Tibbett, 9–11 September 2009, Perth, Australian Centre for Geomechanics, 203–212.

Validation of the SIBERIA model, its erosion parameters and erosion rate predictions

G Hancock[1] & K Evans

Introduction

Gully incision and development are important factors in soil erosion, catchment development and subsequent increased sedimentation and water quality problems in many catchments. Understanding the spatial and temporal process of gully initiation and development is of critical importance in understanding the long-term stability of post-mining landforms, which often have steeper slopes than the surrounding undisturbed landscape, are devoid of, or have limited, vegetation cover and may export increased sediment loads to receiving streams through erosion as the landform equilibrates to it surrounds. Therefore, the ability to predict where initiation of gullies may start and how they evolve is very important for environmental management particularly mine sites. The insights gained will allow upfront engineering measures to be employed to minimise gully initiation and development.

This study examines erosion feature characteristics at Tin Camp Creek in the Northern Territory, Australia (Figure 1) and builds on a previous work that has been carried out at this site (Hancock & Evans 2006). The catchment has a geology very similar to the Energy Resources Australia (ERA) Ranger uranium mine and is thought to be an analogue for the long-term rehabilitated post-mining landscape. There appears to be a lack of studies examining gullies in undisturbed environments and long-term studies of the evolution of erosion features are scarce. Little is known about rates of headward movement and changes in width together with scour and fill.

The study catchment has largely uniform geology, soils and vegetation. Moreover, because of its small size, the climate conditions can be assumed to be uniform. The aims of this study are to investigate gullying processes in the catchment by examining trends in gully head drainage area and slope characteristics, as well as gully depth, width and length. The link with hillslope erosion is also examined.

Progress and results to date

In August 2002, gullies in the Tin Camp Creek study catchment were initially mapped after 39 representative erosion features within the entire catchment were selected for long-term monitoring. Thirty-four of these were gully heads and 5 were scour holes located in the channel downstream of a channel head. The selected erosion features covered a range of sizes from rills up to large gullies (Figure 2) (Hancock & Evans 2006). Erosion pins were also installed to measure hillslope rates of erosion and deposition adjacent to the monitored gullies.

The Tin Camp Creek gullies have been measured only over a limited 6 year period, yet of course are the product of many years of erosion. The occurrence of well-defined incisions in the catchment (and surrounding catchments) indicate that gullying is an important process in the area. Further, the fact that gullying is present in the absence of European land use practices or other anthropogenic land disturbance in the area is an important finding.

[1] School of Environmental & Life Sciences, The University of Newcastle, Callaghan, NSW 2308, Australia

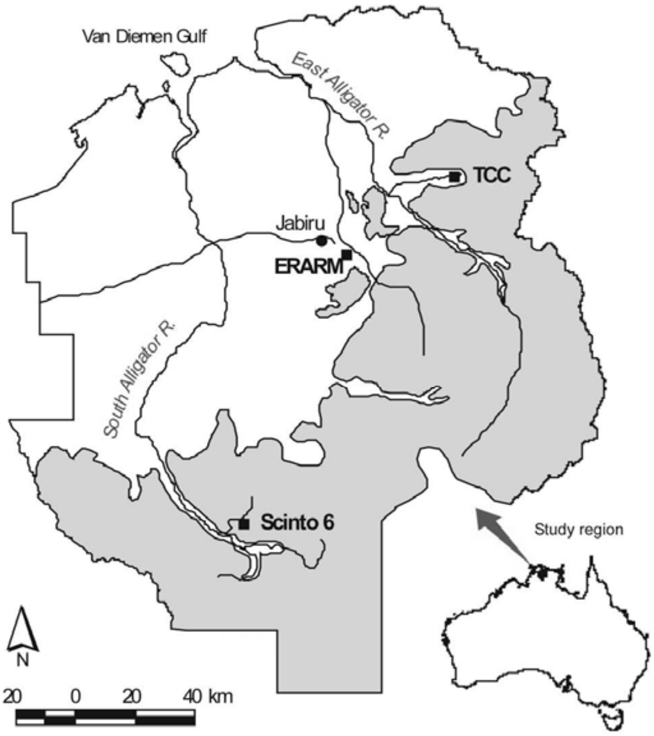

Figure 1 Location of study site (TCC – Tin Camp Creek)

Figure 2 A medium (left) and large (right) size gully within the Tin Camp Creek catchment

No statistical correlation has been found between change in erosion feature width, length or headward movement and hillslope erosion as measured by erosion pins over the study period. Similarly, no relationship was found between upslope area and erosion, nor between slope and erosion as measured by the erosion pins. Nevertheless and as demonstrated in Figure 3, erosion feature depth increases linearly with both hillslope deposition and erosion. This indicates that the deeper the hillslope erosion and depth then the deeper the nearby gully depth. Deeper hillslope erosion and deposition indicate greater overland flow but with different dynamics for each. But for both cases, a greater increase in gully depth indicates greater flow through the gully. This suggests that the gully is an efficient conduit to move water and sediment from the catchment, regardless of which processes are occurring on the nearby hillslope.

Examination of the relationship between hillslope and channel erosion and deposition on an annual basis revealed no strong trends, suggesting that the system is not sensitive to annual variability and that change occurs over longer time periods. While no major change has occurred over the 6 year period, the presence of vertical headcuts, steep side walls and large pot holes, together with the finding that these features have largely been maintained, suggests that the system had reached a quasi-steady state before the start of the study period.

Long-term studies examining erosion features are scarce (Harvey 2001). The finding that the erosion features have changed little over the 6 year monitoring period is a significant result as during this time the catchment has been subject to fire every second year (Hancock, personal observation) with all surface grass cover removed, as well as both average and in 2007, well above average rainfall. In 2006 the majority of trees were pushed over by winds from a Category 4 cyclone.

The findings reported here are similar to those of Harvey (2001). Though the climate for his study (in northwest England) is very different from Arnhem Land, over a 30-year monitoring period the gullies began a progressive trend towards stabilisation. He suggested that destabilisation may occur in response to rare extreme rainfall, with return periods greater than that on record. A change in landscape management or use may also destabilise the system.

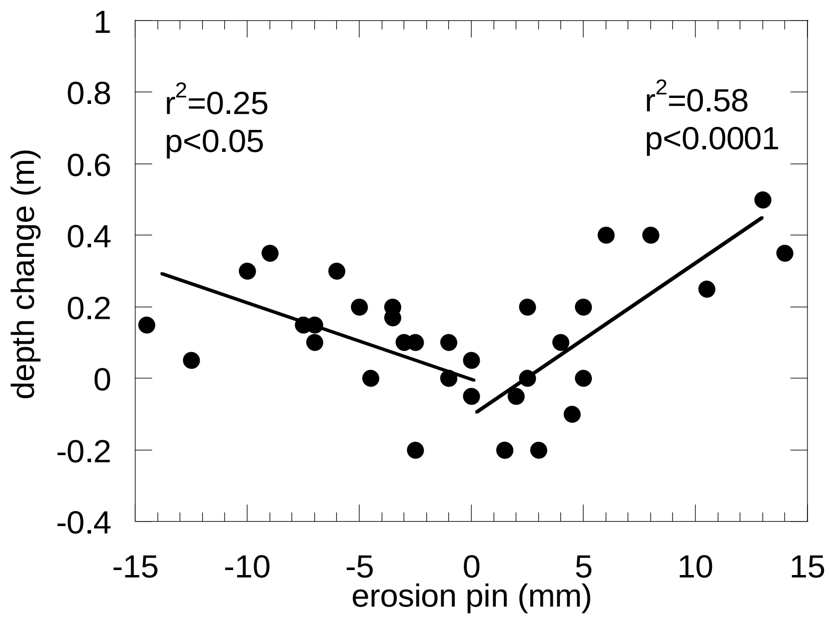

Figure 3 Erosion as measured by erosion pin and the change in erosion feature depth at Tin Camp Creek. Negative values represent deposition while positive values represent erosion.

While some of the erosion features at Tin Camp Creek became shallower (ie filled in) and reduced their width due to deposition, they were maintained and appear to be permanent features of the landscape. From a post-mining landscape perspective, this study shows what erosion features, such as gullies, may evolve to look and behave like in the very long term.

Steps for completion

This is a joint study between Associate Professor Greg Hancock, University of Newcastle, and ***eriss***. Geomorphic studies require lengthy periods of data collection and the study of gullies at Tin Camp Creek will continue to contribute to a database of natural erosion rates for the region. These data will be used for calibration and evaluation of the Siberia and CAESAR erosion and landscape evolution models (Hancock et al 2009). From these data, input parameter values can be derived for Siberia and CAESAR and applied to simulation of erosion of post-mining landforms since parts of the digital elevation model used to represent the mining landform will have areas of undisturbed land.

References

Hancock GR, Evans KG. 2006. Gully position, characteristics and geomorphic thresholds in an undisturbed catchment in Northern Australia, *Hydrological Processes* 20, 2935–2951.

Hancock GR, Lowry JBC, Coulthard TJ, Evans KG & Moliere DR 2009. A catchment scale evaluation of the SIBERIA and CAESAR landscape evolution models, *Earth Surface Processes and Landforms*, in press.

Harvey AM 2001. Coupling between hillslopes and channels in upland fluvial systems: implications for landscape sensitivity, illustrated from Howgill Fells, northwest England. *Catena* 42, 225–250.

Definition of sediment sources and their effect on contemporary catchment erosion rates in the ARR: landslips

GW Staben, MJ Saynor & JBC Lowry

Background

As a result of record rainfall and floods in the 2006/07 wet season (Moliere et al 2007), a number of landslides occurred in the upper reaches of the Magela Creek and East Alligator River. These landslides occurred on well vegetated, exhumed Oenpelli Dolerite surfaces surrounded by Mamadwerre Sandstone, and had the potential to supply sediment to both Magela Creek and the East Alligator River (Figure 1). Currently the Supervising Scientist Division monitors water quality in Magela Creek to assess the impact on stream sediment loads by mine-related activities at the Ranger mine. These landslides have the potential to increase the baseline load of fine suspended sediment to Magela Creek during future wet seasons and are not mine related. It is therefore important to quantify the impact and extent of natural events affecting stream sediment loads, to enable them to be differentiated from mine impacts.

These landslides occurred in areas only accessible by helicopter, thus remotely-sensed data such as ALOS AVNIR-2 have the potential to offer an affordable means to investigate the impact of these landslides across the landscape.

This study's objectives were:

1 To evaluate the use of remotely sensed data in detecting landslides

2 To determine whether the available data can be used to accurately quantify the area of each landslide

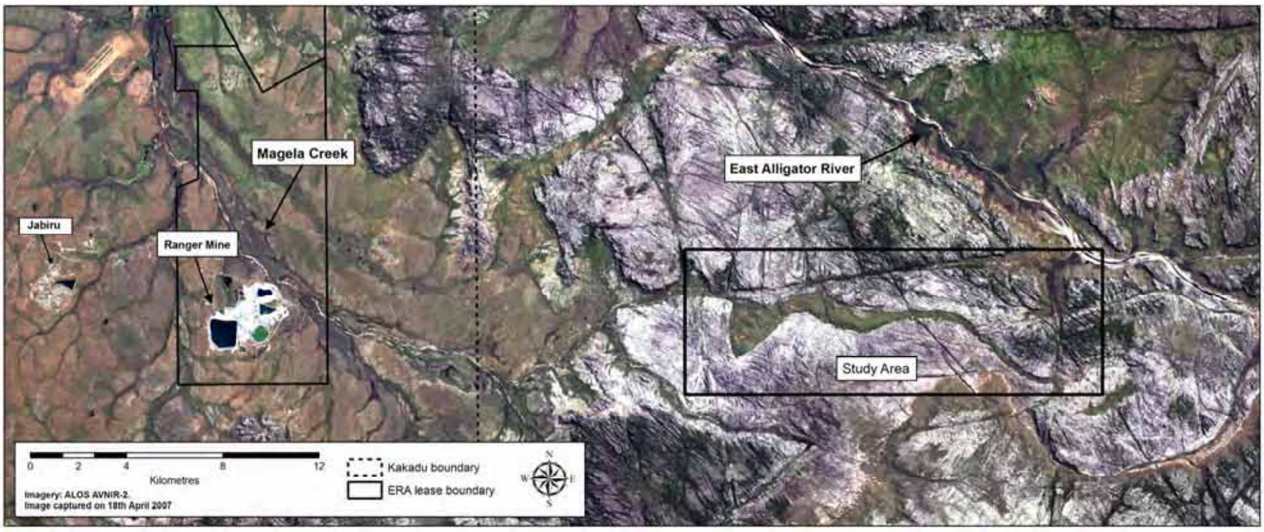

Figure 1 General map showing the location of the landslides study area

Methods

Materials

Two ALOS AVNIR-2 (Advanced Visible and Near Infrared Radiometer type 2) multispectral satellite images obtained from Geoscience Australia were used for this study. The ALOS AVNIR-2 is a push broom sensor with four spectral bands capturing data in three visible bands (blue, green and red) and one near infrared band. It has a swath width of 70 km and a spatial resolution of 10 m at nadir. The spatial resolution of an ALOS AVNIR-2 image is determined by the point angle of the sensor at time of image capture (ALOS User Handbook 2007). The only suitable image available to assess the area prior to the landslides was captured in June 2006 at a spatial resolution of 20 m. The second (post landslide) image was acquired in April 2007 at a spatial resolution of 10 m. Both images consisted of four spectral bands. A one second DEM (digital elevation model) was also used in this project. It was produced by the Defence Imagery and Geospatial Organisation and has a spatial resolution of ~30 m and a vertical accuracy estimated to be ± 17 m.

Image analysis

Image pre-processing

A number of image pre-processing steps were performed including re-registration of the 2006 image to the 2007 image and extraction of slope values (in degrees) from the DEM using the topographic modelling tool in the remote sensing software ENVI. The normalised difference vegetation index (NDVI) was then derived from the two ALOS AVNIR-2 image dates. Vegetation indices were used, as the contrast between the boundaries of the landslides was high. The four spectral bands and vegetation index for each image date and the slope data (derived from the DEM) were then combined in an 11 band layer stack at a spatial resolution of 10 m for further analysis in eCognition.

Classification

A multi-resolution image analysis was undertaken using the geographic object based image analysis (GEOBIA) software, eCognition. GEOBIA using eCognition involves two main steps, firstly, segmentation of the image into objects and then classification based on the statistics derived from these objects (Al-Kudhairy et al 2005). The multi-resolution segmentation of an image enables the user to investigate objects at a variety of spatial scales (Benz et al 2004) allowing targets to be identified and classified at a range of different spectral and spatial scales. Information from the different spatial scales can then be combined into a hierarchical network where each object is related to its context, neighbourhood, and sub- or super-objects (Bock et al 2005).

To exploit the different information available at different spatial scales, a number of segmentations were performed on the data (Figure 2). The initial segmentation (Level 1) was set at a scale to define individual landslides whilst the second (Level 2) was undertaken to enable discrimination between the sparsely vegetated sandstone plateau/escarpment regions of the image and the more fertile soil substrates which support vegetation growth. The level 2 segmentation was important as it enabled areas such as the sandstone regions in the image, which have similar spectral properties as landslides to be identified and correctly classified.

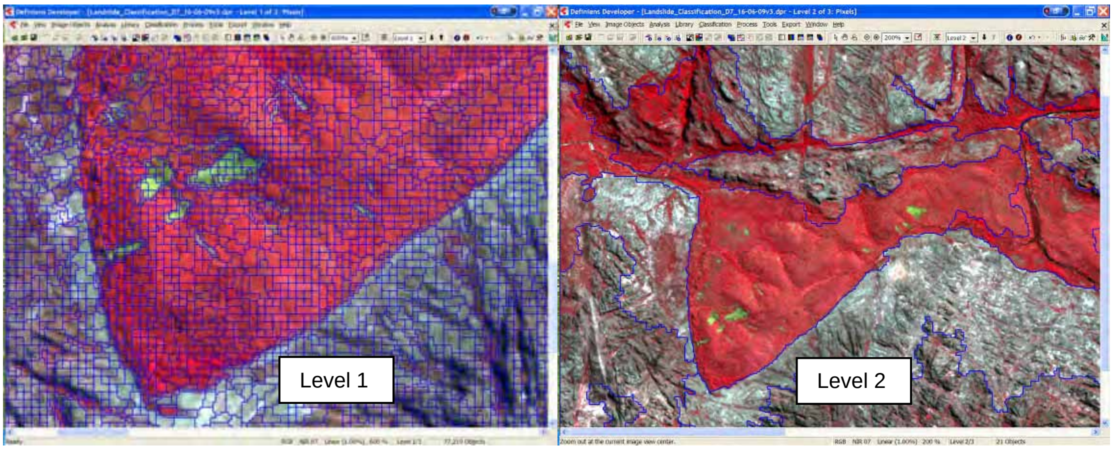

Figure 2 Examples of the multi-resolution segmentation performed on the data

A number of steps were involved in the classification process. The first was to classify the image into two broad categories (sandstone plateau/escarpment and vegetated areas) using the level 2 segmentation. This information was then used in the subsequent classification of the finer scale segmentation (level 1). A classification using membership functions was then developed for the finer scale segmentation with classes identifying areas of shadow/water, vegetation, bare ground, landslides and their associated debris flow. Investigation of the results of the level 1 classification indicated that some areas around the landslides were being incorrectly classified as debris flow. In order to enhance the classification, a chessboard segmentation (Level 0) was undertaken on the areas classified as debris flow in level 1 with further refinement of the classification then undertaken.

Accuracy assessment

Accuracy assessment was undertaken to assess both the thematic and geometric accuracy of the automated classification. Two methods were used, first a manual classification of landslides was undertaken. This included identifying and estimating the area of each landslide derived from visual interpretation of the multispectral ALOS AVNIR-2 satellite data. Vector polygons were produced using heads-up digitising methods for each landslide in a GIS (Geographical Information System) software, ArcMap. This process was undertaken with the image set at a consistent scale (1: 5000) and with the aid of oblique digital photographs taken during aerial surveys. Estimates derived from the satellite data were undertaken to quantify the number of landslides and to obtain areal estimates for all landslides in the study area, with the presence of landslides validated by aerial surveys conducted by helicopter. The second method assessed the geometric accuracy of the classification. Ground truth data were collected for 11 of the landslides located within the Magela catchment. This was done by capturing points around the perimeter of each of the landslides at approximately 3 to 10 m spacing's using a GPS (± 5m horizontal spatial accuracy). These data were then input into a GIS and converted to vector polygons for further analysis.

Progress/results

Visual assessment of the ALOS AVNIR-2 data and aerial surveys (by helicopter) identified a total of 56 landslides across the 88 km^2 study area. Comparison of these data with the results of the classification show that we were able to successfully detect 60.7% (n = 34) of the 56 landslides. Twenty-two landslides were not detected by the classification algorithm while 8 objects were incorrectly classified (commission error) as landslides. The results of the classification estimated the total area of landslides to be 11.3 ha, representing a significantly higher value than the manual classification estimate of 6.4 ha (Table 1).

Of the 6.4 ha in area estimated to be impacted by landslides the classification was able to detect 5.96 ha, while the 22 undetected landslides covered an area of 0.39 ha. This suggests that while the detection rate for individual landslides was only 60.7%, the classification was still able to detect the majority (94%) of the area impacted by landslides. Assessment of the mean size difference (based on the manual classification area calculations) between the 34 detected (0.18 ha) and 22 undetected (0.02 ha) landslides, would suggest that the ability to detect and correctly classify landslides using ALOS AVNIR-2 imagery and the GEOBIA approach is related to the size of the landslide.

The geometric accuracy of the mapped landslides was assessed by directly comparing 11 of the landslides measured by imagery, with ground truthed data for the same landslides. All 11 landslides were detected by the classification algorithm, though again, there was a significant over-estimation of the area classified as landslides using image data (Table 2).

Table 1 Summary statistics for the automated and manually-classified landslides

	No. landslides identified.		**Total area (ha)**		**Mean area of individual landslides (ha)**	
	Automated classification	Manual classification	Automated classification	Manual classification	Automated classification	Manual classification
Detected	34	56	11.27	5.96	0.33	0.18
Omission error	(22)			0.39		0.02
Commission error	8		1.50		0.19	
Total	42	56	12.77	6.36	0.33	0.11

Table 2 Direct comparison between the areas of the 11 landslides measured during the collection of ground truth data and the classified landslides

Landslide	**Ground truth area (m^2)**	**Classification (m^2)**	**Difference in area (m^2)**
1	118	1000	882 (88%)
2	1241	2600	1359 (52%)
3	233	1600	1367 (85%)
4	262	700	438 (63%)
5	301	600	299 (50%)
6	1547	3500	1953 (56%)
7	1293	4800	3507 (73%)
8	5296	7900	2604 (33%)
9	4321	8700	4379 (50%)
10	4641	12900	8259 (64%)
11	174	1000	826 (83%)
Total	19427	45300	25873 (57%)

Comparison of the actual (ground-truthed) area of the 11 landslides to the classified areal data showed that 94% of the area covered by each of the 11 landslides on the ground was captured in the classification (Table 3). However, the commission error was very high with an average of 60% for the 11 landslides. This over-estimation of landslide area can be attributed to the miss-classification of debris flow as landslide. The area of each landslide not detected (omission error) was low with exception of landslide five where 66% went undetected. This was due to the geometric shape of the landslide which was predominantly long and thin (less than a pixel in width) resulting in the mixed pixel effect, which is due to the spectral response of the tree canopy and landslide combined. However, the scour depth of this landslide resulted in significant debris flow down-slope which aided in its detection .

Conclusions

The results of this study show that GEOBIA classification methods detected 60.7% of the landslides across the study area. It was found that even though the detection rate was modest, those landslides that were detected were estimated to represent 94% of the total area impacted by landslides in the study area. The spatial and spectral resolution of ALOS AVNIR- 2 data alone were not sufficient to discriminate between landslides and their associated debris flows. The spectral response of bare soil areas, landslides and their associated debris flow was very similar. With the addition of slope values derived from the DEM data, the discrimination between landslides and other bare soil areas in the image was possible, as landslides did not occur on flat areas. Analysis of the ground-truthed data shows that while 94% of the landslide area was captured in the classification, significant areas of debris flow were also miss-classified as landslide area. This miss-classification of debris flow as landslides was found to be due to the existence of debris flow on areas with high slope values. The results of this project show that object-based analysis of ALOS AVNIR-2 remotely sensed data can be used to detect the significant landslides in the study area. However, the use of these results to measure the landslide area is not feasible as discrimination between a landslide and its associated debris flow is not possible.

Table 3 Results of the analysis between the automated classification and the 11 landslides measured during the collection of ground truthing (GT) data

Landslide	Ground truth area (m^2)	Classification (m^2)	Area classified within the extent of the GT data	Commission error (m^2)	Omission error (m^2)
1	118	1000	93	907	25
2	1241	2600	1170	1430	71
3	233	1600	232	1368	1
4	262	700	232	468	30
5	301	600	102	498	199
6	1547	3500	1547	1953	0
7	1293	4800	1230	3570	63
8	5296	7900	4583	3317	713
9	4321	8700	4295	4405	26
10	4641	12900	4599	8301	42
11	174	1000	123	877	51
Total	19427	45300	18205 (94%)	27095 (60%)	1222 (6.29%)

References

ALOS User Handbook 2007. Earth observation Research Centre, Japan Aerospace Exploration Agency, http://www.eorc.jaxa.jp/ALOS/doc/alos_userhb_en.pdf

Al-Kudhairy DHA, Caravaggi I & Giada S 2005. Structural damage assessments from IKONOS data using change detection, object-oriented segmentation, and classification techniques. *Photogrammetric Engineering & Remote Sensing* 71, 825–837.

Benz UC, Hofmann P, Willhauck G, Lingenfelder I & Heynen M 2004. Multiresolution, object-oriented fuzzy analysis of remote sensing data for GIS-ready information. *ISPRS Journal of Photogrammetry & Remote Sensing* 58, 239–258.

Bock M, Xofis P, Mitchley J, Rossner G & Wissen M 2005. Object-oriented methods for habitat mapping at multiple scales – Case studies from Northern Germany and WyeDowns, UK. *Journal for Nature Conservation* 13, 75–89.

Moliere DR, Evans KG & Saynor MJ 2007. Hydrology and suspended sediment transport in the Gulungul Creek catchment, Northern Territory: 2006–2007 wet season monitoring. Internal Report 531, June, Supervising Scientist, Darwin. Unpublished paper.

Assess the impact of tailing subsidence on rehabilitated landform erosional stability

R Houghton

Introduction

Where the design of a rehabilitated mine landform includes disposal of unconsolidated tailings in worked-out open pit(s) and the capping of the tailings with benign material, it is critical that the backfill and capping design accommodates tailings settlement and that surface drainage quickly removes water to reduce infiltration, surface erosion and possible exposure through gully incision. If tailings consolidate at a rate and to an amount that causes the cap to subside and fracture, drainage line direction can change, causing surface flow to accumulate and discharge through a single point, causing severe incision. Fractures may also expose tailings and allow both direct inflow and percolation of water through the containment area and transport of contaminants into the groundwater system.

It is important that the impact of tailings subsidence is understood so that the effect can be allowed for in the final design of the capping structure for Pits 1 and 3 at the Ranger mine site. Investigation of uranium mine pits in the region (Nabarlek and Rum Jungle) in which tailings have been placed and subsequently backfilled and capped provides the opportunity to better understand the longer term effects of tailings subsidence in capped pits. For this research Dyson's Pit at Rum Jungle was selected as the study site to provide information that could be used to assist the design process for capping Pits 1 and 3 at the Ranger mine site.

The Rum Jungle site, comprising Whites Pit, Intermediate Pit and Dyson's Open Cut and Whites underground workings, produced uranium, copper, lead and zinc and was operational between 1952 and 1971. Rehabilitation of Dyson's Open Cut (the primary uranium pit) was completed in late 1984. This involved backfilling the pit with the processed tailings, contaminated low grade copper ore and contaminated subsurface soils from locations around the mine site. This material was capped with a constructed interblanket drainage layer shaped with a continuous low angle grade longitudinally and a transverse concave grade which converges to a central rock lined diversion channel with a final layer of plant growth medium (Pidsley 2002) (Figure 1). Information on how the rehabilitated Dyson's Pit has performed to its design characteristics provides important information that could be used in the proposed strategies for the backfill capping of tailings at Ranger mine site.

Aims

The project aim was to measure and model settlement of the capped Dyson's pit to identify specific issues that may need to be considered and addressed for capping of the in-pit tailings at the Ranger Mine. As the landform matures, slope angles and elevations may change as a result of consolidation. Thus it is important to understand how consolidation may impact on settlement of the backfilled pit landforms and, in particular, how this differential settlement may effect erosion of material from the capped pits and the behaviour of the capping itself in the context of being a sustainable water shedding construct. Specifically, it needs to be determined how the effects of differential consolidation can be incorporated in landform evolution modelling and therefore improve (in the reverse engineering sense) the final design of the rehabilitated landform.

Figure 1 Longitudinal drain in the cap of Dyson's Open Cut. Significant signs of subsidence can be seen in the vicinity of the single tree located towards the mid-background in the top right of the photograph. The drain has subsided here and acts as a basin to accumulate runoff. Signs of acid rock drainage are visible in the rock mulch in the bottom left of the photograph and in the drain toward top centre where rock mulch colour changes from grey to rust/brown.

Figure 2 Accumulation of acidic and metalliferous runoff in subsided section of surface drain. Riprap seen in foreground of Fig 1 is upslope in this photo.

Methods

The present day surface of the Dyson's landform was surveyed in 2008 to obtain a digital elevation model (DEM). The 1986 'as-constructed' surface survey was only available as a hard-copy contour plan. This plan was manually digitised and ArcGIS software used to create a DEM. A comparison of the 2008 DEM to the 1986 DEM enabled the change in surface level between the DEMs to be quantified. It was assumed that the measured differences were the result of consolidation/settlement of the buried tailings between 1986 and 2008.

The Tezaghi (1945) model (see Smith 1990, 361) using simplified linear and elastic consolidation theory for fine grained soils was used to mathematically quantify one-dimensional settlement of the cap. The main assumptions in the theory are:

1. The soil is essentially homogenous and saturated,
2. Both the soil pore water and soil particles are incompressible,
3. The coefficient of consolidation (C_v) is constant through time,
4. Darcy's law of saturated flow applies,
5. The compression-induced consolidation is in one dimension (vertically down) only,
6. Expulsion of water from the soil voids is in one direction only (vertical),
7. The change in the bulk volume of buried material is only due to a change in the void ratio, which in turn is due to a corresponding change in the effective stress. The relationship between void ratio (e) and effective stress (σ') is linear.
8. There is no instantaneous volume change on application of the overburden pressure increase (total stress increase).

Results

The topographic survey results comparing 1986 with 2008 are shown below in Figures 3, 4 and 5. Settlement due to one dimensional consolidation was estimated using the numerical method of finite difference. The model parameter values were iteratively fitted by varying the coefficient of volume decrease (m_v) until the current, observed settlement value of 2.9 m was returned for the period of 22 years. The simulated variation of settlement with time is shown in Figure 6.

The observed maximum settlement of the landform surface approximately 1.5 y after construction was 900 mm (Allen & Verhoeven 1986). This value compared well with the modelled settlement of ≈800 mm after 1.5 y. The 100 mm settlement difference suggests a reduced pore water dissipation rate in the model and is an acceptable discrepancy due to the simplicity of the 1-dimensional model. The model indicates that the amount of settlement decreases with time (Figure 6) and that equilibrium will slowly be achieved as excess pore waters are dissipated.

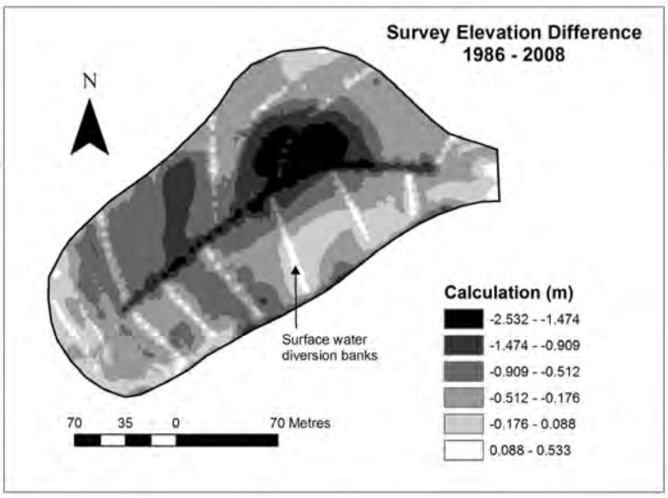

Figure 3 Dyson's surface overlay of 1986 and 2008 data showing difference in height

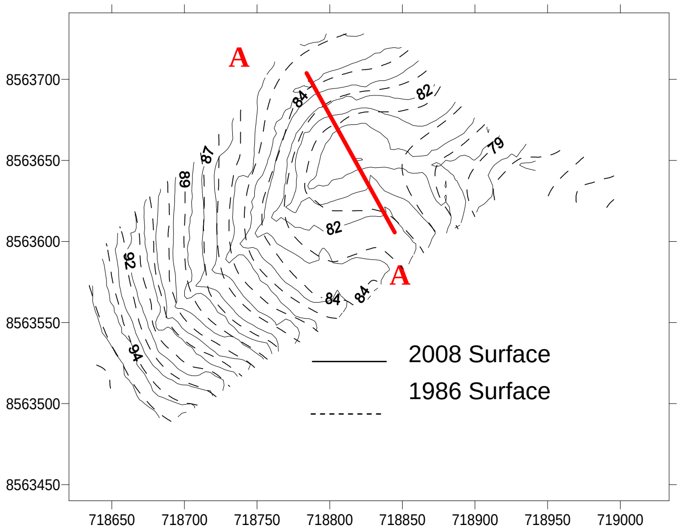

Figure 4 1986 & 2008 contour map overlay showing Section A-A. Dimensions are in metres.

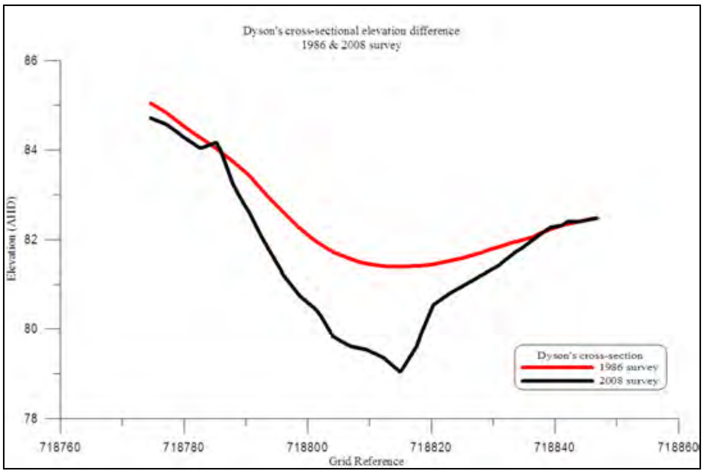

Figure 5 Section A-A showing the subsidence of the cap that has occurred between 1986 (top line) and 2008 (bottom line)

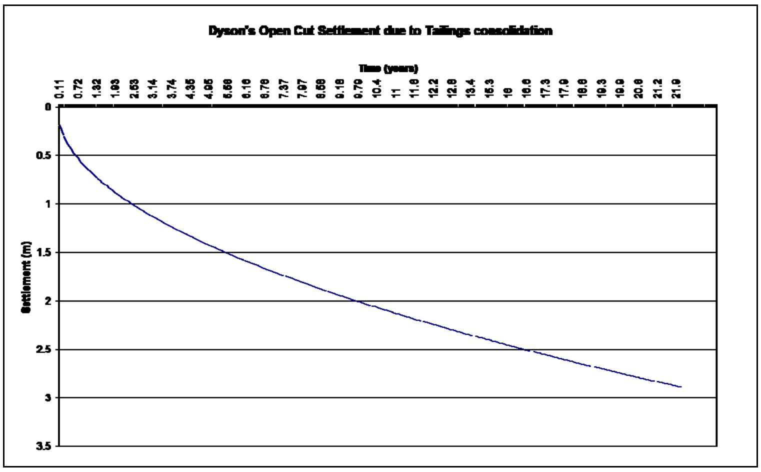

Figure 6 Plot of simulated one-dimensional consolidation/settlement against time

Conclusions

The capped pit area of Dyson's Pit was observed to have settled in the years following its construction, indicating that the original design did not adequately account for consolidation. Evidence of this consolidation has been disruption of the surface line drainage causing ponding of acidic water that promotes infiltration rather than runoff. Steeper slopes within the structure flowing into the depression could also increase erosion. These consequences of not adequately accounting for consolidation at Dyson's Pit emphasise the importance of appropriately accounting for this process in the design of the capping and backfill for the Ranger pits.

The lack of specific site geotechnical data meant that only Terzaghi's (1945) one dimensional consolidation model could be used for this work (Smith 1990, 361). The major limiting factors influencing a realistic comparison of model-predicted settlement to the observed settlement at Dyson's is the limited knowledge of the tailings geotechnical characteristics. The geotechnical characteristics could be determined by a drilling program that recovers relatively undisturbed tailing samples for laboratory testing. A drilling program has the potential to yield substantial critical information such as:

1. The vertical structure of the backfilled pit could be mapped, quantifying layer depths both in the capping layer and the underlying tailings.
2. The volume change behaviour of the tailings would be better understood through laboratory testing of the samples recovered.
3. The process by which further settlement occurs can be determined by installing piezometers in the bore holes from which the samples for geotechnical testing have been extracted. The pore water pressures may then may be monitored through time. If the recorded pressures remain relatively constant, further settlement may be attributed to secondary compression. The pore pressure information would provide the data needed to develop a more realistic settlement model.
4. Water pressure data in the pit bore could be compared to data from groundwater bores in the natural terrain downgradient from the pit to determine the flow path boundary conditions for water in the pit.

In the event that these additional data are obtained then much more comprehensive two dimensional and three dimensional models be applied.

Several successful outcomes of the study were achieved, including:

1 The settlement of the Dyson's rehabilitated landform between the years 1986 and 2008 has been quantified with the use of precision survey instruments and current gridding and mapping software to the best possible accuracy given the historical data available.

2 The major characteristics that will influence the potential instability of a landform constructed above a layer of impounded tailings are now better understood.

Whilst the modelling outputs from this study are not specifically applicable to Pits 1 and 3 at the Ranger mine, the findings clearly emphasise the need to account for the effects of consolidation during the design phase.

References

Allen CG & Verhoeven TJ (eds) 1986. The Rum Jungle rehabilitation project final project report, Department of Mines and Energy, Northern Territory.

Pidsley SM (ed) 2002. Rum Jungle Rehabilitation Project Monitoring, Report 1993–1998. Northern Territory Government Department of Infrastructure, Planning and Environment.

Smith GN 1990. *Elements of soil mechanics.* 6th ed, BSP Professional Books, London.

Pre-mining radiological conditions at Ranger mine

A Bollhöfer & A Esparon

Introduction

The total annual effective radiation dose to a member of the public from practices such as uranium mining should not exceed 1 milli Sievert (mSv) as recommended by the International Commission on Radiation Protection (ICRP 2007). This dose is on top of the natural pre-mining background dose. In a high natural background area such as the area around the Ranger mine, determining an additional dose due to mining activities presents a challenge, especially when pre-mining data are scarce and focus on delineating the extent and location of an orebody, rather than determining area wide radiological conditions. Hence, pre-mining conditions need to be assessed accurately so that post-mining changes can be quantified in the context of the success of rehabilitation in complying with radiological exposure standards.

Historical airborne gamma surveys (AGS) coupled with ground truthing surveys, have the potential to provide a powerful tool for an area wide assessment of the pre-mining terrestrial gamma dose rates. Recent AGS coupled with ground truthing surveys have been used for regional assessments of radiological conditions at rehabilitated and historic mine sites elsewhere in the Alligator Rivers Region (Martin et al 2006, Bollhöfer et al 2008). The aim of this project is to ground-truth historical AGS data at an appropriate undisturbed radiologically anomalous site in order to extrapolate to pre-mining radiological conditions at Ranger.

An AGS of the Alligator Rivers Region was flown in 1976 and has been used to identify undeveloped radiologically anomalous areas in the vicinity of the Ranger lease as potential candidates for groundtruthing. A comparison of signal intensity with known uranium occurrences in the MODAT database suggested that Anomaly 2 to the south of the Ranger lease may be a suitable analogue site for Ranger pre-mining radiological conditions, as it exhibited a strong airborne gamma signal in the 1976 data (Esparon et al 2009). Based on the assessment of the historical AGS data it was decided to obtain groundtruthed data in the greater region of Anomaly 2. An extensive fieldwork program to the south of the Ranger lease was started in 2007, and this has continued through subsequent dry seasons.

Methods

The 1976 AGS data were acquired from Rio Tinto by the NT Government, and are available on the public domain (the *Alligator River Geophysical Survey*). Data were re-processed in 2000 by the Northern Territory Geological Survey (NTGS) and then resampled by NTGS at a pixel size of 70 m in 2003. The line spacing of this survey was 300 m. However, the flying height is unknown.

Extensive ground gamma radiation surveys of Anomalies 2A and 2B were conducted in 2007. In addition, ERA has made available to SSD data from an AGS that was flown in 1997 at a low flying height (50 m) and a higher spatial resolution (200 m line spacing) than the 1976 survey. This data set was used for refining further extensive groundtruthing fieldwork conducted in July – October 2008 to precisely establish the location of Anomalies 2A and 2B.

As of August 2009 more than 1800 individual gamma dose rate readings (and some readings with an in-situ NaI detector) have been obtained in the area.

Radon exhalation was measured in July 2009 at 25 sites using conventional charcoal canisters, with 3 charcoal canisters deployed at each site for a period of three days. In addition, external gamma dose rates were measured and soil scrape samples were taken at these radon sites. Track etch detectors were also deployed for three months to measure dry season airborne radon concentration. Analysis of airborne radon concentration data and soil activity concentrations are underway.

Results

Gamma radiation

During the 2008 ground gamma surveys a spatial shift was confirmed of approximately 200 metres between the groundtruthed and the 1997 AGS data. This spatial shift needs to be accounted for before the two datasets can be quantitatively compared. Figure 1 shows the shifted and interpolated 1997 AGS data (total counts) and a comparison with the raw gamma dose rate results (μGy·hr^{-1}) from SSD's groundtruthing in 2007–2009, respectively.

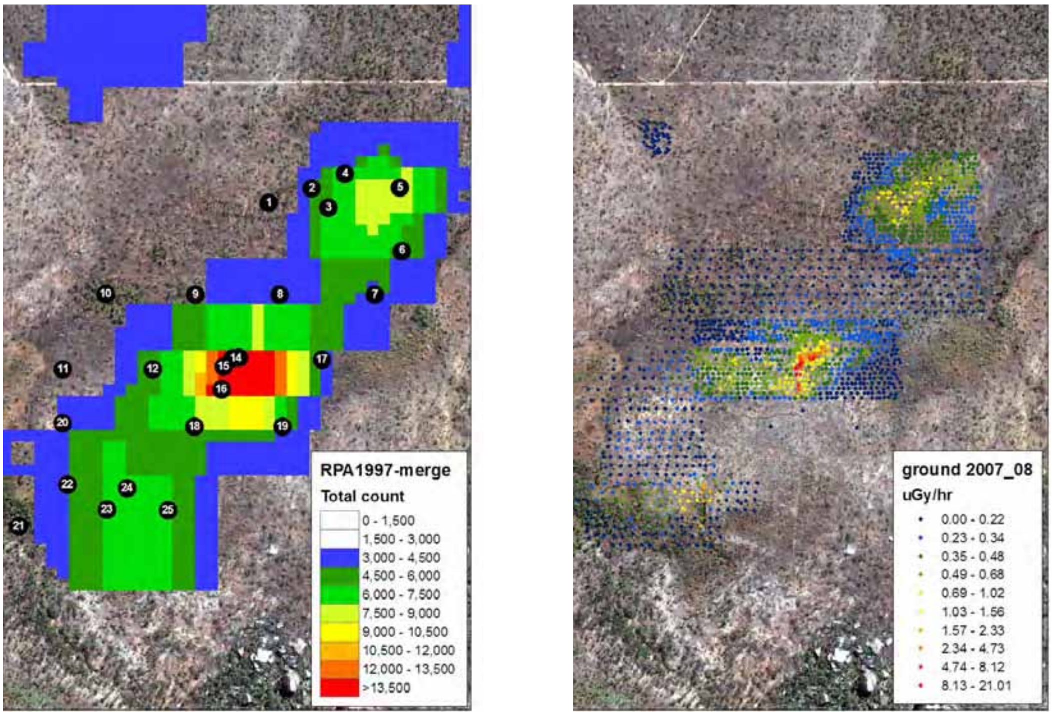

Figure 1 Interpolated1997 AGS data and locations of the radon exhalation survey points (left) and the results of the on ground gamma dose rate measurements (right) performed from 2007 to 2009

To groundtruth an AGS, the data acquired in the field (uranium, thorium, potassium concentrations, and/or gamma dose rates) are plotted against the count rates in the respective channels from the AGS. Ground-based data acquired along transects, for example, are then typically smoothed using an *n*-point average resulting in a resolution similar to the resolution of the AGS (see for example Martin et al 2006, Bollhöfer et al 2008). The groundtruthed data at Anomaly 2 have been acquired at a much higher resolution than both the 1997 and 1976 AGS data (the image is much 'sharper') and it is thus essential to determine appropriate 2-dimensional smoothing algorithms to allow a comparison to be made between the groundtruthed and the AGS data. For example, choosing too small a pixel size to smooth the groundtruthed data will lead to a comparatively steeper slope when plotting groundtruthed versus AGS data, as localised areas on the ground that exhibit high gamma dose rates will have more influence on the regression. In contrast, a pixel size that is too large will lead to a gentler slope, approaching zero as pixel size is increased. Algorithms are being developed at present using *MatLab*, to smooth the 2-dimensional data such that the correlation of the two data sets shows the best regression coefficient (R^2). Once a relationship between the 1997 AGS and the 2007–09 ground truthed data has been established a correlation will be determined between the 1976 and 1997 AGS data sets. Initial screening of the two (1997 and 1976) AGS data sets (see Figure 2) shows a good correlation ($R^2 = 0.7$) between the two. Note that both datasets needed to be spatially corrected before a comparison was made.

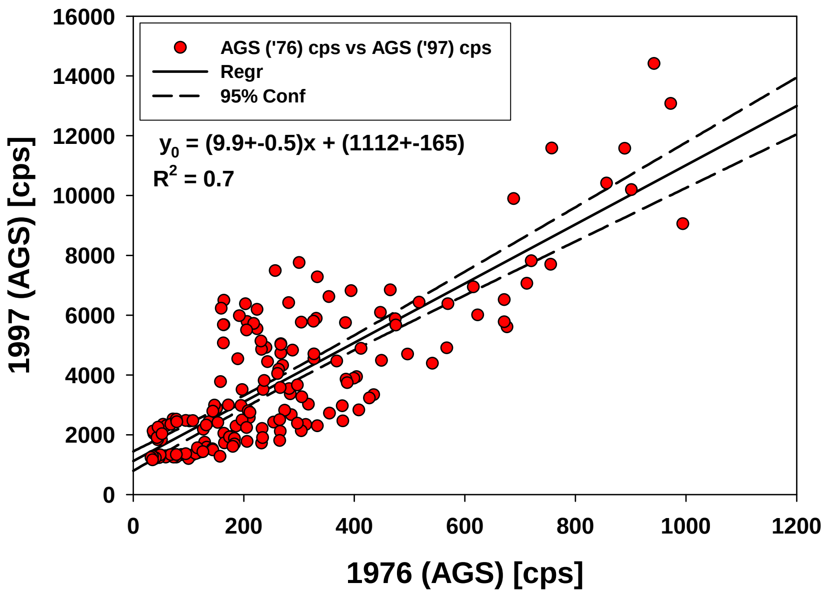

Figure 2 Total counts from the 1997 AGS plotted against total counts from the 1976 AGS

Radon flux densities

Radon flux densities have been measured across the Anomaly 2 area (black dots in Figure 1) and show large variations due to the large variations in radium activity concentrations in the soils. Figure 3 shows the geometric means of the radon flux densities plotted versus the terrestrial gamma dose rates measured at the sampling sites on a log-log scale (the inset shows the data from 0–2.5 $\mu Gy \cdot hr^{-1}$ on a linear scale).

The sampling sites have been divided according to soil type, identified by visual inspection in the field, and the results are plotted for fine gravel, loamy sand and coarse gravel/rocks, respectively. There is no correlation between radon flux densities and gamma dose rates measured for coarse material and rocks. This effect has been observed previously and is due to the variable pore space and thus highly variable radon exhalation originating from coarser material (Lawrence et al 2009). However, there is a strong positive correlation between radon flux densities and gamma dose rates for both, fine gravel and loamy sand soil types.

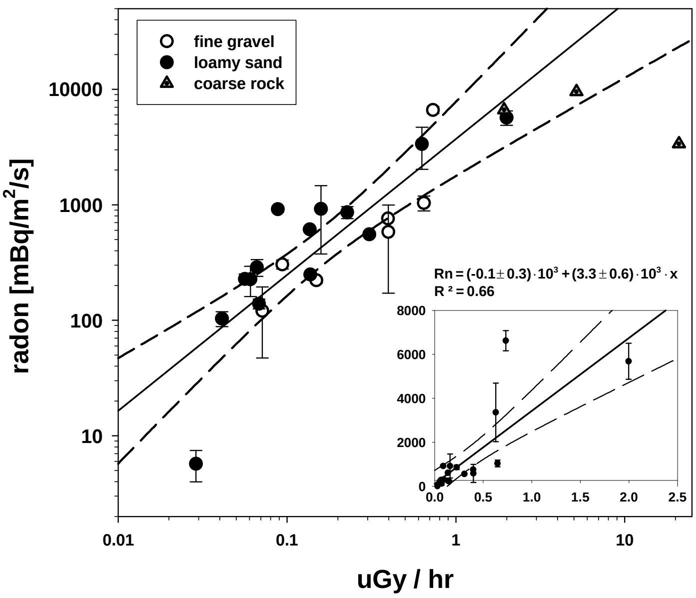

Figure 3 Radon flux densities plotted versus the terrestrial gamma dose rates measured at Anomaly 2

Statistical analysis of the variance of the data using a general linear model shows that there is no difference between the radon flux densities determined for fine gravel and loamy sands. However, there is a small but statistically insignificant difference (p= 0.06) in the slopes. Consequently, the regression using the combined (gravel and loamy sand) dataset is shown in Figure 3, which allows calculation of radon exhalation from known terrestrial gamma radiation data at Anomaly 2. A similar regression (radon flux density versus ^{226}Ra soil concentration) will be plotted once gamma spectrometry results for the soils collected are available. This regression will then be compared with results obtained previously in the region (Lawrence et al 2009) and may be used to determine the radon flux from Anomaly 2 to extrapolate to orebodies 1 and 3.

Steps for completion

Analyses of soil radionuclide activity concentrations need to be finalised. This will allow a correlation to be established between ^{226}Ra soil activity concentrations and radon flux densities at Anomaly 2, similar to the R_{E-R} determined by Lawrence et al (2009). This correlation can then be used to determine radon fluxes from Anomaly 2. Radon fluxes will also be related to radon activity concentrations measured in the air via track etch detectors deployed across the Anomaly. Track etch detectors were placed in the field in July 2009 and collected in October 2009. Results are pending.

Algorithms need to be developed to upscale results from the groundtruthing data so that comparison can be made with both the 1997 and 1976 AGSs. Once data analysis is complete, the radiological conditions on ground around Anomalies 2A and 2B will be correlated to the pre-mining airborne signal in an attempt to extrapolate to the area wide radiological conditions at Ranger before mining commenced.

Acknowledgments

Jared Selwood, Bruce Ryan, Alan Hughes, Rocky Cahill and Gary Fox are thanked for assistance in the field. The Northern Territory Geological Survey and ERA are thanked for the provision of the 1976 and 1997 AGS data, respectively. Thanks to the Mirarr people for allowing access to the sites.

References

Bollhöfer A, Pfitzner K, Ryan B, Martin P, Fawcett M & Jones DR 2008. Airborne gamma survey of the historic Sleisbeck mine area in the Northern Territory, Australia, and its use for site rehabilitation planning. *Journal of Environmental Radioactivity* 99, 1770–1774.

Esparon A, Pfitzner K, Bollhöfer A & Ryan B 2009. Pre-mining radiological conditions at Ranger mine. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 111–115.

ICRP 2007. *The 2007 Recommendations of the International Commission on Radiological Protection*. International Commission on Radiological Protection Publication 103, Elsevier Ltd.

Lawrence CE, Akber RA, Bollhöfer A & Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Martin P, Tims S, McGill A, Ryan B & Pfitzner K 2006. Use of airborne γ-ray spectrometry for environmental assessment of the rehabilitated Nabarlek uranium mine, northern Australia. *Environmental Monitoring and Assessment* 115, 531–553.

Radon exhalation from a rehabilitated landform

A Bollhöfer, B Ryan, A Esparon & J Pfitzner

Introduction

A trial landform has been constructed on the minesite to study the effects of different growth media on the establishment of vegetation, both via direct seeding and planting of tubestock. It is located immediately adjacent to the north west corner of the Ranger tailings dam and covers an area of approximately 8 ha. Radon exhalation measurements will be carried out through time on the different surface treatments to provide data for the development of a post rehabilitation radiological dose assessment model. For more detail on construction of the landform and erosion plots see Saynor et al (2009) and the erosion studies project report 'Revegetation trial and demonstration landform: erosion and chemistry studies (pp 109–112, in this volume).

In this project, radon (^{222}Rn) exhalation rates for various covers and vegetation types, taking into account weathering, erosion and compaction effects, and the effect of developing vegetation on the landform, will be determined. Specifically the ^{222}Rn exhalation from the four SSD erosion plots (30 m × 30 m) will be measured over several years.

Soil closure criteria for the rehabilitation of Ranger Uranium mine need to incorporate radiological aspects to ensure that exposure of the public to radiation after rehabilitation of the mine is as low as reasonably achievable. As the inhalation of radon is likely to be a main contributor to radiological dose (Supervising Scientist 2007), in particular in the vicinity of the rehabilitated landform, radon exhalation and its temporal variability need to be estimated. The radon exhalation rate may change as the final landform evolves after rehabilitation of the site, and the trial landform will provide a unique opportunity to determine this variability under experimental conditions over a period of several years. The project will enable ***eriss*** and ERA to predict a long-term radon exhalation flux from the rehabilitated landform and contribute to the development of closure criteria.

The project started prior to the building of the landform, to determine the radon exhalation baseline of the original, undeveloped trial landform footprint, since ^{222}Rn can diffuse from depths of several meters to the surface (with lower layers making a decreasing contribution). The diffusion of ^{222}Rn can generally be described by Fick's law and ^{222}Rn diffusion length for dry soils has been reported to be in the 1–2 m range (Graaf et al 1992) and 2–5 m for sandy type materials (Holdsworth & Akber 2004).

Methods

Conventional charcoal canisters (or 'radon cups) are used to measure radon exhalation. The charcoal canisters used are a standard brass cylindrical design with an internal diameter of 0.070 m, depth 0.058 m and a wall thickness of 0.004 m. Details on the charcoal canister methodology are given in Bollhöfer et al (2005) and Lawrence (2004).

To get a true average radon exhalation from the four erosion plots, radon cups were placed randomly and put onto rocks if required, sealing the rim using a putty. This is in contrast to many other studies where radon cups are put at 'convenient' spots where they can easily be embedded into the finer grained soil. Fine grained material exhibits higher radon exhalation flux densities than solid rock. Hence, results of radon exhalation measurements can potentially be skewed towards higher values if the sampling design is not random (Bollhöfer et al 2005).

Radon cups were deployed in August 2008 on an undisturbed area of the trial landform footprint and exposed for three days to determine the radon exhalation from the substrate underlying the constructed landform. As heavy machinery was already deployed and used for trial landform construction during the survey, the entire area was not accessible. It was planned to repeat the radon exhalation survey at the same area after the top 20 cm of soil were stripped, prior to construction of the trial landform. However the same area was not accessible after stripping. Consequently, a post-stripping survey was conducted approximately 200 m to the west, in a different area of the trial landform, from 13–16 October 2008. It is important to note that heavy rainfall occurred on 14 October 2009, while the radon cups were deployed.

Construction of the landform was finished late in the 2008–09 wet season. Since then irrigation water has been regularly applied to all areas apart from a 40 m buffer strip that contains the SSD erosion plots. This watering has been done to aid the establishment of vegetation on the trial landform through the 2009 dry season. As soil moisture content has a substantial effect on radon exhalation (and because some of the areas were constantly wet) radon exhalation was measured from the four erosion plots only, which were not irrigated. In addition, there is the possibility that the irrigation water contained significant concentrations of radium, which would be adsorbed onto the surface of particulates in the top 5–10 cm of the soil (Akber & Marten 1992) and hence increase radon exhalation rates from the surface (Lawrence 2004). Consequently, radon exhalation measurements were not made on the irrigated landform areas.

The location and a description of the four erosion plots where measurements were performed is shown in Figure 1 and Table 1. Twenty radon cups were deployed randomly across each erosion plot and exposed for four days in May 2009. They were then collected and sent to the Darwin laboratories, where they were analysed using a NaI gamma detector.

Figure 1 shows the location and the distribution of measurement results for the May 2009 radon exhalation survey on the four erosion plots. Also marked are the locations of the 2008 pre-construction survey points (open red diamonds: pre-construction August 2008; open black diamonds: post stripping October 2008). The area where the post-stripping survey was conducted is now covered by waste rock and a 5 metre layer of a 30% laterite–waste rock mix, which has been irrigated over the past few months.

Progress to date

Radon flux densities from the pre-construction substrate follow a log-normal distribution with a range from 24 to 144 $mBq \cdot m^{-2} \cdot s^{-1}$. The geometric mean and the median of the pre-construction radon flux density are both 73 $mBq \cdot m^{-2} \cdot s^{-1}$. This is similar to late dry season radon flux densities of 64±25 $mBq \cdot m^{-2} \cdot s^{-1}$, which were previously determined in non-mineralised areas in the region (Todd et al 1998, Lawrence et al 2009).

Radon flux densities measured immediately after stripping the top 20 cm of soil ranged from 57 to 919 $mBq \cdot m^{-2} \cdot s^{-1}$. The geometric mean and median are 350 and 385 $mBq \cdot m^{-2} \cdot s^{-1}$, respectively, much higher than radon flux densities prior to the area being disturbed. This difference may have been caused by differences in soil radium concentration, but may also be due to the heavy early wet season rain occurring during the sampling interval. Similar spikes in radon exhalation after short but heavy tropical rainfall events in the region have been reported elsewhere (Lawrence et al 2009).

Another reason for the higher radon flux densities observed after stripping may be that steady state conditions had not been reached when the measurements were conducted. Large radon concentration gradients at the soil/air interface immediately after stripping may have lead to an increase in radon flux densities (see Porstendörfer 1994 for background). It was not possible to repeat these measurements later to test the assumption of non-equilibrium owing to the covering of the area with waste rock. As a result of this uncertainty the radon flux densities measured after stripping are not deemed representative for the substrate at the sampling sites investigated. A pre-construction radon flux density of 73 mBq·m^{-2}·s^{-1} is thus assumed for the trial landform.

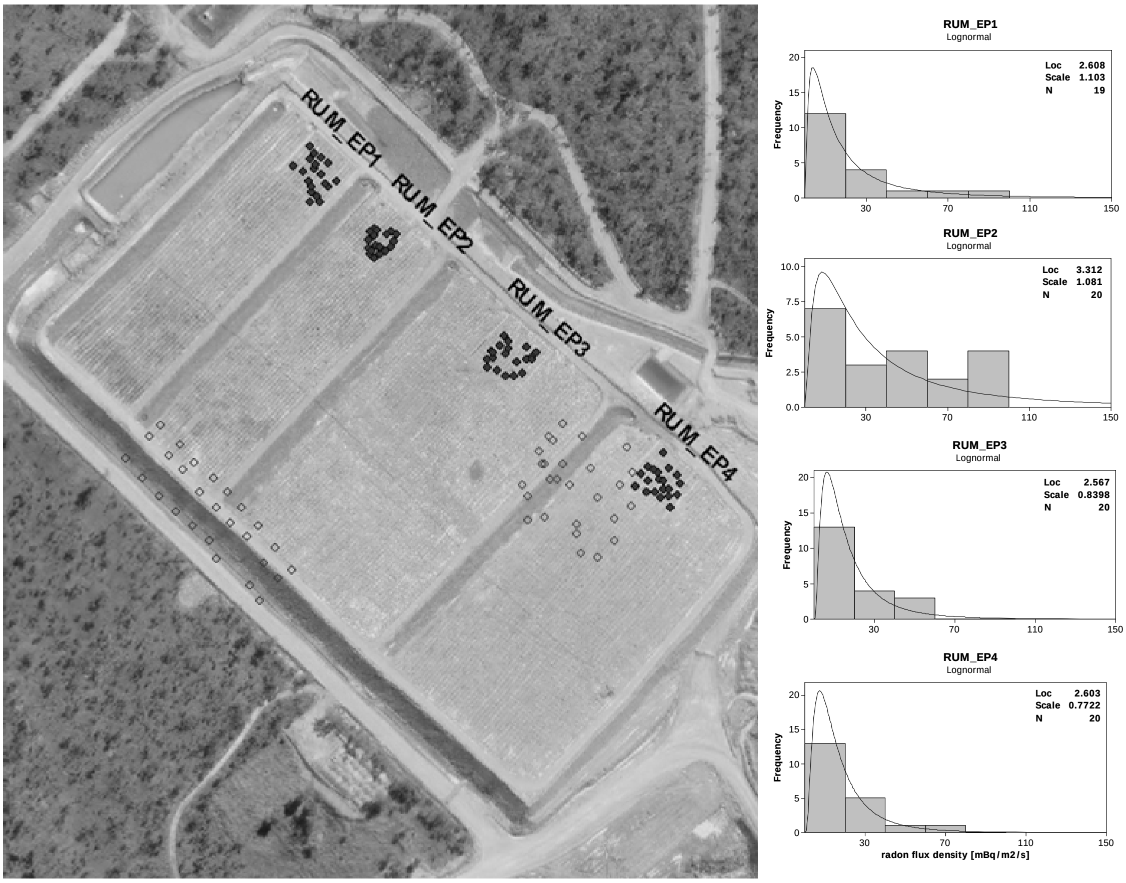

Figure 1 Locations of the radon exhalation measurements conducted in May 2009 (black diamonds). Open diamonds represent the location of the radon cups deployed in August and October 2008, respectively. In addition, the right hand side shows the distribution plots of radon flux densities measured at the four erosion plots (EP1–4).

Generally, radon flux densities measured from the four un-irrigated erosion plots are much lower than those measured on the ground prior to landform construction (Table 1). This can be explained by the rocky nature of the trial landform with typical rock sizes reaching up to 300 mm in diameter and larger. It has previously been reported that radon exhalation from fine grained soils is typically much larger than for solid rock (Lawrence et al 2009).

Radon exhalation from the 30% laterite–waste rock mix appears to be more uniform and exhibits a smaller standard error, reflecting the fact that the material in this area was mixed with a back hoe after dumping. In contrast, the waste rock only was dumped without any further mixing and consequently standard errors are larger. Radon exhalation from erosion plot 2 is approximately twice as high as the other three erosion plots, most likely due to heterogeneity in soil radionuclide activity concentrations across the trial landform. Soils from the four erosion plots were sampled for laboratory characterisation, and radionuclide activity concentration measurements were underway at the time of completion of this report.

Table 1 Description of the four erosion plots and average radon flux densities

	Treatment	^{222}Rn flux density [$mBq \cdot m^{-2} \cdot s^{-1}$] Arithmetic (*geometric*) average ± std error
RUM_EP1	Waste rock material planted with tube stock	22 (*14*) ± 11
RUM_EP2	Waste rock material planted by direct seeding	41 (*26*) ± 16
RUM_EP3	30% lateritic material mixed with waste rock (2 m), direct seeding	19 (*13*) ± 7
RUM_EP4	30% lateritic material mixed with waste rock (2 m), tube stock.	18 (*13*) ± 7

Future work

Radon exhalation surveys across the four erosion plots will be conducted every 4 months to investigate seasonal and temporal changes in radon exhalation from the trial landform. In addition, soil samples will be collected from the four erosion plots annually and radionuclide activity concentrations will be measured for a range of size fractions.

References

Akber R & Marten R 1992. Fate of radionuclides applied to soil in Ranger Uranium Mine land application area. In *Proceedings of the workshop on land application of effluent water from uranium mines in the Alligator Rivers Region*. Supervising Scientist for the Alligator Rivers Region, Australian Government Publishing Service, Canberra, 139–165.

Bollhöfer A, Storm J, Martin P & Tims S 2005. Geographic variability in radon exhalation at a rehabilitated uranium mine in the Northern Territory, Australia. *Environmental Monitoring and Assessment* 114, 313–330.

Graaf ER, Heijs S, Meijer RJd, Put L W & Mulder HFHM 1992. A facility to study transport of radon in soil under controlled conditions. *Radiation Protection Dosimetry* 45, 223–226.

Holdsworth S & Akber R 2004. Diffusion length and emanation coefficient of Rn-220 for a zircon and monazite sample. *Radiation Protection in Australia* 21, 7–13.

Lawrence CE 2004. Measurement of ^{222}Rn exhalation rates and ^{210}Pb deposition rates in a tropical environment. PhD thesis. School of Physical and Chemical Sciences, Queensland University of Technology, Brisbane.

Lawrence CE, Akber RA, Bollhöfer A & Martin P 2009. Radon-222 exhalation from open ground on and around a uranium mine in the wet-dry tropics. *Journal of Environmental Radioactivity* 100, 1–8.

Porstendörfer J 1994. Properties and behaviour of radon and thoron and their decay products in the air. *J. Aerosol Sci.* 2, 219–263.

Saynor MG, Evans KG & Lu P 2009. Erosion studies of the Ranger revegetation trial plot area. In ***eriss*** research summary 2007–2008. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 12–19.

Supervising Scientist 2007. *Annual Report 2006–2007*. Supervising Scientist, Darwin.

Todd R, Akber RA & Martin P 1998. ^{222}Rn and ^{220}Rn activity flux from the ground in the vicinity of Ranger Uranium Mine. Internal report 279, Supervising Scientist, Canberra. Unpublished paper.

Development of surface water quality closure criteria for Ranger billabongs using macroinvertebrate community data

C Humphrey, D Jones D & K Turner

Background and aims

This paper provides a status report on the development of surface water quality closure criteria (for operations and closure) for Ranger billabongs using macroinvertebrate community data. Specifically, the study aims to quantify macroinvertebrate community structure across a gradient of water quality disturbance in the ARR so as to provide a basis for surface water quality closure criteria for Georgetown and Coonjimba Billabongs.

After the Ranger mine ceases operations, disturbed areas need to be rehabilitated to a condition consistent with the values of Kakadu National Park (KNP), and to be suitable for reincorporation back into the Park. Ideally in the case of natural waterbodies, this would mean that their post-rehabilitation environmental values should be consistent with the expectations of the traditional owners of the land, and hence be consistent with those of similar, undisturbed habitats of KNP.

A concern with mine-site closure, is the potential for delivery of solutes from the rehabilitated mine landform. These solutes, if present at too high a loading, could affect water quality and hence impact on the ecological values of adjacent waterbodies. Hence a key objective of the closure planning process will be to produce a design for the current disturbed area such that the delivery of solutes and suspended sediment from the disturbed footprint in the mine site catchments (eg Corridor and Georgetown Creeks) will not compromise the post closure environmental objectives for the waterbody.

In this project, monitoring data derived from lentic macroinvertebrate communities are being used as one approach towards developing closure criteria for relevant water quality indicators in waterbodies immediately adjacent to the mine site. Georgetown Billabong (GTB), the largest waterbody located in close proximity to the current operational mine area, serves as an example of how closure criteria would be derived using macroinvertebrate data. The approach to deriving such criteria from local biological response data follows that outlined in the Australian and New Zealand Water Quality Guidelines (ANZECC & ARMCANZ 2000). Specifically, if the post-closure condition in Georgetown Billabong is to be consistent with similar undisturbed (reference) billabong environments of Kakadu, then the range of water quality data from the billabong over time that supports such an ecological condition in Georgetown Billabong (as measured by suitable surrogate biological indicators) may be used for this purpose.

Macroinvertebrates have been collected from mainly macrophyte (water column) habitat in GTB several times, including 1978 (pre-mining) , 1995, 1996 and 2006. On the first three of these occasions, the macroinvertebrate communities of this billabong consistently resembled those of reference waterbodies in the ARR. The environmental objectives post closure would be expected to embed the similar maintenance of this biological diversity (ie values consistent with those of similar, undisturbed habitats of KNP). However, in 2006, macroinvertebrates were collected from both macrophyte and benthic (sediment) habitats with the samples from each habitat being processed separately. The aggregated data showed essentially no difference to that of the reference billabongs and hence were in agreement with the results from the previous surveys. However, when the macrophyte and benthic data sets were analysed separately it was found that, in contrast to communities from the macrophyte habitat, the sediment-dwelling communities were less diverse in GTB than in reference waterbodies (Humphrey et al 2008).

In 2007–08, an investigation commenced to determine whether this lower diversity was mine or habitat-related, through studies of sediment chemistry and sediment physical structure. Sediment samples collected from littoral areas of the billabongs in 2007 showed levels of U (~42 mg/kg) higher than values measured in samples collected from references waterbodies at the same time, and higher than measured in GTB over the past 25 years by ERA's routine monitoring program and by others (Humphrey et al 2009). In particular there appeared to have been a substantial rise in U between 2001 and the present.

Because of (1) a substantial gap of 5 years in the collection of U data between 2001 and 2007, (2) differences in locations within GTB at which samples have been collected over time, and (3) differences in methods of chemical analysis, it was unclear whether there had actually been a recent substantial rise in U in the billabong as a whole. To address these questions, the following investigations were identified: (i) a number of GTB sediment samples collected in 2006 by EWLS should be reanalysed using similar methods to the 2007–08 study to establish whether any increase in U concentration was evident, and to quantify dependence of U extraction on method used, and (ii), sampling should be conducted in 2009 across a transect of the billabong to investigate U levels as a function of location across the billabong.

GTB sediments consist mainly of fine cracking clays, and are generally devoid of surface vegetation during the dry when the sediment exposed around the gently sloping margins undergoes desiccation-induced cracking. Of the billabongs sampled, these conditions are unique to Georgetown and Coonjimba Billabongs and suggest that the physical nature of the sediments may be an important factor contributing to the low diversity of benthic organisms. Indeed, and as reported in 'Effects of fine suspended sediment on billabong limnology', (pp 143–149, in this volume), Walker et al (1982) also noted the very fine particle size of suspended sediments in GTB relative to other billabongs studied.

To further investigate the nature of GTB sediments, the dependence of U concentration on particle size, as well as mineralogical characterisation, was also identified as a study requirement to be carried out for selected samples collected in 2007. The results of these investigations, together with a dedicated field assessment of U sediment toxicity proposed for a three-year period commencing in May 2009 (see 'The toxicity of uranium (U) to sediment biota of Magela Creek backflow billabong environments', pp 37–45, in this volume) would determine whether or not the low benthic diversity in GTB is related to current mining activities. If unrelated to mining activities, it would confirm the findings from the previous surveys, that macroinvertebrate communities in the billabong have remained unchanged since mining commenced, despite there having been an increase in solute loads between 1996 and 2006.

Pending the outcome of sediment-related studies described above and for now, the water quality record for 2006 will be included in the database being used to specify an acceptable ceiling water quality sufficient to protect the health of the waterbody post closure. This will enable the derivation of provisional water quality closure criteria for the water column. Of course derived numerical targets based on a limited range of measured water quality parameters should be accompanied by the caveat that periodic surveys of macroinvertebrate community will still need to be carried out to confirm that no other types of impacts from the rehabilitated landform were affecting the biological integrity of the waterbody. (Any new data arising from periodic sampling will also contribute to a rolling assessment and enable refinement of the water quality closure criteria.) The results from the U sediment toxicity study referred to above will also inform the setting of water quality criteria for GTB, ensuring criteria were not relaxed to thresholds that could adversely impact upon sediment quality, by virtue of the adsorption to sediment of U initially present in the water column.

Progress to date

There have been delays in aspects of the proposed work program for 2008–09 because of higher priorities in the environmental chemistry program in this period (including continuous monitoring) and because of new resources required to establish the related U-spiking sediment experiment in Gulungul Billabong in the second half of the reporting period. The following reports on progress against the study objectives for 2008–09:

1. Assess physical sediment characteristics of billabongs from samples collected in 2006:

   This work has been delayed. However, a penetrometer was acquired in the reporting period that will be used in subsequent dry seasons to measure the extent of compaction of littoral sediment (expected to be greatest in GTB and Coonjimba Billabongs compared with reference billabongs).

2. Carry out sediment digest method comparisons to determine if the historical data can be ‘normalised’ by sediment digest efficiency factors, thereby removing or reducing the ‘noise’ introduced by the use of different methods through time:

   Study component delayed.

3. Process and chemically analyse archived sediment samples collected from GTB in 2006:

   Samples acquired from EWLS but not yet processed.

4. Collect sediment samples from GTB in May 2009 to quantify the extent of cross billabong variation of metal concentrations:

   Samples collected, laboratory processed and submitted for chemical analysis.

5. Derive and report provisional water quality closure criteria for the water column:

   Reporting delayed but provisional criteria up to and including 2006 reported by Jones et al (2008).

References

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

Humphrey C, Turner K & Jones D 2009. Developing water quality closure criteria for Ranger billabongs using macroinvertebrate community data. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 130–135.

Jones D, Humphrey C, Iles M & van Dam R 2008. Deriving surface water quality closure criteria – an Australian uranium mine case study, In *Proceedings of Minewater and the Environment*, 10th International Mine Water Association Congress, eds N Rapantova & Z Hrkal, June 2–5, Karlovy Vary, Czech Republic, 209–212.

Walker TD & Tyler PA 1982. Chemical characteristics and nutrient status of billabongs of the Alligator Rivers Region, NT. Open file record 27, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Effects of fine suspended sediment on billabong limnology

D Buckle, C Humphrey & D Jones

Background and aims

This paper provides a status report for a study designed to progress the development of water quality criteria relevant to both mine operations and mine closure: the effects of fine suspended sediment on billabong limnology.

Specifically, the project aims to review, determine and infer the effects of fine suspended sediment on billabong limnology (physico-chemistry, primary production) using field water quality and biological effects data arising from both historic as well as new investigations.

The data arising from the study will be used as:

i a component of a multiple lines of evidence approach to assess direct effects of suspended sediment on tropical freshwater biota (complementing current laboratory assessment approaches on this subject ('The effects of suspended sediment on tropical freshwater biota', pp 32–36, in this volume).

ii assist with developing the suspended sediment/turbidity component of surface water quality closure criteria for billabongs adjacent to the Ranger mine site.

Suspended sediment has been identified as an aquatic ecosystem stressor that will most likely assume greater significance in the future, as a consequence of rehabilitation works on the Ranger site (see The effects of suspended sediment on tropical freshwater biota', pp 32–36, in this volume). Billabongs immediately downstream of Ranger, in particular, are at greatest risk from erosion of fine particulate matter from newly-rehabilitated landforms. Historical evidence for high suspended solids loadings and significant sedimentation that can arise in local billabongs, as a consequence of the failure of new earthwork structures, has been reported by the SSD and its consultants (Humphrey 1985, Nanson et al 1990) in relation to an intense rain-storm event that occurred over the Ranger site in February 1980 (affecting Coonjimba Billabong).

This project aims to draw upon field-effects and observational limnological data from (mainly) local billabongs from past and new studies to assist the development of operational water management triggers and closure criteria for suspended sediment. The experimental focus of the investigations will be the tracking of natural increases in turbidity that are observed in shallow backflow billabongs adjacent to Ranger over the dry season and the changes in billabong limnology that are associated with this.

Turbidity may disrupt a number of ecosystem functions, one of the best-documented of these being primary (plant) production through inhibition of light. By continuous measurement of dissolved oxygen and turbidity, together with regular (grab-sample) measurement of suspended sediment (0.45–63 µm), chlorophyll-a (phytoplankton abundance) and total organic carbon (produced by decay of plant matter and contributing to biological oxygen demand), a threshold in turbidity may be inferred at which important billabong photosynthetic functions are disrupted. This threshold may provide complementary data (to those derived from laboratory studies) upon which to derive operational and closure criteria for suspended sediment in local billabongs.

For this study, data from previous limnological studies conducted in the region are being reviewed to examine possible relationships between turbidity, chlorophyll-a and dissolved oxygen. A field investigation is also being conducted in Georgetown Billabong using in-situ continuous (5 minute intervals) measurement of dissolved oxygen and turbidity, near-surface and at depth, for the period May to October/November 2009 (encompassing the period of significant dry season increase in turbidity in the billabong). Depth profiles of water quality parameters, and surface and depth samples for chlorophyll-a and total organic carbon determinations, have been collected fortnightly over this period.

This project links to laboratory-based concentration-response experiments that are being developed for suspended sediment (Ecotoxicology group) as well as to relevant activities of the Hydrological & Geomorphic Processes (HGP) group so as to ensure maximum relevance of outputs/outcomes.

Methods

Sampling commenced on 27 May 2009 from one sampling site in the deeper waters located near the outflow of Georgetown billabong (S 12^{0}40.705', E 132^{0}55.885') (Map 2). At this site, three sub-sites were established for continuous and spot sampling by use of a floating pontoon secured in place to prevent movement and disturbance of sub-sites. Sub-site 1 was directly attached to the pontoon to measure surface water conditions (10-20 cm depth), while sub-sites 2 and 3 were located approximately 10 m apart near the bottom of the water column. They were fixed $\approx$ 20 cm above the sediment bed using custom designed (tripod) frames to both secure in place and protect (from crocodiles) the sampling and monitoring equipment. Each sub-site was equipped with a Hydrolab datasonde 5X multiprobe to measure temperature, dissolved oxygen, turbidity, EC and pH at five minute intervals. Data were downloaded frequently via mobile phone connection to track changes in the measured parameters, and to detect any equipment malfunction.

More detailed sampling was conducted each fortnight between 8:30 am – 9:30 am to standardise with the same time of day that sampling was conducted in baseline studies conducted in 1980 and 1981 (eg Humphrey & Simpson 1985) and in subsequent routine water quality monitoring carried out by ERA. Water samples, from all sub-sites, were collected each fortnight for measurements of suspended sediment, total (TOC) and dissolved (DOC) organic carbon and chlorophyll-a. Filtered water samples were obtained monthly from all sub-sites for analysis of metals (Al, Cu, Fe, Mn, Pb, U, Zn), ions (Ca, Mg, SO_4) and nutrients (NO_3, PO_4, NH_3).

An additional mid-depth (1.0-1.25 m deep) sample of chlorophyll-a was collected below sub-site 1. This mid-depth sample aimed to provide information on phytoplankton biomass at different depths.

The fortnightly visits also included collection of water quality profile data near sub-sites 2 and 3. Measurements were made at 10 cm below the surface, 25 cm, and then every 25 cm until the bottom of the billabong was reached using pre-measured cable to lower the measuring probes (depth was 2.65 m on 27/5/2009). Parameters measured included, temperature, dissolved oxygen, turbidity, EC and pH using a hydrolab quanta multiprobe and photosynthetically-active radiation (PAR) using a LI-193 underwater Spherical Quantum Sensor. The LI-193 measures photon flux from all directions or Photosynthetic Photon Flux Fluence Rate (PPFFR) and quantifies the amount of light available for photosynthesis.

Results to date

Data are still being collected for this project, thus results presented below are preliminary and subject to change with additional information.

Changes in turbidity over time

Historical turbidity data collected in 1980 and 1981 by Humphrey and Simpson (1985) and between 1982 and 2008 by Energy Resources of Australia (ERA), as well as data collected during the current project, are graphed for each month in Figure 1. The historical data indicate that turbidity during 1980 and 1981 increased earlier in the dry season and with subsequent years increased only towards the very end of the dry season. The increased turbidity early in the dry season in 1980 and 1981 is presumably due to the presence of water buffalo. Water buffalo were mostly removed by 1982 resulting in reduced turbidity early in the dry season. Increased turbidity late in the dry season since the removal of buffalo, is most likely due to the reduction in water levels to quite shallow depths (eg depth reduced from 2.65 m on 27/5/09 to 1.68 m on 29/10/09 in the deepest section of the billabong) coupled with wind or biogenic re-suspension of fine sediments. Whilst the magnitude of wet season rainfall and duration could also cause variation in turbidity events from year to year, it is unlikely to account for the early elevation of turbidity observed in 1980 and 1981. Both these years (1979–80 and 1980–81) recorded above average rainfall, with subsequent years representing both below and above average rainfall (see Figure 2.1 p 10 in Supervising Scientist 2009).

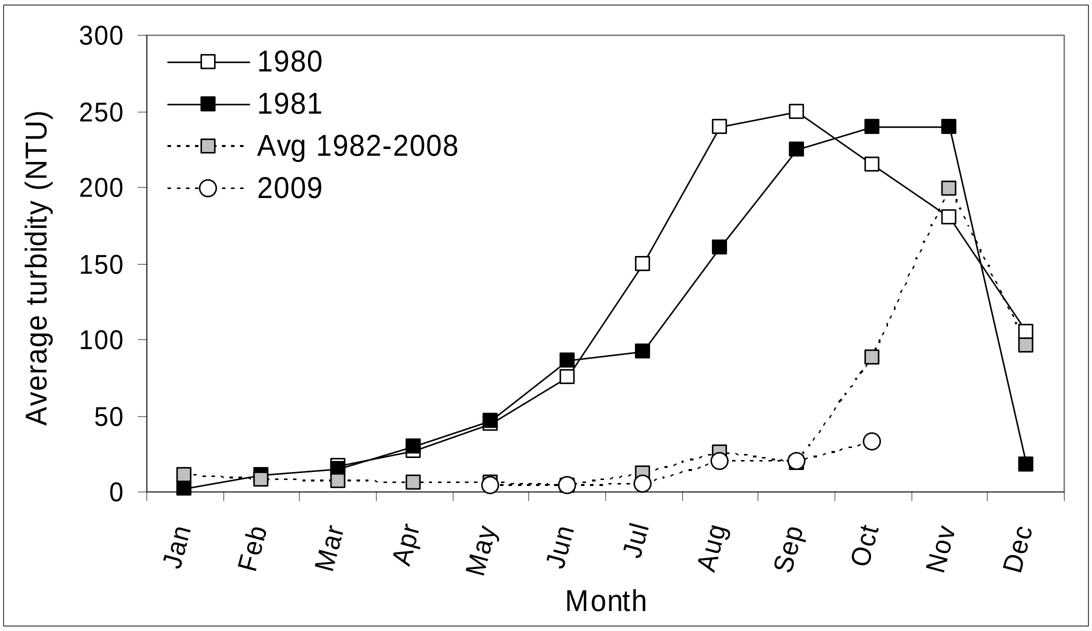

Figure 1 Average monthly Turbidity (NTU) for Georgetown Billabong from 1980 to 2009. ***eriss*** data used for 1980, 1981 and 2009. Energy Resources of Australia data used for 1982–2008.

The data collected in 2009 are consistent with the seasonal pattern of post-buffalo turbidity in Georgetown Billabong. The mean value recorded in October 2009 is below the average for 1982 to 2008 (Figure 1); however, and while data for individual years are not shown, the October mean value is consistent with values for a number of individual years since the removal of water buffalo.

The shorter time period of enhanced turbidity in Georgetown Billabong since the early 1980s (restricted now to the late dry) may, in part, be responsible for the increase in density of aquatic vegetation observed since the removal of water buffalo (Finlayson et al 1994). This suggestion is supported by Dunlop et al (2005) who reported the absence of submerged aquatic vegetation in the Condamine Balonne River (Murray-Darling catchment) when turbidity was greater than 20-30 NTU.

The changes in water clarity and billabong aquatic plant communities discussed above make a number of the findings from the earlier detailed limnological studies (1978–1980) such as those of Walker and Tyler (1982), Walker et al (1982), Walker & Tyler (1983) and Walker et al (1984), not so applicable to early dry season months under current conditions.

Suspended sediment/turbidity relationship

Data obtained during the 2009 dry season have been used to derive a preliminary relationship between turbidity and suspended sediment (SS) in Georgetown Billabong. There is a strong relationship between SS and turbidity ($R^2 = 0.937$), as shown in Figure 2. However, the line of best fit is not quite linear, most likely due to the >0.45 µm filter not capturing ultra fine particles, thus underestimating the total weight of SS. In support of this, Walker et al (1982) found a 0.1 µm filter was required to provide maximum recovery of SS in Georgetown Billabong due to the increased presence of very fine particles relative to other billabongs studied. Future and stored <0.45 µm samples will be further filtered through 0.1 µm filter papers to better understand the potential influence of ultra fine particles in the current SS sample estimates.

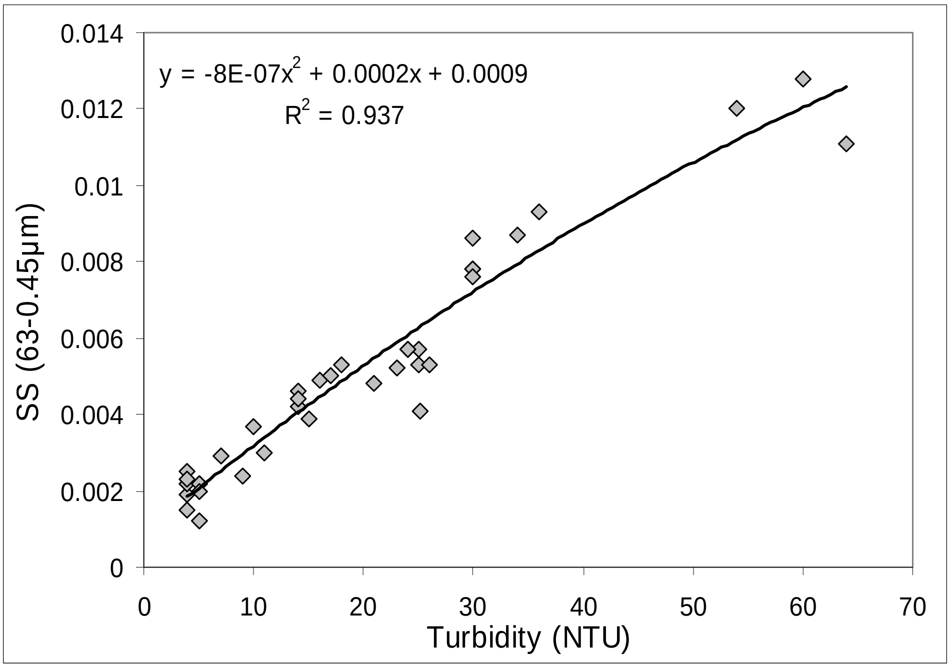

Figure 2 Suspended Sediment (0.45–63 µm) relationship to turbidity (NTU) in Georgetown Billabong during the dry season of 2009

Alternatively or in addition, the slightly non-linear relationship between SS and turbidity could be associated with an increase in the contribution of organic matter (algae etc) at higher turbidities, since organic matter is lighter than inorganic matter.

Turbidity influence over phytoplanktonic productivity

Chlorophyll-a (mg/m^3) is commonly used as a direct measure of phytoplanktonic biomass. For this project it was measured close to the water surface (10–20 cm depth), middle water column and 20 cm above the sediment bed. at fixed sampling locations. Only surface water data are presented here, data from other depths will be analysed at the completion of the project.

Surface chlorophyll-a steadily increased after the sampling program commenced with a peak observed on the 20/8/2009 at 35 mg/m^3 (24 NTU), followed by slightly reduced concentrations, then a second larger peak on the 15/10/2009 (72 & 65 mg/m^3, duplicate samples at 30 NTU). The latter sharp rise in chlorophyll-a followed a short-duration spike in turbidity (54 NTU) recorded on the 1/10/2009. The short duration of the turbidity spike prevented any assessment of the effects on chlorophyll-a over a prolonged period of reduced light availability (Figure 3), but rather may have provided a stimulus for algal growth by way of additional (suspended) nutrients.

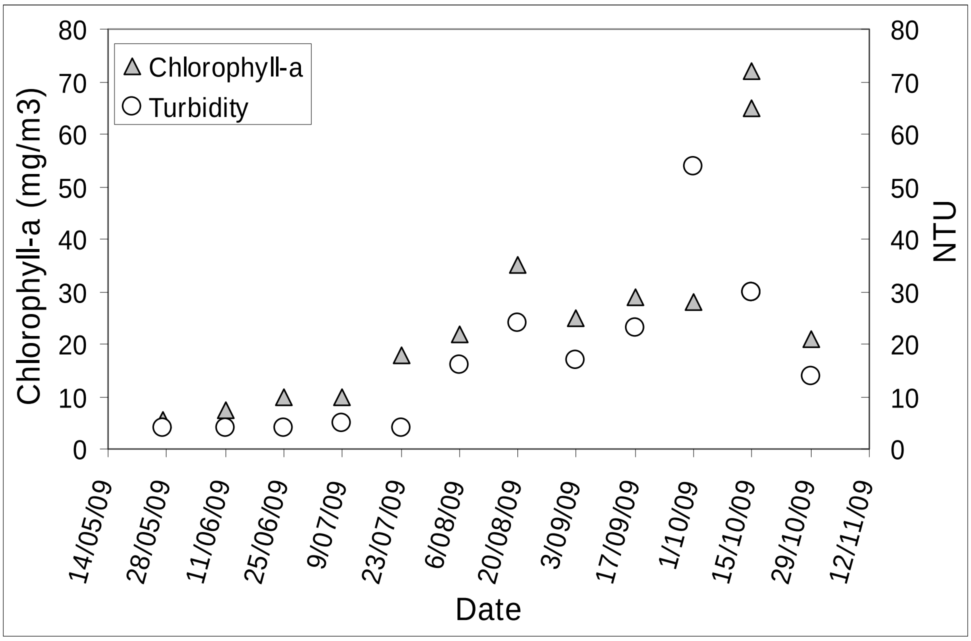

Figure 3 Turbidity (NTU) and chlorophyll-a (mg/m^3) in the surface waters (< 20 cm) of Georgetown Billabong in 2009

The (skewed) unimodal relationship observed between near-surface chlorophyll-a and turbidity from data collected in 1981 by Humphrey and Simpson (1985) suggest inhibited phytoplankton production at turbidity values around 50 NTU (Figure 4). While a similar relationship is less evident in 2009 due to the low turbidity conditions observed in this year (Figure 1) (only one turbidity value recorded above 30 NTU, Figure 3), the data collected in 2009 nevertheless appear to support the historical (1981) relationship and threshold effect. Given the lack of sampling events in 2009 for turbidity values above 30 NTU and the changed turbidity conditions observed since 1981 (Figure 1), particularly the seasonal timing, further chlorophyll-a and turbidity data are required for periods when turbidity is above 30 NTU to better define the relationship.

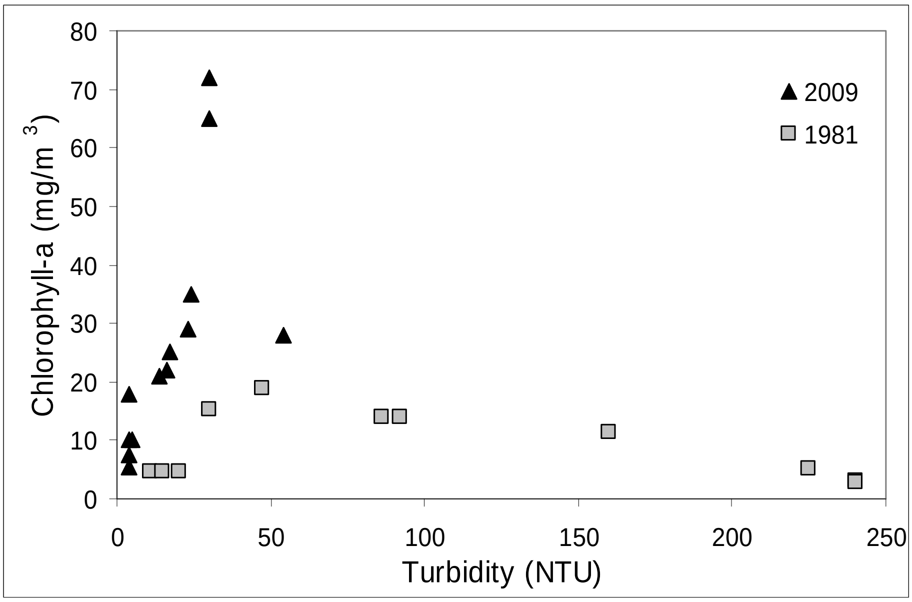

Figure 4 Relationship between turbidity (NTU) and chlorophyll-a (mg/m^3) in the surface waters (10–20 cm) of Georgetown Billabong in 1981 and 2009

Further work

Data presented above are a subset of the full dataset collected. Analysis of the full dataset is yet to be completed and as such, the inferences that have been presented above are preliminary.

Preliminary recommendations

Further samples from Georgetown Billabong are required for turbidity conditions above 30 NTU to better define the relationship between turbidity and chlorophyll-a concentrations.

If an improved understanding of the slightly non-linear relationship between suspended sediment and turbidity is required, then the following analysis on any future samples is recommended:

- The residue from future 0.45–63 µm samples should be analysed for turbidity then further filtered using 0.1 µm filters to determine if the loss of ultra fine particles changes with increasing turbidity.
- Assessment of the proportion of organic and inorganic material contributing to the suspended sediment should be further considered. This information may also provide insights into the effect of algal self shading.

References

Dunlop J, McGregor G & Horrigan N 2005. Potential impacts of salinity and turbidity in riverine ecosystems: characterisation of impacts and a discussion of regional target setting for regional ecosystems in Queensland. National Action Plan for Salinity and Water Quality, Water Quality State Level Investment Project 6. [web] http://www.wqonline.info/Documents/Report_WQ06_Review_reduced.pdf.

Finlayson CM, Thompson K, von Oertzen I & Cowie ID 1994. *Vegetation communities of five Magela Creek billabongs, Alligator Rivers Region, Northern Territory*. Technical memorandum 46, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra.

Humphrey CL 1985. Recent history of Coonjimba Billabong. In *Alligator Rivers Region Research Institute Annual Research Summary 1984–85*. Supervising Scientist for the Alligator Rivers Region, Australian Government Publishing Service Canberra 1985, 48–49.

Humphrey CL & Simpson RD 1985. The biology and ecology of *Velesunio angasi* (Bivalvia: Hydiidae) in the Magela Creek, Northern Territory (4 parts). Open file record 38, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Nanson GC, East TJ, Roberts RG, Clark RL & Murray AS 1990. Quaternary evolution and landform stability of Magela Creek catchment, near the Ranger Uranium Mine, northern Australia. Open file record 63, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Supervising Scientist 2009. *Annual Report 2008–2009*. Supervising Scientist, Darwin.

Walker TD, Kirk JTO & Tyler PA 1982. The underwater light climate of billabongs of the Alligator Rivers Region, NT. Open file record 20, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Walker TD & Tyler PA 1982. Chemical characteristics and nutrient status of billabongs of the Alligator Rivers Region, NT. Open file record 27, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Walker TD & Tyler PA 1983. Primary productivity of phytoplankton in billabongs of the Alligator Rivers Region. Open file record 8, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Walker TD, Waterhouse J & Tyler PA 1984. Thermal stratification and the distribution of dissolved oxygen in billabongs of the Alligator Rivers Region, NT. Open file record 28, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

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

C Humphrey & G Fox

Background and aims

This study aims to classify vegetation communities from a variety of potential analogue sites in the ARR, model these communities in relation to environmental factors, then apply the model as a tool for assessing the success of revegetation of the post-mine landform at Ranger.

Characterisation of plant communities from appropriate natural analogue sites will assist in selection of species for revegetation of the Ranger mine landform following decommissioning of the site. The characteristics of these communities will also assist in developing numerical targets for performance measures against which the success of the revegetation can be tracked. For the range of key vegetation community types that represent the spectrum of environments likely to be found across the rehabilitated footprint, relationships between the occurrence of such communities and key geomorphic features (parent material, slope, effective soil depth etc) of the landscape need to be identified. By identifying the key environmental features that are associated with particular vegetation community types, either (i) the conditions required to support these communities or, alternatively, (ii) the community types that best suit particular environmental conditions, may then be specified for the rehabilitated landform at Ranger.

Data previously obtained by ***eriss*** and Earth Water Life Sciences (EWLS) on plant communities in the ARR, including data from reference and analogue sites, have been used to identify and characterise target communities that will provide a basis for rehabilitation and subsequent post-closure monitoring. The collective data from analogue/reference sites are being used to derive plant-environment relationships with the aim of developing predictive models based upon physical and chemical input variables.

For the 2006–07 and 2007–08 periods, the study focus was on additional characterisation of target plant communities (especially *Melaleuca* communities), gathering of soil physico-chemistry data for analogue sites and modelling (by EWLS) of key plant-landform relationships. The data sets produced by ***eriss*** and EWLS were combined and analysed using multivariate techniques. A preliminary plant classification was derived, with key distinguishing species identified for the main classification groups. The occurrence of these key species were modelled according to major landform characteristics.

The modelling conducted by EWLS did not include the full suite of soil physical and chemical characteristics that had been collected for (a) the analogue sites, nor (b) the soil/substrate media occurring on previous revegetation test sites on Ranger waste rock. It did not, most importantly, include the characteristics of the type of substrate (mixture of waste rock and finer-grained weathered horizon material) being proposed in the rehabilitation plan for the site. The principal aim of future work on the vegetation analogue project is to include in the development of a predictive model, all available environmental (soil and landform) and vegetation data for both analogue and Ranger revegetated sites.

In the first part of 2008–09, it became apparent that it might not be possible to source the complete set of landform variables (climate and water balance, local topography, as well as fire disturbance) that had been used in the previous modelling of the analogue sites by former EWLS staff. Instead and as an interim analysis procedure until the complete environmental dataset could be acquired, analyses conducted during this period focused on identifying possible soil property-plant relationships for the natural analogue sites. However, at best, only weak relationships were found between soil properties and vegetation communities, indicating that physical landscape and landform features were likely to be more critical determinants of community type (Humphrey et al 2009).

Progress

Since ARRTC 23 (March 2009), the following issues have been raised and/or addressed:

1. Additional plant survey data became available as a consequence of studies conducted to assess the impact of Cyclone Monica on the vegetation of impacted areas in the ARR (Staben et al 2009). The opportunity, therefore, was taken to add the data for an additional 55 sites (including sites from Gulungul Creek catchment, Nabarlek and Ranger minesite revegetation trials (‘Heritage’ area)) to the existing combined ***eriss*** and EWLS ARR plant classification to both further validate the integrity of the original classification, and seek additional classification groups that might need to be considered for the final landform design.
2. A large number of soil samples were collected by EWLS in 2009 from the newly-constructed trial landform at Ranger and from existing, constructed (mine-derived) substrates that have provided a medium for vegetation growth on various historical rehabilitation trial sites on the Ranger mine site.
3. As stated above, only weak relationships between soil properties and plant communities for the natural analogue sites have been found. However, a published popular account by a long-term naturalist working in the ARR (Woerle 1987) makes specific reference to deep and shallow soils characteristic of the mixed eucalypt woodland and dry mixed eucalypt woodland classification units, respectively, identified in earlier analogue work (Table 1). K Brennan (NT Government plant ecologist, pers comm) also considered that gross soil profile features similar to those described by Woerle (1987) should be able to distinguish the same two major classification groups.
4. Potential problems have been identified with the landscape physical characterisation variables (eg slope angle and length) produced from past vegetation and landscape modelling conducted by EWLS (Hollingsworth et al 2007, Hollingsworth & McGovern 2004). This work used a synthesised Digital Elevation Model (DEM), the source components of which are being reassessed for accuracy and applicability by ERISS. The results from the original analysis both informed selection of an analogue landform similar in size and shape to a conceptual model of the Ranger final landform, but perhaps more importantly, were used to specify the physical design criteria for the rehabilitated landform.

The following reports on progress against the four issues identified above.

1 Additional plant survey data

After standardising survey data to the same areal units used in earlier surveys, the additional plant survey data for trees and shrubs arising from Staben et al’s (2009) study were added to the existing ***eriss*** and EWLS plant analogue data. Group average cluster analysis was conducted on the combined data using the PRIMER (v6) multivariate software package (Clarke & Gorley 2006). The new (re-)classification is shown in Figure 1 where site labels indicate the original three dominant vegetation classification classes (C1–C3, described in Table 1). All of the original C1–C3 sites reclassify according to their original vegetation classes. Further, additional site data are now available for the *Melaleuca* woodland classification group (C1), formerly with very few representative sites. Despite the data of Staben et al's (2009) being confined essentially to Koolpinyah lowlands (with no hill slopes representing classification C3 sites), their geographical extent is considerable. Incorporation of the additional data in the classification analysis has further confirmed the discrete and dominant vegetation units identified in previous analogue work (Table 1).

Table 1 Descriptions of the analogue communities identified in this study and, where available, the matching vegetation units according to Schodde et al (1987)

Broad vegetation community	Dominant and/or distinguishing tree or shrub species	Classification unit from this study (Fig 1A)	Vegetation units used by Schodde et al (1987)
Melaleuca woodland	*Melaleuca viridiflora* *Pandanus spiralis*	C1	Myrtle–Pandanus savannah
Mixed Eucalypt woodland	*Eucalyptus miniata* *Eucalyptus tetrodonta* *Corymbia porrecta* *Xanthostemon paradoxus* *Acacia mimula*	C2	Open forest
Dry mixed Eucalypt woodland	*Corymbia foelscheana* *Xanthostemon paradoxus* *Erythrophleum chlorostachys* *Eucalyptus tectifica* *Cochlospermum fraseri*	C3	Woodland

2 Additional mine-derived soil samples

These samples, from the trial landform and other minesite revegetation plots, were collected by EWLS and have been submitted for analysis of the same suite of soil quality variables used by Humphrey et al (2009) in their assessment of the influence of soil properties on analogue vegetation patterns. The data were not available at the time of writing of this report.

3 Potential soil-vegetation relationships

Anecdotal evidence suggests that soil properties should be able to be used to distinguish the three dominant vegetation communities in the analogue classification, despite general lack of correlation with the parameters that have been assessed to date. Further evaluation of the Georgetown analogue area will be conducted in 2009–10 to assess whether, by restricting the comparison with soil properties to an area where all three vegetation classification units are represented, that other sources of variation that may have confounded previous comparisons can be removed, Moreover, additional multivariate techniques will be used to explore potential soil-plant relationships not identified in the previous (PRIMER BIOENV-based) analyses.

4 Derivation of landscape variables used in past vegetation and landscape reconstruction modelling

Given the comparatively low relief of most of the analogue sites, concerns have been raised by ***eriss***'s SSDI group (J Lowry, pers comm) that the 50-metre DEM created of the analogue areas, and used previously to derive a range of key environmental topographic variables, may have been deficient in resolution and may not fully support the results that were derived; thus, current modelling may not be as accurate as it could be. It is proposed that a full suite of landscape variables be re-derived for inclusion in the modelling using an updated and validated DEM with appropriate resolution.

Summary of further studies required for 2009–10

A much more expanded data set than that used by EWLS will be used to investigate relationships between plant and environmental data. In particular, the expanded modelling will include current soil description data, possible additional data acquired as a consequence of issue 3 from above, as well as those from the newly-constructed trial landform, and from existing, constructed (mine-derived) substrates that have provided a medium for vegetation growth on various historical rehabilitation trial sites on the Ranger mine site. As noted above (issue 4), a full suite of landscape variables will also be re-derived for inclusion in the modelling using an updated DEM. Collectively, these studies should provide more robust identification and characterisation of target plant communities to provide the basis for specification of closure criteria and for design of post-rehabilitation monitoring programs.

References

Clarke KR & Gorley RN 2006. *Primer v6: User Manual/Tutorial, Primer E: Plymouth*. Plymouth Marine Laboratory, Plymouth, UK.

Hollingsworth ID, Humphrey C & Gardiner M 2007. Revegetation at Ranger: An analysis of vegetation types and environmental trends in analogue areas. EWL Sciences Pty Ltd. Darwin.

Hollingsworth ID & McGovern E 2004. Landscape reconstruction at Ranger Mine – approach and current status. Discussion Paper ARRTC Meeting, March 2004. EWL Sciences Pty Ltd.

Humphrey C, Fox G & Lu P 2009. Use of vegetation analogues to guide planning for rehabilitation of the Ranger minesite. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 136–146.

Schodde R, Hedley AB, Mason IJ & Martensz PN 1987. Vegetation habitats, Kakadu National Park, Alligator Rivers Region, Northern Territory, Australia. Final report to Australian National Parks and Wildlife Service, CSIRO Division of Wildlife and Rangelands Research, Canberra.

Staben G, Saynor MJ, Moliere DR, Hancock GR & Evans KG 2009. 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. In ***eriss*** *research summary 2007–2008*. eds Jones DR & Webb A, Supervising Scientist Report 200, Supervising Scientist, Darwin NT, 12–19

Woerle F 1987. Ranger's territory: Adventure in Australia's far north, the story of Frank Woerle as told to Colin Thiele. Angus & Robertson, North Ryde, New South Wales.

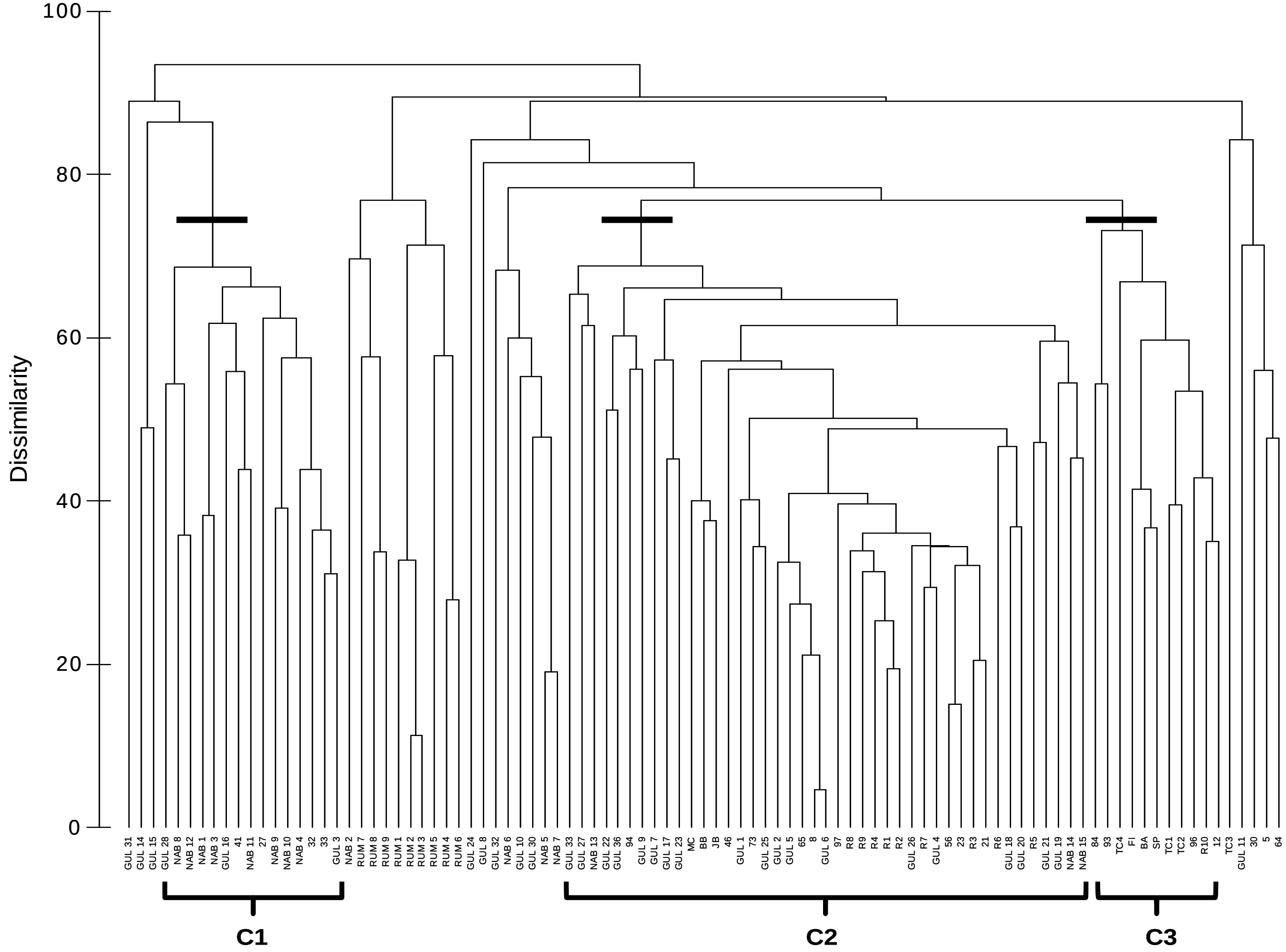

Figure 1 Cluster analysis (group average linkage) of trees and shrubs data for Alligator Rivers Region (ARR) vegetation analogue sites. (Vegetation data log transformed density/hectare units.) *Key to site codes*: Site label suffix (C1-C3) = original classification class for the site (Humphrey et al 2006, 2007; see Table 1). Numbered-only sites = EWLS Georgetown analogue area; 'R' sites = lowland Koolpinyah sites around Ranger; hill sites,'TC' (Tin Camp Creek) and 'F1' (Fisher) = schist hills; JB, BB, BA, SP = sandstone hills; MC = quartzite hill; 'GUL' sites = Gulungul catchment; 'NAB' sites = Nabarlek; RUM = Ranger minesite revegetation trials ('Heritage' area)

Charles Darwin University seed biology research

S Bellairs[1], M McDowell[1], C Humphrey, M Daws[2] & P Christophersen[3]

Introduction

Charles Darwin University (CDU) staff are undertaking seed biology research to optimise germination of local native species to support the rehabilitation of the Ranger mine site. The project involves collaboration between the CDU researchers and staff from ***eriss***, Energy Resources of Australia (ERA), Kakadu Native Plant Suppliers (KNPS), Earth Water Life Sciences (EWLS), Greening Australia and Top End Seeds (TES).

ERA are required to sustainably establish a range of local native species on rehabilitation areas at the Ranger mine site. According to the applicable Environmental Requirements for rehabilitation, the company is required to establish an environment similar to the adjacent areas of Kakadu National Park, using local native plant species similar in density and abundance to those existing in adjacent areas of the Park.

To rehabilitate the areas impacted by the mine footprint, large numbers of plants comprising a broad range of species will be required. Therefore, effective techniques will be needed to source, store and germinate seeds, whether for direct seeding or for production of tube stock. KNPS are both providing seeds and producing tube stock for current rehabilitation areas using nursery facilities in Jabiru. They are collecting seeds from the local area to produce native plants that are adapted to local conditions.

Most Australian species germinate poorly from seeds unless specific seed treatments are applied since they have seed dormancy mechanisms that prevent or delay germination, except in response to specific cues. However, pre-treatment information is lacking for the vast majority of NT species. Very little information is known about the seed biology of the local (ARR) species, including how to optimise viability of seeds during collection, how to store the seeds, or how to overcome dormancy and germinate the seeds (Bellairs 2007, Bellairs & Ashwath 2007). Tropical flora species are likely to differ in their seed biology responses to environmental cues compared to those of other Australian flora. Therefore, although information from southern Australian studies can be used as a guide, for most species results obtained for similar southern species are unlikely to be directly applicable. KNPS is also identifying species that are difficult to germinate as part of their nursery operations.

The aim of the project has been to investigate seed collection, viability, germination, dormancy and storage for up to 50 species that occur on the Ranger mine lease that have been identified as potentially important for rehabilitation. The project aims to develop protocols for effective seed storage and germination.

1 School of Environmental & Life Sciences, Charles Darwin University, NT 0909.
2 Earth Water Life Sciences, GPO Box 518, Darwin, NT 0801
3 Kakadu Native Plant Suppliers, PO Box 319, Jabiru NT 0886

Approach

Seed lots are being supplied by KNPS, or from Top End Seeds and Greening Australia when KNPS is unable to provide supplies of seeds. While most seed lots have been obtained from KNPS, factors such as unusual rainfall patterns and cyclones have prevented KNPS from supplying seeds during some seasons and so alternate supplies have been obtained. It is also becoming evident that collection of seeds in sufficient quantities for many of the species is more difficult than originally anticipated. TES collection data indicate that seeds for some species has not been able to be collected in sufficient quantities to make up a seed lot for seven years.

Testing of seeds is being carried out by CDU research staff under standard conditions in laboratory incubators to control for any light, temperature and moisture variations to which seed responses are highly susceptible. In this manner, the most effective treatments may be accurately identified without having to use large quantities of seeds.

Testing is based on the International Seed Testing Association guidelines and methodologies, with modifications to enable fewer seeds to be utilised. The methodologies used by the Australian Millennium Seed Bank Projects and other published studies are both guiding treatment selection and being used so that results can be compared with findings elsewhere.

Factors being tested include seed viability following collection, the types of dormancy mechanism(s) present, effective treatments to overcome dormancy and seed longevity under various storage conditions.

In 2008–09 an undergraduate student funded by the CDU UTROP research experience scheme investigated dormancy of *Persoonia falcata*.

Progress to date

CDU propagation trials

The seed biology project commenced in July 2006. At a meeting held at Jabiru with ERA, ERISS, EWLS, KNPS and CDU project staff in that year, fifty priority species were chosen based on (i) their abundance in the Georgetown analogue sites (data provided by EWLS), (ii) their difficulty in propagation (information provided by Peter Christophersen (KNPS)), and (iii) preference for perennial species not likely to create a fire risk when established on the rehabilitation areas. This list was reviewed in December 2008 and modified slightly in early 2009, omitting those species for which sufficient seeds were unlikely to be able to be collected, thus reducing the priority list to 37 species. A major review and re-prioritisation were also carried out after the July 2009 project meeting (see section 'Re-prioritisation of species for propagation' below).

Seed lots of 26 of the original 50 priority species and an additional three species from the revised (early 2009) priority list have been received by CDU. Project progress on those species is summarised in Table 1. KNPS supplied five seed lots in August 2006, two in October 2006, four in January 2007, fourteen in July 2007, five in August 2009 and one in September 2009, including some additional seed lots of previously-supplied species. Thirty-two seed lots have been supplied by TES or Greening Australia NT between November 2006 and September 2008 and an additional six seed lots have been collected by CDU. As well as the priority species, some testing has occurred for 21 other species (Table 2). Where CDU has not been able to source seed lots of the priority species, local seeds of other species in the same genus have been obtained and tested. In some cases, other species that also occur on the

Ranger mine lease have been tested for student projects or to provide a more detailed assessment of germination and dormancy trends.

The second annual report on the viability, germination and dormancy present in thirty species was provided to project sponsors in October 2008 (Bellairs & McDowell 2008). Additional species to those included in the 2007 report were: *Cymbopogon* sp, *Ectrosia* sp, *Eragrostis* sp, *Eriachne glauca*, *Eriachne schultziana*, *Eriachne triseta*, *Eulalia* sp, *Fimbristylis* sp, *Gomphrena canescens*, *Haemodorum coccineum*, *Livistona inermis*, *Schizachyrium fragile* and *Spermacoce stenophylla* (species identification to be confirmed). Test procedures have been developed for these species and initial viability, germination and dormancy testing has been conducted.

In addition, the germination and viability of seeds lots to be used in the trial landform plot were tested. Seventeen species had germination tests carried out and fifteen species had viability tests carried out. EWLS have developed an Access Database to store the seed biology data.

Table 1 Summary of progress investigating seed biology of the priority species

Species	# Lots received	Weight / Number	Viability	Imbibition	Germination	
Alloteropsis semialata	2	C	C	C,N	C	1
Aristida inaequiglumis	2	C	C,N	C,N	C	2
Brachychiton diversifolius	1	C	C	C	C	3
Brachychiton megaphyllus	1	C	C	C	C	4
Buchanania obovata	2	C	C	I,C	C	5
Chrysopogon fallax	2	C	C	C,N	C	6
Clerodendrum floribundum	1	C	C	N	P	7
Denhamia obscura	3	C	C	C,C,N	C,C,P	8
Eragrostis sp TBI	1	C	C	C	C	9
Eriachne obtusa	2	C	C	C,N	C	10
Eriachne schultziana	1	C	C	C	C	11
Eriachne triseta	1	C	C	N	C	12
Gomphrena spp TBI	3	C	C	N	C,C,P	13
Haemodorum coccineum	2	C	C	I,N	C,P	14
Heteropogon triticeus	1	C	N	N	C	15
Livistona humilis	2	C	C	C	P	16
Livistona inermis	2	C	C	C	P	17
Owenia vernicosa	5	C,C,C,P,P	I,C,C,N,N	I,N,N,N,N	I,N,N,N,N	18
Persoonia falcata	2	C	C	C	C	19
Petalostigma pubescens	1	C	C	N	P	20
Petalostigma quadriloculare	1	C	P	N	P	21
Schizachyrium fragile	1	C	C	C	C	22
Setaria apiculata	1	C	C	C	C	23
Setaria sp TBI	1	C	C	N	C	24
Spermacoce sp1 TBI	2	C	C	N	C,P	25
Spermacoce sp2 TBI	1	C	C	N	P	26
Terminalia carpentariae	3	C	C	C,I,N	C,C,P	27
Terminalia ferdinandiana	4	C	C,C,N,N	C,N,C,N	C,P,P,N	28
Terminalia pterocarya	1	C	C	N	N	29
Verticordia cunninghamii	1	C	C	C	C	30

C Completed; P In progress; N Not started, I insufficient seeds.

Table 2 Investigations of species that are related to the priority species or other species that occur on the Ranger area, including student projects

Species	# Lots received	Weight / Number	Viability	Imbibition	Germination
Chrysopogon latifolius	1	C	P	N	C
Crotalaria novae-hollandiae	1	C	C	N	P
Cymbopogon bombycinus	1	C	C	N	C
Cymbopogon sp TBI	1	C	C	C	C
Dichanthium sericeum	1	C	C	N	C
Ectrosia leporina	1	C	C	N	C
Ectrosia sp TBI	1	C	C	N	C
Eriachne burkittii	1	C	C	C	C
Eriachne glauca	1	C	C	C	C
Eulalia aurea	1	C	N	N	C
Eulalia sp TBI	1	C	C	C	C
Ficus benjamina	1	I	C	C	C
Ficus virens	1	I	C	C	C
Fimbristylis sp TBI	1	C	C	N	C
Heteropogon contortus	1	C	N	N	C
Hibiscus sabdariffa	1	C	C	N	P
Sarga intrans	1	C	P	N	C
Sarga plumosum	1	C	P	N	C
Tephrosia rosea	1	C	C	C	P
Themeda triandra	1	C	P	N	C
Triodia bitextura	1	C	C	N	C

C Completed; P In progress; N Not started, I insufficient seeds, TBI – identification to be confirmed.

The CDU researchers were successful in gaining CDU research training program funding and this was used to investigate the fruit-imposed dormancy of *Persoonia falcata*. The type of fruit treatment required to allow the seeds to germinate has been identified. Extracting the seed and applying gibberellic acid resulted in 60% germination, which was equivalent to the proportion of viable seeds. A second seed lot is being investigated to confirm this result prior to publication.

In 2009, summary reports were provided to sponsors and to KNPS. Meetings were also held in December 2008 and July 2009 to present data and discuss progress. Two other meetings with KNPS, EWLS and CDU were held in Jabiru to maximise information and technology transfer. The research work will continue to test new species and new seed lots of existing species. Considerable literature has been obtained on the taxa (although often on southern species of the genera in the priority list).

Re-prioritisation of species for propagation

At the July 2009 project meeting, a more objective prioritisation process was recommended for seed biology research, based upon the ranking of relative abundance of plant species of each life-form (trees, shrubs and ground cover), and the success to date in collecting and propagating the various species based upon the collective experience of EWLS, KNPS and CDU. For this ranking and scoring approach, EWLS used quantitative abundance information for plants arising from the combined EWLS and ***eriss*** vegetation analogue sites (Humphrey et al 2007), adopting the following methodology:

Separately for the tree (>2 m height) and shrub (<2 m height) species, the combined ***eriss*** and EWLS analogue data used in Humphrey et al (2007) were used to determine the average density of each species (stems ha^{-1}). These stem density values were then ranked and plotted as a frequency distribution. For each species, it was assessed whether: (i) tubestock has been used for revegetation at Ranger and Jabiluka; (ii) whether seeds have been successfully germinated by KNPS or CDU; and (iii) whether seeds are available from KNPS. (In Figures 1 & 2, the three categories are coded ‘Used in reveg?’, ‘Germination?’ and ‘Seeds available?’ respectively.) Different symbol codings were assigned in the ranking graphs, depending upon whether the criterion was met, where a knowledge gap was identified or where the criterion was not met. Note that the ‘Germination’ or ‘Seeds available’ designation has been applied wherever this has occurred on at least one occasion. It does not necessarily imply that germination will occur consistently or that seeds are reliably available. Thus in future, there may well be seed biology issues to be overcome with these species. Nevertheless, research needs for these species have a lower priority than species for which no knowledge is available.

For herb and grass species, the analogue data of Brennan (2005) relevant to sites adjacent to Ranger were used. Species were ranked according to plant biomass (g dry mass m^{-2}), and plotted similarly as a frequency distribution. For each species, it was assessed whether (i) plants have been recorded as natural recruits in revegetation at Ranger (primarily in the ‘heritage’ revegetation site on the east of the tailings dam wall); (ii) whether seeds have been germinated successfully by KNPS or CDU; and (iii) whether seeds are currently available from KNPS. (In Figure 3, the three categories are coded ‘Used in reveg?’, ‘Germination?’ and ‘Seeds available?’ respectively.) As for the trees and shrubs, different codings were assigned in the ranking graphs, depending upon whether the criterion was met, where a knowledge gap was identified or where the criterion was not met.

Ranked species abundances for all three categories (trees, shrubs and ground cover), together with the three categories of collective knowledge and knowledge gaps for each species, are provided in Figures 1–3 respectively. For each life-form category, the greatest priority for plant biology research would be expected to be placed on the abundant species for which there are significant knowledge gaps and/or concerns that seed stock is not readily available locally. The plots will be revised as additional information is made available but the most obvious observation at this stage is the dearth of seed biology information available for shrubs and ground cover species (Figures 2 and 3, respectively) compared to the amount of information available for trees (Figure 3). Thus a key recommendation from this review is that the focus for future seed biology research be shifted to shrubs and groundcover species. However and as noted above, we do not imply that the data in this report indicate that seed biology issues have necessarily been overcome for the other species. For reliable future rehabilitation, further testing of seed lots of all species to be used for trials is recommended, at the very least until quantitative data are acquired for three or more seed lots.

Where poor availability of seed stock is identified in the plots, other commercial sources of seed from outside of the ARR might be sought for research (propagation) purposes (eg www.topendseeds.com.au/price-list.php) and indeed, for rehabilitation generally noting that provenance issues will need to be resolved for these species.

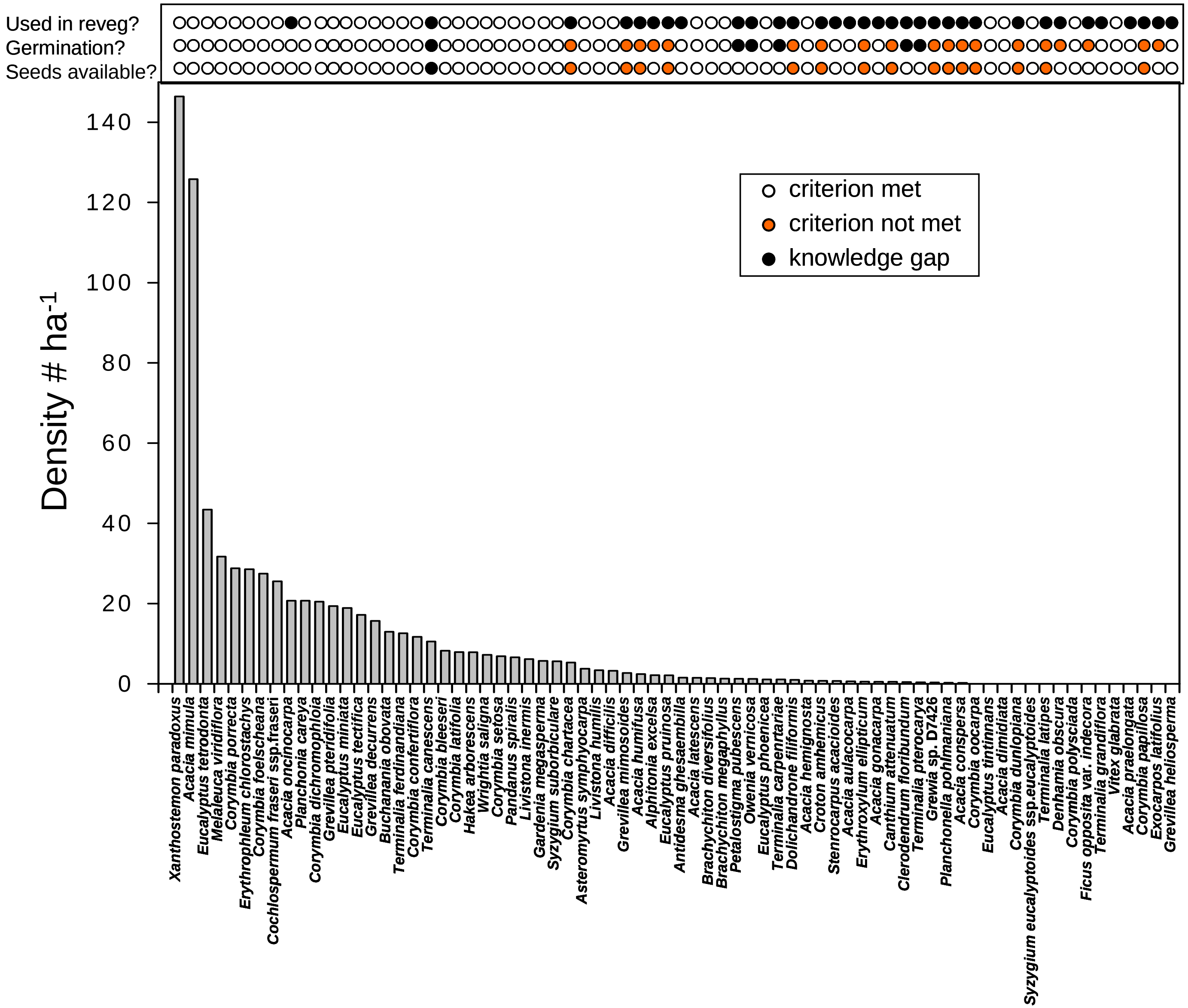

Figure 1 Relative density of trees present in ARR vegetation analogue sites (Humphrey et al 2007) and knowledge available for seed propagation and revegetation

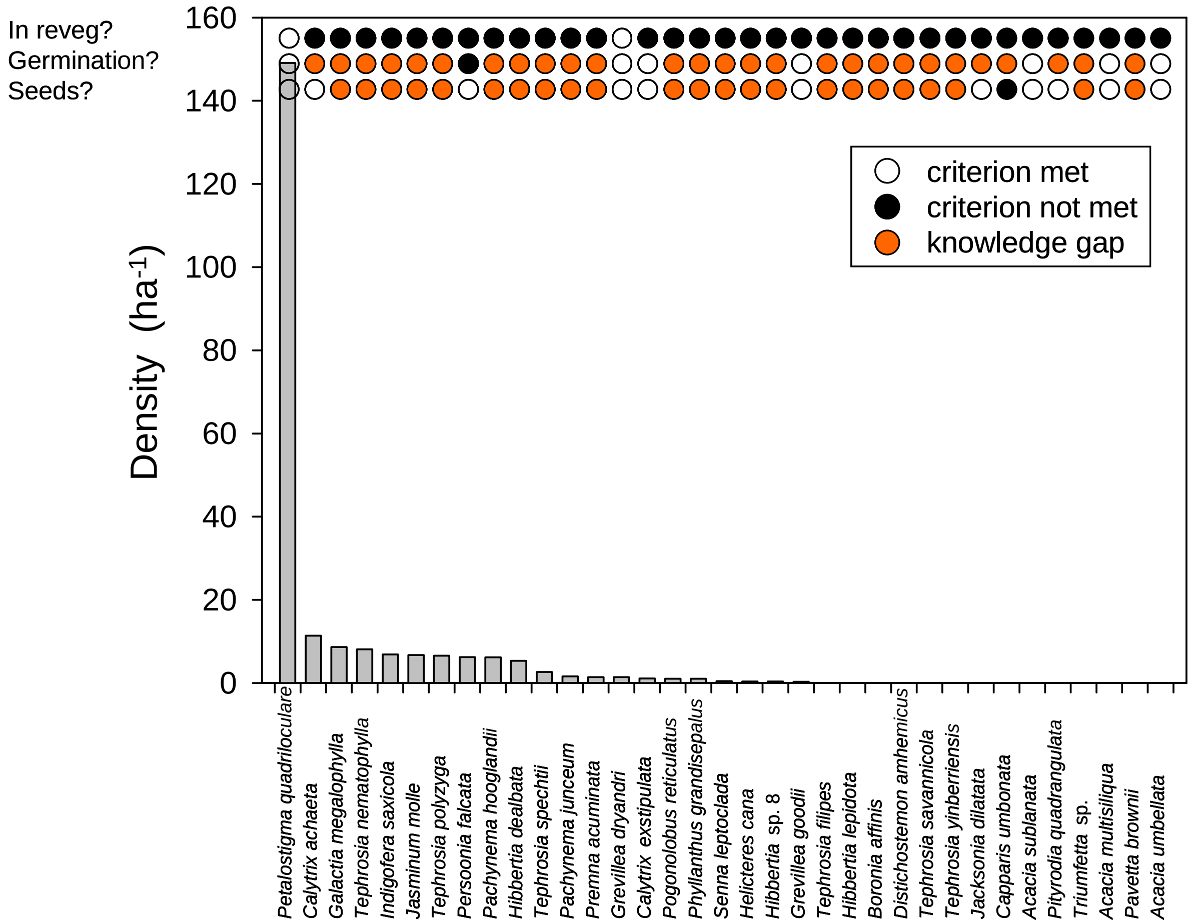

Figure 2 Relative density of shrubs present in ARR vegetation analogue sites (Humphrey et al 2007) and knowledge available for seed propagation and revegetation

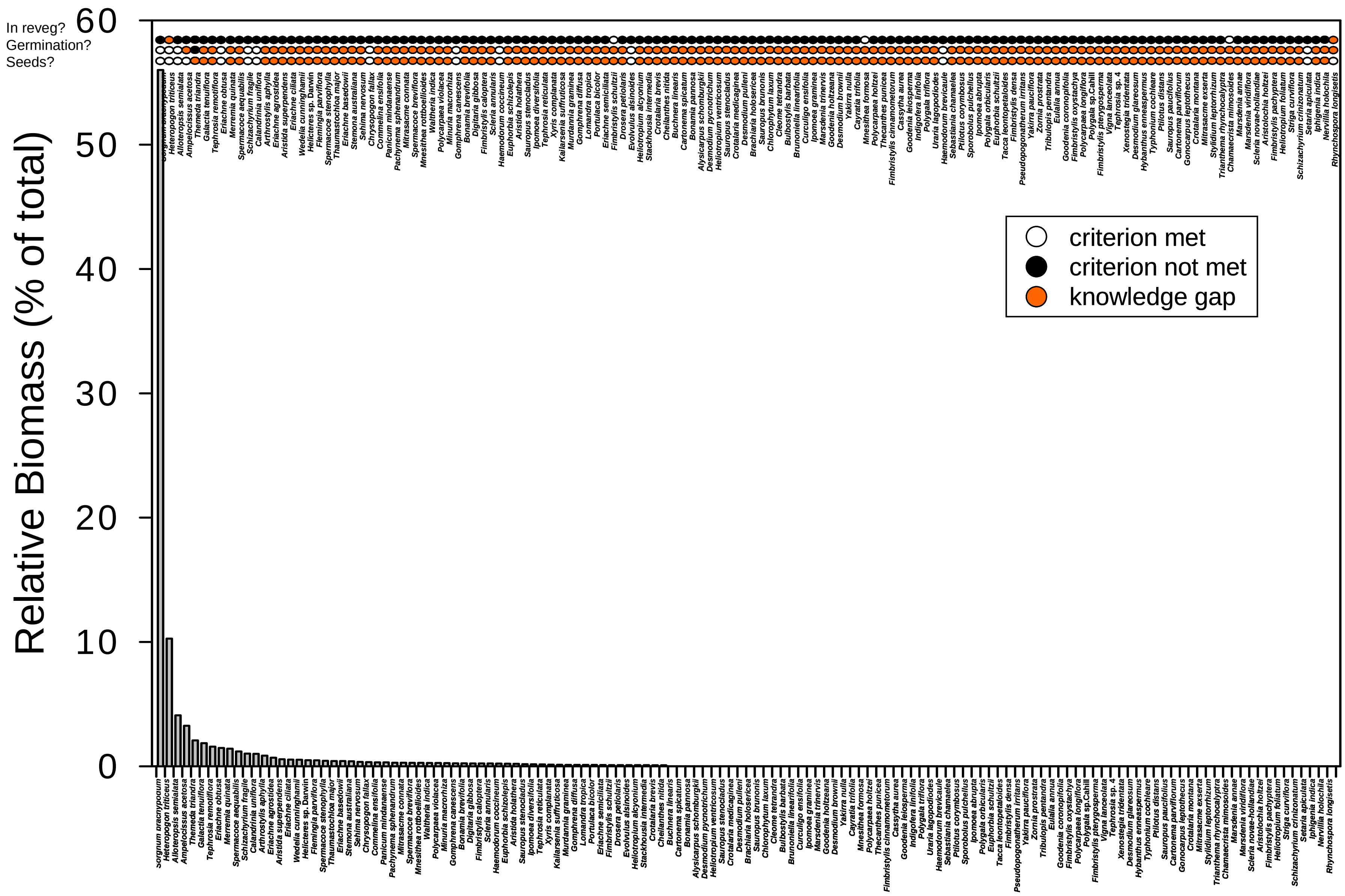

Figure 3 Relative density of herbs and grasses present at Ranger vegetation analogue sites (Brennan 2005) and knowledge available for seed propagation and revegetation

References

Bellairs SM & Ashwath N 2007. Seed biology of tropical Australian plants. In *Seeds: biology, development and ecology*. eds SW Adkins, S Ashmore & SC Navie, CABI Publishing, Oxford UK, 416–427.

Bellairs SM 2007. Seed biology of plants of the Australian wet/dry tropics and implications for Ranger mine site rehabilitation. Internal Report 523, March, Supervising Scientist, Darwin. Unpublished paper.

Bellairs SM & McDowell M 2008. ERA Ranger Uranium Mine Seed Biology Project Annual Report 2007–2008. School of Environmental and Life Sciences, Charles Darwin University, Darwin NT.

Brennan K 2005. Quantitative descriptions of native plant communities with potential for use in revegetation at Ranger uranium mine. Internal Report 502, August, Supervising Scientist, Darwin. Unpublished paper.

Humphrey C, Hollingsworth I, Gardener M & Fox G 2007. Use of analogue plant communities as a guide to revegetation and associated monitoring of the post-mine landform at Ranger. In ***eriss** research summary 2005–2006*. Jones DR, Evans KG & Webb A (eds), Supervising Scientist Report 193, Supervising Scientist, Darwin NT, 84–86.

Investigating radium uptake in *Passiflora foetida* (bush passionfruit)

P Medley, A Bollhöfer & D Parry[1]

Introduction

Our current ability to predict radiological dose received via the ingestion of radionuclides in terrestrial plants growing on a rehabilitated minesite is limited because the uptake mechanisms of radionuclides in plants are not well understood. The most common approach to determine doses is to use concentration factors for each food item and radionuclide to assess radionuclide uptake, and a model diet to estimate the quantity of each radionuclide that is ingested. Reported concentration factors, expressed as the ratio of the radionuclide activity concentration in the food item to the activity concentration measured in the soil in the plant root zone can, however, vary by up to three orders of magnitude, even for individual soil-crop combinations (Simon & Ibrahim 1990, Ryan et al 2005, IAEA 1994; Vandenhove et al 2009). Radium uptake in particular has been reported as being highly variable in tropical environments (Velasco et al 2009). In addition, although the International Atomic Energy Agency (IAEA) provides default concentration factor values for some food items (which are currently under review), the analogues available for local foods in the Alligator Rivers Region (eg. potato as an analogue for yam) have been shown to be inaccurate. In some cases, the concentration factors are different by several orders of magnitude (Ryan et al 2005).

The need exists to develop concentration factors specific for a region or even for individual sites to enable a more reliable prediction to be made for radionuclide activity concentrations in plants growing on a rehabilitated minesite. Site-specific studies for the purpose of determining concentration factors have also more recently been suggested by Vandenhove et al (2009) and Velasco et al (2009). Of particular importance for mine site rehabilitation are ^{226}Ra, ^{210}Po and ^{210}Pb. These radionuclides, which have comparatively high ingestion dose coefficients (ICRP 1996), have previously been identified as the main contributors to radiological dose via the ingestion pathway from eating traditional terrestrial bush food items such as fruits and yams in the Alligator Rivers Region (Martin & Ryan 2004).

Various approaches have been suggested to operationally define the transfer mechanisms of radionuclides from soil to plant, in an attempt to lower variability in concentration factors and to account for differences in bioavailability of metals between different soil types and/or sites. Many of these approaches have attempted to define the 'bioavailable' fraction within the soil using chemical techniques such as sequential extraction (eg Tessier et al 1979). In our study, a soil sequential extraction procedure was developed to assess partitioning of radium between soils and plants in the Alligator Rivers Region. Correlation of ^{226}Ra/^{228}Ra activity ratios in the different extractable fractions from soil in the plant root zone with those in the edible fruit has been used to infer the specific pools of radium in soil that are available for uptake into the fruit.

[1] Australian Institute of Marine Science, Arafura Timor Research Facility (ATRF), Darwin NT

Methods

An introduced passionfruit species, *Passiflora foetida* (Figure 1), was selected for study since it is eaten by indigenous groups in the Alligator Rivers Region and is also commonly eaten by children. This is important because ingestion dose coefficients are higher for children than for adults. *Passiflora foetida* is a shallow rooted, fast growing weed that may flower and fruit at any time of the year. This makes it ideal for short-term studies, and more likely to have a higher uptake of radium when growing in areas where contaminated material is retained in the surface layers.

Figure 1 Passiflora foetida

Samples of fruit and associated soils were collected from a wide range of sites including: the rehabilitated Nabarlek uranium mine in western Arnhem Land; a historic mine area impacted by uranium mine tailings (‘Rockhole residues’) in the South Alligator River valley; a land application area at Ranger mine impacted by mine waters; the black plain soils to the south-west of the Ranger tailings dam; and background sites not impacted by mining. The soil was then subjected to a sequential extraction procedure and radium isotopes (^{226}Ra and ^{228}Ra) measured in the soil extracts and fruit to determine concentration factors for the various fractions. The fractions (and extraction techniques) defined by this study were: the bioavailable fraction (water followed by a 1 M $MgCl_2$ leach, with the Ra extracted by the two steps being combined); the fraction bound to iron and manganese oxides (1 M HCl leach); the fraction that could be bound by organic complexes, particularly Ra-sulfate (Willett & Bond 1995) due to the very high levels of sulfate in the irrigation waters and extremely low solubility of radium in water and acid solutions (Kirby 1964) (0.2 M EDTA in 1.7 M NH_4OH); and the residual fraction (measured via gamma spectrometry) (Medley 2007).

Results

Final results of the ^{226}Ra and ^{228}Ra activity concentration measurements are now available to identify which extract fraction may be most suitable to use to determine concentration factors for the uptake of radium in *Passiflora*.

Total ^{226}Ra activity concentrations in the soils associated with the collected samples of *Passiflora* range over three orders of magnitude (35–11700 Bq/kg dry weight). The highest values measured are from the Rockhole residues site in the South Alligator River valley where passionfruit was growing in soil that contained tailings rich in ^{226}Ra. Activity concentrations are also elevated in the soils of the Magela land application area (MLAA) at Ranger mine due to the application of pond waters containing elevated ^{226}Ra, and in soils from the Gulungul catchment. The lowest soil activity concentrations were measured at a site close to Magela Creek, downstream of but un-impacted by the minesite. It is noted that the sites sampled cover a range of different origins for the ^{226}Ra, from being applied in solution to the top of the soil profile in the MLAA to residual ^{226}Ra in mine tailings at the Rockhole site. Hence it could be expected that the Ra might be bound in different ways in each of these soil types. The ^{226}Ra activity concentrations [Bq·kg^{-1} dry weight] measured in the samples of *Passiflora* range from about 3 Bq·kg^{-1} at the Magela Creek downstream site, to 520 Bq·kg^{-1} at the Rockhole residues site.

In contrast to ^{226}Ra, the ^{228}Ra activity concentrations in the soils were similar at all sites and reflect typical values seen throughout the region, ranging from 12–84 Bq/kg. The variability of ^{228}Ra activity concentrations in the fruits is very small and ranges from 1.3–5.1 Bq/kg.

Table 1 shows the calculated concentration factors derived from the ^{226}Ra activity present in the total soils and in the different selective extraction treatments for the various sites.

Table 1 ^{226}Ra Concentration Factors for *passiflora foetid*a measured relative to the various leach fractions

Sampling site	Concentration factors relative to extraction method			
	Total soil	**Bioavailable**	**1M HCl**	**EDTA + NH_4OH**
Rockhole residues	*0.030±0.001*	*0.27±0.012*	*0.201±0.006*	*0.70±0.02*
Nabarlek	*0.086±0.005*	*1.39±0.11*	*0.316±0.017*	*1.59±0.09*
Magela LAA	*0.018±0.001*	*0.45±0.03*	*0.079±0.005*	*0.34±0.02*
Gulungul 1	*0.0050±0.0002*	*0.35±0.02*	*0.010±0.001*	*0.086±0.004*
Gulungul 2	*0.0037±0.0002*	*0.32±0.02*	*0.0068±0.0003*	*0.066±0.003*
Magela DS	*0.238±0.021*	*1.87±0.11*	*1.131±0.074*	*2.86±0.28*

The concentration factor is defined as the activity concentration in the dry fruit divided by the activity concentration in the dry soil or in the leach fraction. Activity concentration in the leach fraction is expressed as the total activity leached divided by the dry soil weight. Uncertainties are one standard deviation based on counting statistics only.

Using total soil, and the HCl and EDTA extraction steps, as the basis for the calculation there is an up to two order of magnitude range of values for the concentration factors spanned by the data shown in Table 1. However, when the bioavailable fraction only is considered, a much smaller variation is observed with values ranging from 0.27–1.87. Critically, the use of the ^{226}Ra/^{228}Ra activity ratio as a tracer of the origin of the radium in passionfruit confirmed that the bioavailable fraction is the most likely source of the radium taken up by the plant. That is, the ratio of ^{226}Ra/^{228}Ra in the plant is most similar to the ratio that is found in the bioavailable extraction. The calculated concentration factor based on the bioavailable fraction is plotted against total soil activity concentration in Figure 2. The ^{226}Ra concentration factors are highest at the lowest soil activity concentrations and approach a saturation value of approximately 0.3 at high soil ^{226}Ra activity concentrations. This suggests that Ra uptake is non-linear – this has previously been suggested for plants from more temperate regions (Martinez-Aguirre et al 1997, Madruga et al 2001).

The non-linearity of concentration factors has important implications for, and further complicates, dose models that use concentration factors to derive ingestion doses, as ^{226}Ra activity concentrations in plants may be over- or underestimated, depending on the degree of contamination of the soil. Site-specific concentration factors should thus be determined relative to the bioavailable fraction of ^{226}Ra in the substrate to increase the confidence in the modelled ingestion doses.

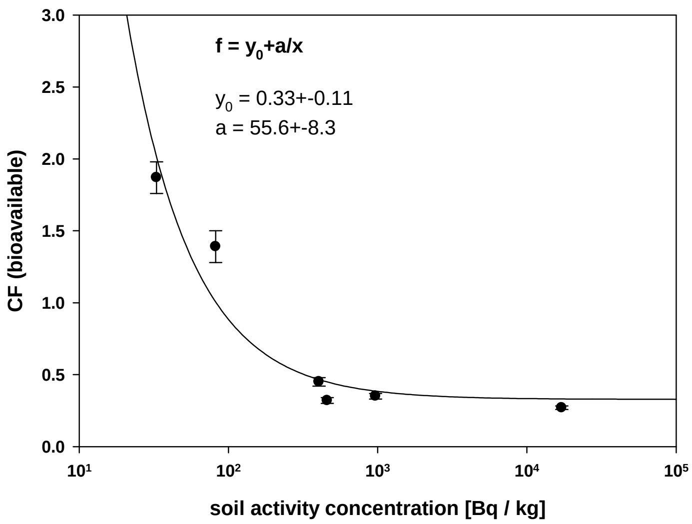

Figure 2 Concentration factors based on the bioavailable fraction plotted versus soil ^{226}Ra activity concentration (log scale)

Steps for completion

It is planned, using the selective extraction and radium isotope ratio approach described above, to extend the *Passiflora* work to other plant species commonly eaten by indigenous people in the Alligator Rivers Region.

Acknowledgments

We would like to acknowledge Anthony Sullivan and Sally Atkins from SSD for providing advice and for their help with sample collection.

References

IAEA 1994. *Handbook of parameter values for the prediction of radionuclide transfer in temperate environments*. IAEA Technical Report Series No 364, Vienna.

ICRP 1996. *Age-dependent doses to members of the public from the intake of radionuclides: part 5. Compilation of ingestion and inhalation dose coefficients*. International Commission on Radiation Protection Publication 72.

Kirby HW & Salutsky ML 1964. *The radiochemistry of radium*. National Academy of Sciences, Nuclear Science Series NAS-NS 3057.

Madruga MJ, Brogueira AG & Cardoso F 2001. ^{226}Ra bioavailability to plants at the Urgeirica uranium mill tailings site. *Journal of Environmental Radioactivity* 54, 175–188.

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

Martin P & Ryan B 2004. Natural-series radionuclides in traditional Aboriginal foods in tropical northern Australia: a review. *TheScientificWorldJOURNAL* 4, 77–95.

Martínez-Aguirre A, García-Orellana I & García-León M 1997. Transfer of natural radionuclides from soils to plants in a marsh enhanced by the operation of non-nuclear industries. *Journal of Environmental Radioactivity* 35(2), 149–171.

Medley P 2007. Validation and refinement of a method for determination of ^{228}Ra, via gamma spectrometry using the ^{228}Ac daughter, or alpha spectrometry using the ^{228}Th daughter. Honours Thesis, Charles Darwin University, Darwin, Australia.

Ryan B, Martin P & Iles M 2005. Uranium-series radionuclides in native fruits and vegetables of northern Australia. *Journal of Radioanalytical and Nuclear Chemistry* 264(2), 407–412.

Simon SL & Ibrahim SA 1990. Biological uptake of radium by terrestrial plants. In *Environmental behaviour of radium*. IAEA Technical Reports Series 310, 545–599.

Tessier A, Campbell PGC & Bisson M 1979. Sequential extraction procedure for the speciation of particulate metals. *Analytical Chemistry* 51, 844–851.

Vandenhove H, Olyslaegers G, Sanzharova N, Shubina O, Reed E, Shang Z & Velasco H 2009. Proposal for new best estimates of the soil-to-plant transfer factor of U, Th, Ra, Pb and Po. *Journal of Environmental Radioactivity* 100 (9), 721–732.

Velasco H, Juri Ayub J & Snason U 2009. Influence of crop types and soil properties on radionulcide soil-to-plant transfer factors in tropical and subtropical environments. *Journal of Environmental Radioactivity* 100 (9), 733–738.

Willett IR & Bond WJ 1995. Sorption of manganese, uranium and radium in highly weathered soils. *Journal of Environmental Quality* 24, 834–845.

Storing, accessing and communicating the bushtucker project information

D Walden, R Bartolo, B Ryan & A Bollhöfer

Background

The intake of radionuclides in traditional bush food items sourced from radiologically enhanced/contaminated areas has been identified as a significant potential contributor to the radiological dose to traditional people living the Alligator Rivers Region, in particular after minesite rehabilitation (Ryan et al 2005). There are a large number of dietary items, radionuclides and exposure pathways to be considered to assess potential ingestion doses. The ***eriss***/EnRad bushtucker database contains over 1500 records (individual samples) of a wide variety of plants and animals (and water and sediments) collected from over 100 sites in the Alligator Rivers Region and as far east as Maningrida in Arnhemland. The challenge is to develop a user friendly system to store and retrieve the data, and to present this information to the local people of the area, and a wider public audience, using media such as the internet and other graphical user interfaces. Employing an interactive spatial interface will greatly simplify locating sites and presenting data spread over such a broad geographic area.

Methods

The Department of the Environment, Water, Heritage and the Arts employs Google Maps for their Heritage website where Google Maps is embedded into the web pages (see Figure 1 www.environment.gov.au/heritage/index.html).

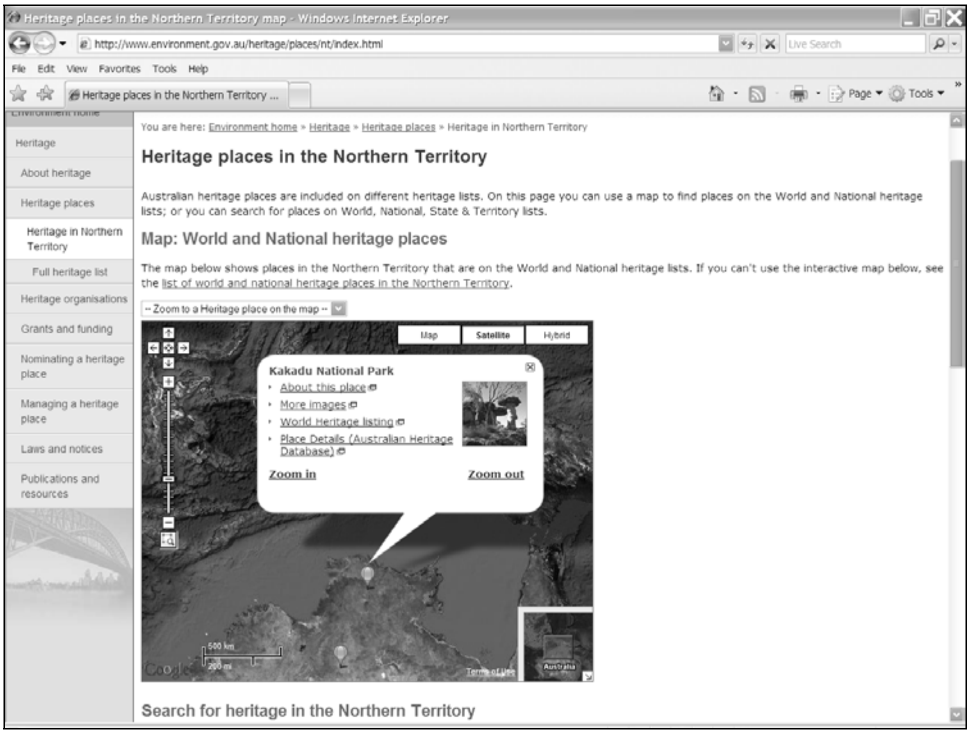

Figure 1 DEWHA website with Google Maps used to find Heritage sites in Australia

The user selects a state from a map of Australia, zooms into and clicks on any number of placemarks revealing a callout box with text, images and hyperlinks to a host of information relevant to that site.

A similar bushfoods website with these features is currently being developed whereby users can easily locate the sites sampled for bushtucker, what species were examined and other relevant information. The website will be formatted according to the Departmental template with appropriate headings, banners, logos menus and links as per other DEWHA web pages.

In addition, Keyhole Markup Language (KML) files containing site locations and associated data are being developed in parallel to the website information. The files are in a compressed format and can be made available to the public via download from the internet or on storage media such as CD/DVD. They can be used by a growing number of freeware 3D browsers such as Google Earth, ArcExplorer and Arc Globe, enabling users to zoom into sites using high resolution satellite imagery and manipulate the tilt, roll, angle, altitude and terrain of the landscape. In most cases the KML files can also be imported into web-based viewers such as Google Maps, Virtual Earth (Bing Maps), Yahoo Maps and Whereis.com. The advantage of the KML files in a full screen environment such as Google Earth, is the greater available space for images and information directly in the placemark callout box without having to hyperlink to other pages. The disadvantage is that the viewing program has to be installed manually and performance and refresh times can be limited by computer hardware.

At this stage of the project, information will be presented in general terms as outlined in Table 1. The actual results for radionuclides and other elements will not be presented for viewing or use by external stakeholders and the general public because interpretation of the radionuclide results is complex, and general users may find the numbers confusing and/or misleading.

Table 1 Information to be presented for external stakeholders and the general public

Information	Details
Differing names of species	• Common • Latin (scientific) • Bininj (local Aboriginal)
Why is the site important?	• Proximity to contamination source or control site • Used for food frequently • Special cultural significance
Why has the food been sampled/tested?	• Eaten frequently • Position on the food web and feeding habits – bioaccumulation issues • Proximity to a contamination source • Totem/cultural significance
How is the food sampled/obtained/prepared?	• Methods used by scientists – eg dredge • Traditional harvesting methods • How is the food prepared for eating? • How is the sample prepared for testing?
What parts of the animal/plant have been tested?	• Flesh • Various internal organs • Fruit, leaves, stems or roots
What has the food been tested for?	• Which radionuclides and/or heavy metals?

Progress/results

The database contents were converted to KML files early in 2008[1]. Many results are expressed both as radionuclide activity concentration and concentration factors, giving at least two separate radionuclide values for any one sample at any given site. Individual sites often have dozens of samples (Figure 2). Consequently, the default presentation method for multiple attributes for one location in Google Earth whereby the sample icons expand out from a representative site icon, while impressive, is simply overwhelming. In addition, the format in which the data translates from the GIS files is far from meaningful to the lay person. Names are truncated and the default callout box reflects the original database structure (Figure 2). Consequently, the database content that will be presented in the KML files has been 'streamlined' to suit the presentation method. A number of positional errors have also been corrected.

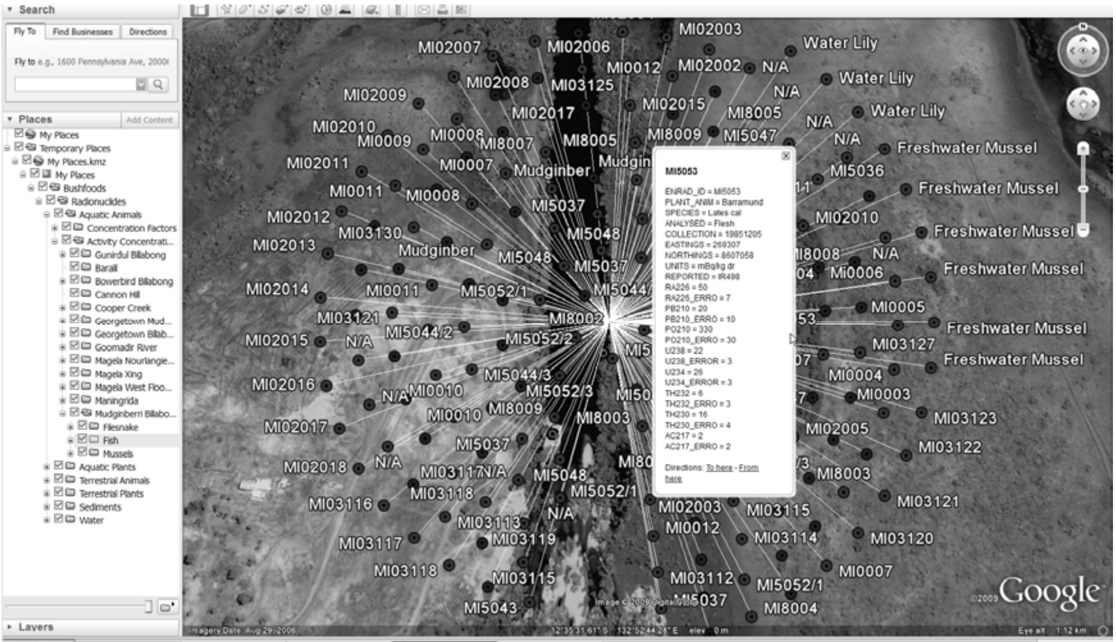

Figure 2 A single sampling site at Mudginberri Billabong selected in Google Earth showing all samples

Early in 2009, work recommenced on the KML files in the Google Earth environment. As the actual values are generally not being presented it is possible to present summarized information directly in the callout boxes as free text along with images and hyperlinks to either other KML files or web pages if necessary. A number of KML based templates have been designed to display information in a more user friendly and visually aesthetic way. The template content can be easily edited to describe different sites and samples. A collection of customized icons has been created representing most of the species sampled. These were created from photographs owned by SSD and some were provided by ***eriss*** staff. Examples of the templates and icons can be seen in Figures 3 and 4.

Liaison has commenced with the Web and Intranet Management Section (WIMS) of the Department of the Environment, Water, Heritage and the Arts to assess the feasibility of uploading all of the bushfoods information as outlined in this document, onto the Department's internet site.

---

[1] This follows on from earlier work by Bartolo et al 2007, who created ArcGIS shapefiles from the original database to develop a Bushtucker Spatial Information System (SIS).

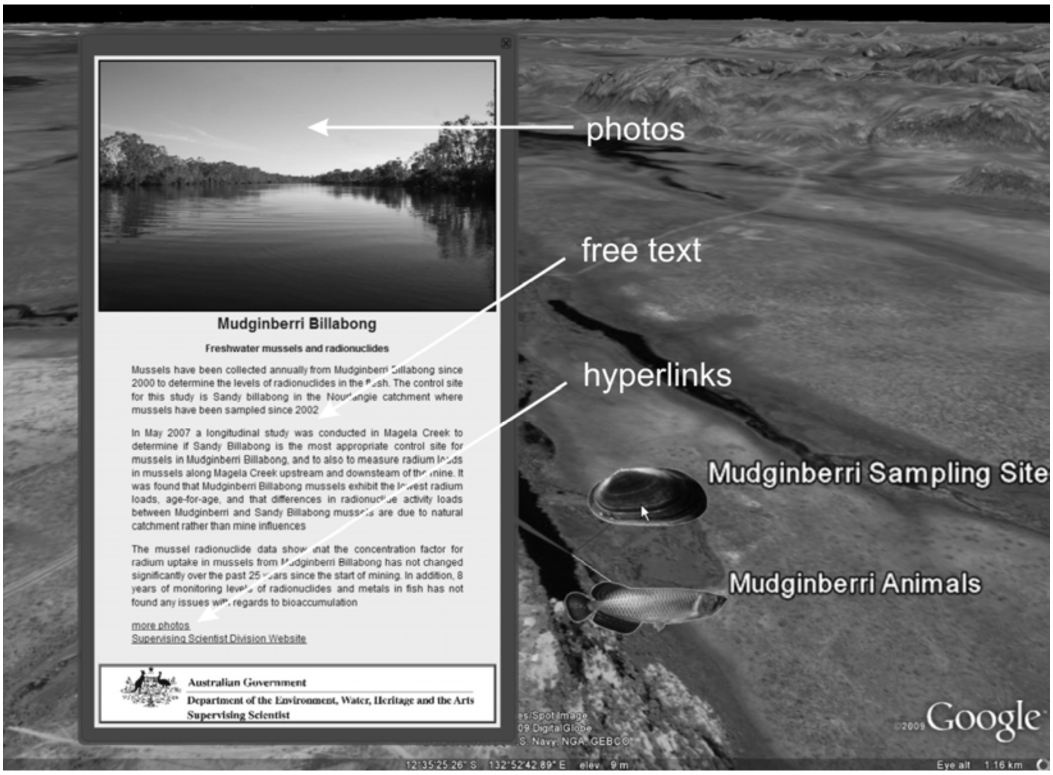

Figure 3 An example of a large callout box showing an image of the site, a description of the sampling and species and hyperlinks to more images and relevant websites for example

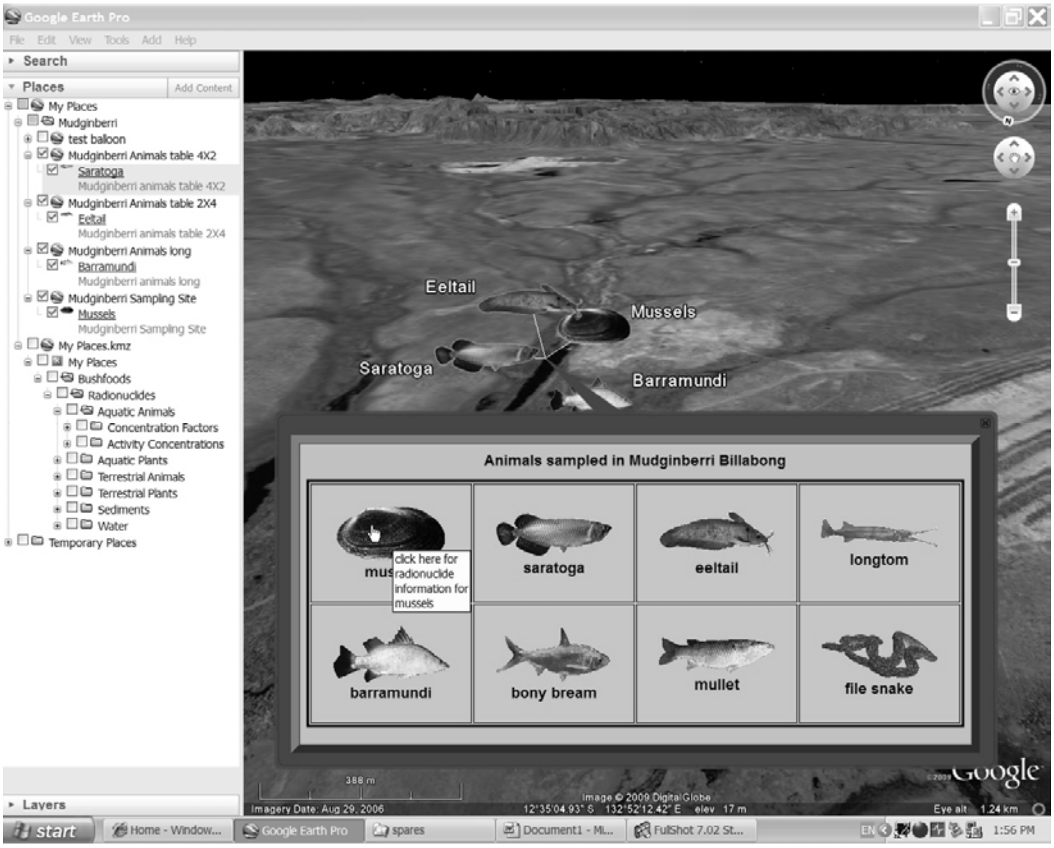

Figure 4 An example of a callout box representing species at a site. The icons can be links to further information about that species. Note the mouse pointer over the mussel icon. The view is looking south along the Magela Ck to Ranger in the background.

Conclusions

The use of this technology represents a new phase of communication of SSD research. Although there is abundant SSD information on the intranet and internet, to date there has been no real spatial presentation of data nor any spatial searching facilities. Internet accessibility will reach the widest audience searching for information on radionuclides and bushfoods. If each site is represented by a web page, there will be 100 or so pages to draft and it may become necessary to group sites and reduce detail to get the project on the web.

The KML products, although time consuming to produce initially, once completed and the content approved, can be distributed on CD or simply posted (and easily updated) on an existing SSD web page as a download for users with compatible browsers. These browsers are free, easy to install and use and are in general popular use.

Once the methods and procedures have been established for this project, it is anticipated that other SSD databases and indeed a host of general information will be presented using this technology.

Steps for completion

- Continue work on the summaries of site and species information.
- Complete the design of the species icons.
- Obtain photos of all sites.
- Populate the KML templates with the above information.
- Test the KML product prior to public access.
- Work with the WIMS to agree on how the internet product will be hosted. The task at hand is to draft some content, test its performance under a variety of conditions and seek approval to proceed under the Department guidelines.
- Once the above is completed, work can commence on drafting the web pages.

References

Bartolo R, Ryan B & Bollhöfer A 2007. Traditional diet in a modern world: Implementing a bushtucker SIS to communicate radiological issues. Paper presented at Spatial Sciences Institute Biennial International Conference 2007, Hobart Tasmania, 14–18 May.

Ryan B, Martin P & Iles M 2005. Uranium-series radionuclides in native fruits and vegetables of northern Australia. *Journal of Radioanalytical and Nuclear Chemistry* 264(2), 407–412.

Radio- and lead isotopes in sediments from the Nourlangie and Koongarra catchments (PhD project)

A Frostick, A Bollhöfer & D Parry[1]

Introduction

This project aims at developing an innovative and sensitive methodology to investigate and monitor sources, pathways and deposition of materials eroded from past, present and future uranium mining activities in the wet-dry tropics. Funded through the ARC Linkage Projects scheme, the project is a collaboration with researchers from Charles Darwin University. The objective of the project is to characterise sources and pathways of pollutants in catchments in the Alligator Rivers Region at the decommissioned and rehabilitated Nabarlek mine site (Frostick et al 2008), the operating Ranger mine, and at natural analogues, in order to develop a joint lead isotope/radionuclide approach for monitoring transport and dispersion of erosion products from a (rehabilitated) uranium mine site.

Due to the source-specific lead isotope signatures and the fact that no physical or chemical fractionation of lead isotopes occurs during transport and deposition, stable lead isotopes are ideally suited as a contaminant source tracer. Lead isotopic fingerprinting relies on the fact that three of the four stable lead isotopes, lead-204, lead-206, lead-207 and lead-208, are produced by the decay of uranium-238 ($^{238}U \rightarrow {}^{206}Pb$, $t_{1/2} = 4.5 \cdot 10^9$ yrs), uranium-235 ($^{235}U \rightarrow {}^{207}Pb$, $t_{1/2} = 0.7 \cdot 10^9$ yrs) and thorium-232 ($^{232}Th \rightarrow {}^{208}Pb$, $t_{1/2} = 14 \cdot 10^9$ yrs), respectively. ^{204}Pb is of primordial origin only.

In uranium and thorium rich minerals, radiogenic lead is continuously produced over time. For example, monazites present in many sands with high Th/U exhibit $^{208}Pb/^{207}Pb$ and $^{206}Pb/^{207}Pb$ ratios much higher than the present day average crustal (PDAC) lead (Bosch et al 2002) which has $^{206}Pb/^{207}Pb \approx 1.20$ and $^{208}Pb/^{207}Pb \approx 2.48$, respectively. On the other hand, uranium ore bodies show elevated $^{206}Pb/^{207}Pb$ ratios but are low in $^{208}Pb/^{207}Pb$, as ^{208}Pb is formed by the radioactive decay of thorium rather than uranium. Gulson et al (1992), for example, measured $^{206}Pb/^{207}Pb$ ratios in particulates from uranium tailings at Ranger as high as 9.69, whereas $^{208}Pb/^{207}Pb$ is as low as 0.0494, in agreement with results from a study of airborne dispersion of Ranger mine origin dust (Bollhöfer et al 2006). There have also been several studies utilising the lead isotope fingerprinting technique elsewhere in the Alligator Rivers Region including: the use for mineral exploration at the Jabiluka and Koongarra ore bodies (Gulson & Mizon 1980, Gulson 1986); lead isotopes in soils and plants from the Koongarra deposit for biogeochemical prospecting (Dean & Gulson 1987); an assessment of sediments from Ngarradj (Swift Creek), a catchment potentially influenced by the Jabiluka mineral lease (Bollhöfer & Martin 2003); and an investigation of sediments from Cooper Creek potentially influenced by the rehabilitated uranium mine at Nabarlek (Frostick et al 2008).

Coupled with the measurement of radionuclide activity and trace metal concentrations, the lead isotope fingerprinting technique enables identification and quantification of the

[1] Australian Institute of Marine Science, Arafura Timor Research Facility (ATRF), Darwin NT

depositional patterns of solids eroded due to past and present mining activities (Munksgaard et al 2003, Frostick et al 2008). This part of the PhD project aims at investigating natural, undeveloped uranium mineralised analogues in the Alligator Rivers Region. Anomaly 2 to the south of the Ranger lease and the undeveloped Koongarra uranium deposit in the Nourlangie Creek catchment were investigated. This present report focusses on the results from the Nourlangie Creek catchment.

Location and methods

The Koongarra uranium deposit was discovered in 1970 by Noranda Australia Ltd. The lease covers approximately 1197 hectares and is located next to the Arnhem Land escarpment, 30 km south of Ranger uranium mine and 3 km east of Nourlangie Rock in a valley bounded by the Mount Brockman outlier and the Arnhem Land plateau. The orebody lies entirely within the catchment of Koongarra Creek which enters Nourlangie Creek and into the South Alligator River. Anbangbang Billabong is located approximately 5 km downstream of Koongarra mineral deposit, whereas Sandy Billabong is located in the Nourlangie Creek system upstream of the confluence of Koongarra with Nourlangie Creek, and is thus unaffected by runoff from Koongarra.

Surface sediments were collected from the Koongarra Mineral Lease during the dry season in 2007. Approximately 2 cm of the top sediment (< 2 mm) was placed in acid-cleaned poly-ethylene bags before drying at 60°C then grinding in an automated agate mortar and pestle. A fraction of the sample was taken for heavy metal and lead isotope analysis via inductively coupled plasma mass spectrometry (ICPMS). Around 150 g of the ground sediment was pressed into a standard geometry for direct gamma spectroscopy.

Sediment cores were retrieved from Anbangbang and Sandy Billabongs in late 2007. Cores were taken either by boat or direct bank access using lengths of PVC pipe driven into the sediment. Cores were cut into 10 mm sections using an acid cleaned stainless steel knife. Each section was then centrally cored using a stainless steel ring cutter and the external sediment, which had been in contact with the PVC, was discarded. Core samples were then dried and ground as described above. A fraction of the sample was taken for heavy metal and lead isotope analysis via ICPMS. For gamma spectroscopy samples from progressive depths were combined until approximately 15 g total of material was obtained. This mass was then pressed into a standard geometry and counted for 1–2 days.

Samples were analysed for radionuclides at ***eriss*** using high resolution High Purity Germanium (HPGe) gamma detectors in 2008-09. Procedures for sample collection, preparation and measurements of radionuclide activities via gamma spectrometry at ***eriss*** are described in Murray et al (1987), Marten (1992) and Esparon and Pfitzner (2010).

Results

Radionuclide activity concentrations of scrape samples taken on the Koongarra lease and from the Nourlangie Creek bed are relatively low and comparable with concentrations at other creeks within the Alligator Rivers Region. ^{210}Pb activity is higher than ^{226}Ra and ^{238}U, and ^{228}Ra and ^{228}Th are in equilibrium in all surface samples. The ^{40}K activities follow similar trends to ^{228}Th. Activities of ^{226}Ra and ^{210}Pb are highest at the two downstream sites before Koongarra Creek exits the lease area.

The top of the two billabong cores exhibit higher concentrations of radionuclides and heavy metals, probably related to the higher content of clay and organics compared to the sandier nature of the sediment in deeper sections of the cores. This has also been observed in sediment cores taken along Cooper Creek in Western Arnhem Land (Frostick et al 2008). Both Anbangbang and Sandy Billabong cores exhibit higher ^{210}Pb activity concentrations in the top sections as compared to ^{226}Ra (Figure 1) but they are generally in equilibrium (to within 2σ) in the bottom sections of the cores. ^{210}Pb excess dating of the two sediment cores reveals a sedimentation rate at the Sandy Billabong site of approximately 0.15 ± 0.01 cm yr^{-1}, whereas the sedimentation rate for Anbangbang Billabong is higher at 0.34 ± 0.12 cm yr^{-1}. ^{137}Cs activity concentration profiles measured support these sedimentation rates (Figure 1).

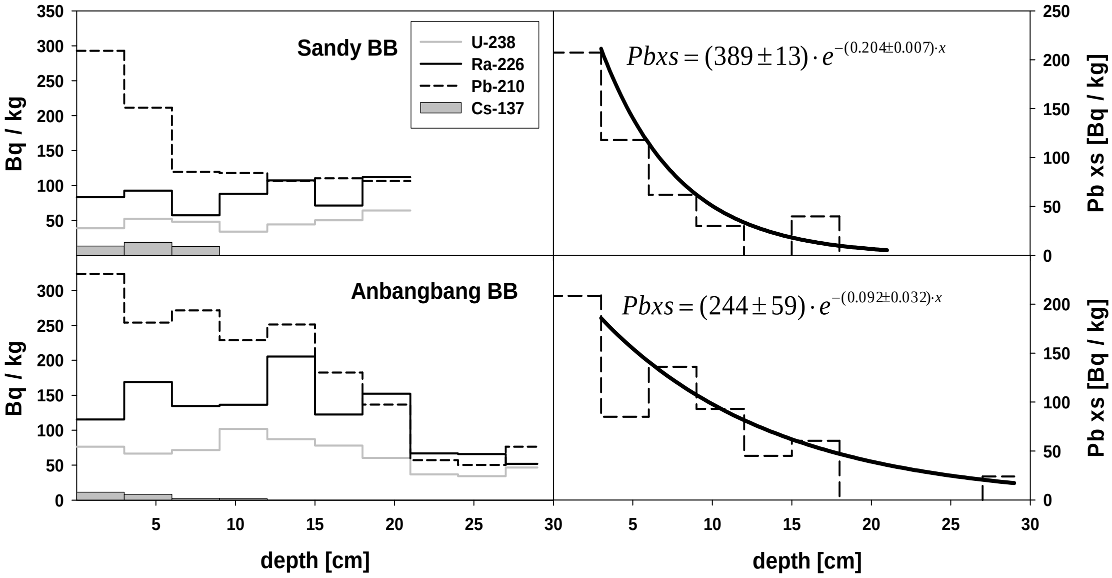

Figure 1 ^{238}U, ^{210}Pb, ^{226}Ra and ^{137}Cs activity concentrations, and ^{210}Pb excess versus depth in Sandy and Anbangbang Billabongs. Exponential fits to the ^{210}Pb excess (Pb_{xs} = ^{210}Pb – ^{226}Ra) activity concentration data are also shown (from Frostick et al 2010).

Variations in metal concentrations through the cores are due to changes in core lithology. The Sandy Billabong core showed a distinct layering of dark humic sections alternating with sandier layers at 10-14 cm (~65 yrs BP) and 19-21 cm (~125 yrs BP), respectively. Anbangbang Billabong contained only one major sand dominated layer from 22-29 cm (~65 yrs BP). The presence of sand layers is reflected in the lower trace metals concentrations and also radionuclide activities of the horizons. It appears that both cores recorded a sand layer deposited about 65 years before present.

The sediment samples from Sandy Billabong located at Nourlangie Creek, upstream of the confluence with Koongarra Creek, exhibit lead isotope ratios similar to the general background within the Alligator Rivers Region area, as reported in Frostick et al (2008). Trends in ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb are comparable to trends seen in Cooper Creek upstream of Nabarlek mine (Frostick et al 2008) and Ngarradj (Bollhöfer & Martin 2003) although the variations are much smaller (Figure 2). At Cooper Creek it was assumed that sands bearing heavy minerals with comparatively low Pb concentrations and high ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb ratios, are mixed with natural dust, clays and silts that contribute Pb with a significantly different lead isotopic composition, closer to present day average crustal (PDAC). The contribution of the heavy minerals to total Pb within the samples may be small but can be seen in the isotope ratios in the Sandy Billabong sediment samples, which have higher than PDAC ^{206}Pb/^{207}Pb and ^{208}Pb/^{207}Pb.

Anbangbang Billabong is located ~5 km downstream of the Koongarra mineral deposit and shows higher radionuclide activity concentrations in the sediment than Sandy Billabong. The samples also exhibit consistently more radiogenic $^{206}Pb/^{207}Pb$ but less radiogenic $^{208}Pb/^{207}Pb$ ratios than sediments from Sandy Billabong. Although higher ^{238}U activity concentrations can also be associated with an increase in the amount of silts and clays deposited in a sediment core, common silts and clays usually exhibit lead isotope ratios more similar to PDAC lead which would shift the ratio in the Anbangbang core to lower $^{208}Pb/^{207}Pb$ and lower $^{206}Pb/^{207}Pb$ isotope ratios. This is only the case in the top sediment sample. Throughout the remainder of the core $^{208}Pb/^{207}Pb$ is reasonably constant at about 2.57, whereas a rise in the $^{206}Pb/^{207}Pb$ ratio can be observed from about 80 - 60 years before present to an average of ~1.375 (Figure 2). The lower $^{208}Pb/^{207}Pb$ and higher $^{206}Pb/^{207}Pb$ compared with Sandy Billabong may be caused by uraniferous material originating from the Koongarra Mineral deposit.

Scrape samples collected for our study exhibit $^{206}Pb/^{207}Pb$ ratios up to ~ 1.80 and $^{208}Pb/^{207}Pb$ ratios of 2.53 (sample KML1). If we assume an end member ratio for uranium rich material from the Koongarra deposit of $^{206}Pb/^{207}Pb$ = 7.364 (data from Dickson et al 1985) and a background ratio of 1.328 (see Figure 2) this would indicate a relative contribution of about 1 percent of uraniferous sediment originating from the Koongarra mineral lease to the top sediments in Anbangbang Billabong.

The input of sediments with a higher $^{206}Pb/^{207}Pb$ but little different $^{208}Pb/^{207}Pb$ ratio in the upper section of the Anbangbang Billabong core is not apparent in Sandy Billabong.

Steps for completion

The method using trace metals and stable lead isotope ratio analysis is now being investigated for its potential to assess impacts from non-uranium mine sites. Samples have been taken from the vicinity of the former Mount Todd gold mine and are currently being analysed.

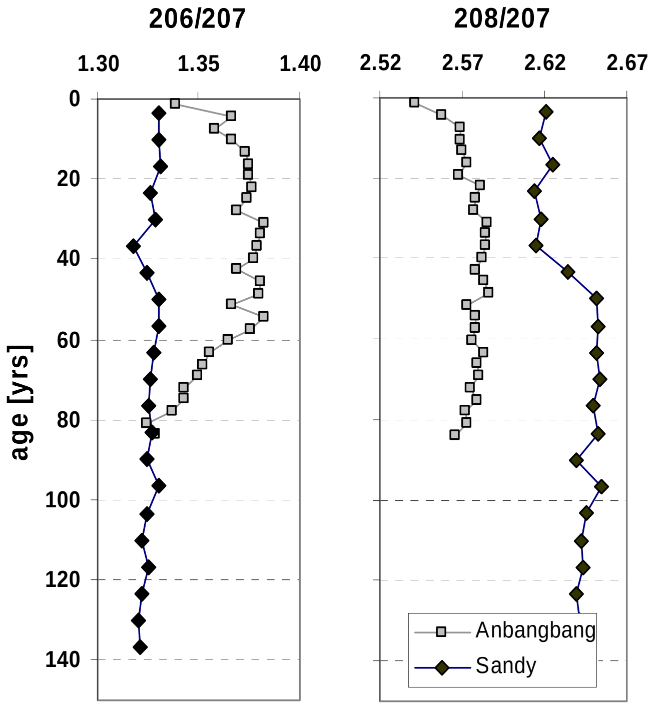

Figure 2 $^{206}Pb/^{207}Pb$ and $^{208}Pb/^{207}Pb$ ratios plotted versus age in both, Sandy and Anbangbang Billabong cores (from Frostick et al 2010)

References

Bollhöfer A, Honeybun R, Rosman KJR & Martin P 2006. The lead isotopic composition of dust in the vicinity of a uranium mine in northern Australia and its use for radiation dose assessment. *Sci. Total Environ.* 366, 579–589.

Bollhöfer A & Martin P 2003. Radioactive and radiogenic isotopes in Ngarradj (Swift Creek) sediments: a baseline study. Internal Report 404, February, Supervising Scientist, Darwin. Unpublished paper.

Bosch D, Hammor D, Bruguier O, Caby R & Luck JM 2002. Monazite 'in situ' $^{207}Pb/^{206}Pb$ geochronology using a small geometry high-resolution ion probe. Application to Archaean and Proterozoic rocks. *Chem. Geol.* 184, 151–165.

Dean JA & Gulson BL 1987. Biogeochemical prospecting using lead isotopes. *J. Geochem. Explor.* 29, 391–392.

Dickson BL, Gulson BL & Snelling AA 1985. Evaluation of lead isotope methods for uranium exploration, Koongarra area, Northern Territory, Australia. *J. Geochem. Explor.* 24, 81–102.

Esparon A & Pfitzner J 2010. Visual Gamma 2009, Gamma Analysis Manual. Internal Report 539. Supervising Scientist, Darwin. Unpublished paper.

Frostick A, Bollhöfer A, Parry D, Munksgaard N & Evans K 2008. Radioactive and radiogenic isotopes in sediments from Cooper Creek, Western Arnhem Land. *J. Environ. Radioact.* 99, 468–482.

Frostick A, Bollhöfer A & Parry D 2010. A study of radionuclides, metals and stable lead isotope ratios in sediments and soils in the vicinity of natural U-mineralisation areas in the Northern Territory, Australia. submitted to the *Journal of Environmental Radioactivity*.

Gulson BL 1986. *Lead isotopes in mineral exploration*. Elsevier, Amsterdam.

Gulson BL & Mizon KJ 1980. Lead isotope studies at Jabiluka. In *Uranium in the Pine Creek Geosyncline. Proceedings of the International Uranium Symposium on the Pine Creek Geosyncline.* eds Ferguson J & Goleby AB, International Atomic Energy Agency, Vienna, 439–455.

Gulson BL, Mizon KJ, Korsch MJ, Carr GR, Eames J & Akber RA 1992. Lead isotope results for waters and particulates as seepage indicators around the Ranger tailings dam: A comparison with the 1984 results. Open file record 95, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Marten R 1992. Procedures for routine analysis of naturally occurring radionuclides in environmental samples by gamma-ray spectrometry with HPGe detectors. Internal report 76, Supervising Scientist for the Alligator Rivers Region, Canberra. Unpublished paper.

Munksgaard NC, Lim K & Parry DL 2003. Rare earth elements as provenance indicators in North Australian estuarine and coastal marine sediments. *Est. Coast. Shelf Sci.* 57, 399–409.

Murray AS, Marten R, Johnston A & Martin P 1987. Analysis for naturally occurring radionuclides at environmental concentrations by gamma spectrometry. *J. Radioanal. Nucl. Chem. Art.* 115, 263–288.

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

K Evans & D Moliere[1]

Introduction

Severe category 5 tropical cyclone Monica crossed the Northern Territory coastline near Maningrida on 24 April 2006 and passed through the Magela Creek catchment near the Ranger Mine on 25 April 2006 (Figure 1), approximately 9 hours after landfall having reduced intensity to a Category 2 cyclone. Treefall damage, caused by high wind velocities, occurred around lower regions of the Magela Creek catchment, including the area surrounding the Ranger mine (Staben & Evans 2008). The treefall damage occurred primarily in areas with saturated soils (ie the terraces and palaeochannels which flank the creek channels).

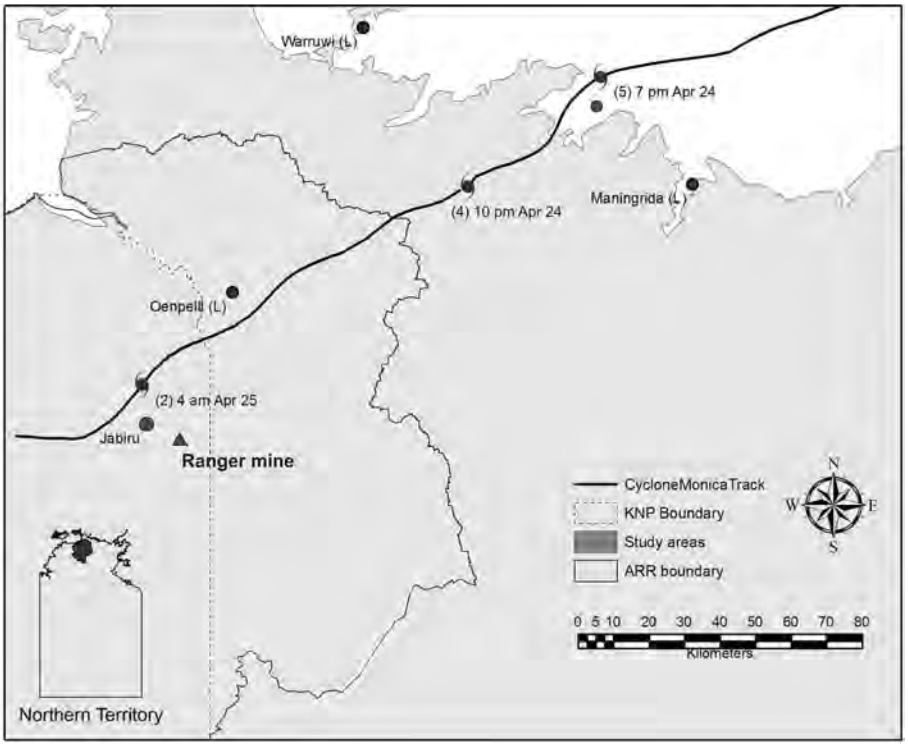

Figure 1 Track of Cyclone Monica over the Top End

The cyclone contributed to high stream suspended sediment concentrations in Magela Creek *during* the event (Figure 2). However, it is also important to assess the subsequent changes to the catchment sediment transport characteristics as a result of the substantial treefall

[1] Water Division – Bureau of Meterology, NT Regional Office

associated with this event, since such information can inform the risk assessment process for determining possible effects of future similar events on the catchment lines containing the rehabilitated Ranger minesite.

Suspended sediment data collected at Magela Creek Upstream (MCUS) and Magela Creek Downstream (MCDS) (Map 2) during 2006–07 and 2007–08 have been compared to suspended sediment data collected prior to the cyclone (2005–06 data) to assess the post-cyclone behaviour of the system. In particular, an assessment was made to determine whether or not a change in sediment transport characteristics has occurred in Magela and Gulungul Creeks and the duration of such impacts.

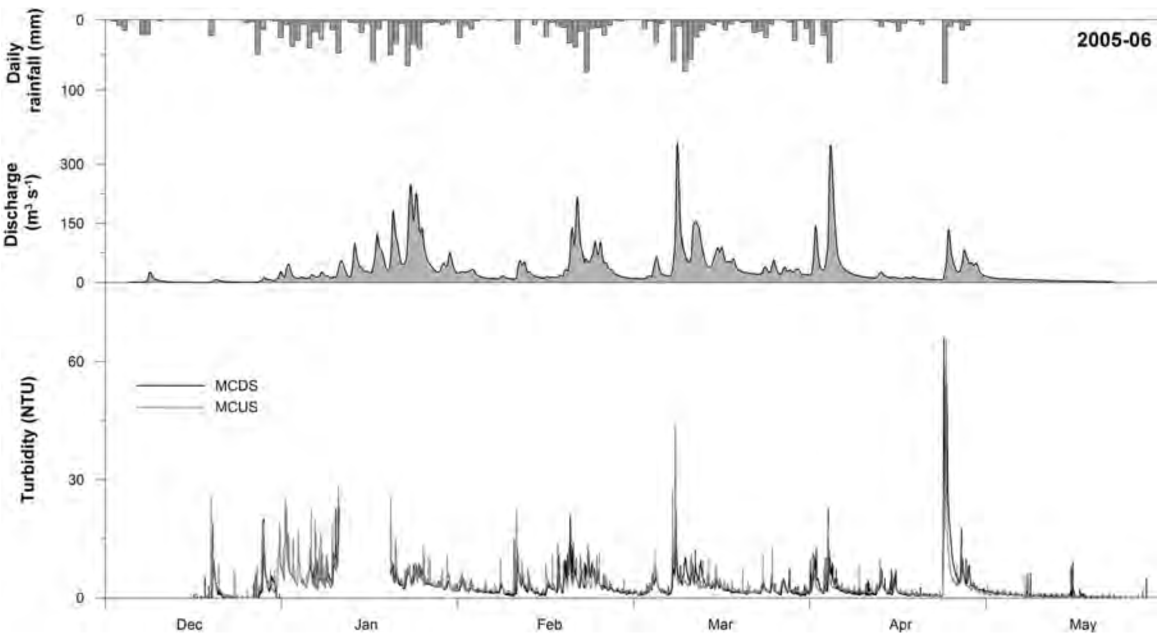

Figure 2 Annual hydrograph and sedigraph for Magela Creek (location) during 2005–06 highlighting the annual peak in turbidity on 25 April 2006 associated with the cyclone

Results

Fine suspended sediment (FSS – <63 μm and >0.45 μm diameter) was monitored at MCUS and MCDS using as a surrogate, turbidity data arising from field-calibrated turbidity sensors. A statistically significant relationship between laboratory-determined FSS concentration and field-derived turbidity was used to indirectly measure continuous FSS concentration.

The continuous turbidity data were used to quantify the FSS loads transported by an FSS pulse event, rather than assessing instantaneous FSS concentrations in the stream, an approach which is typically complicated by hysteresis effects[2] (Moliere et al 2005). FSS load during a pulse is defined as the area under the event sedigraph, where FSS concentration rises above, and then returns to, approximate base-flow levels (2–5 mg L^{-1}).

[2] The hysteresis effect occurs when the same FSS concentration value is measured during the rising stage and falling stage of an event hydrograph but for different instantaneous discharge values. This effect is a significant complicating issue when developing regression relationships between instantaneous FSS concentration and instantaneous discharge.

Two methods based on event FSS loads were used to determine whether there has been a change in sediment transport characteristics within Magela Creek as a result of Cyclone Monica:

1 A student *t*-test based on a Before-After-Control-Impact Paired difference design (BACIP) (Stewart-Oaten et al 1986, 1992, Humphrey et al 1995)

2 A regression relationship between event FSS load and corresponding event discharge characteristics.

Student *t*-test based on BACIP

For this work the BACIP analysis uses the differences between event FSS loads measured at MCDS and MCUS and between two time periods, to test for significant change between the two intervals. In particular, the difference data distributions observed for events prior to the cyclone (2005–06 wet season) were compared to those events observed post-cyclone (2006–07 and 2007–08 wet seasons). A student *t*-test (assuming equal variances) showed that the differences in the event FSS loads for post-cyclone events during 2006–07 were not significantly different to those for events observed prior to the cyclone (2005–06 data). Similarly, post-cyclone events during 2007–08 were also not significantly different from the 2005–06 event data indicating that the relative difference in event FSS loads between the two stations has not changed as a result of tree damage associated with Cyclone Monica along the Magela Creek channel.

Relationship between FSS load and discharge characteristics

Event FSS load and corresponding event discharge characteristics for MCUS and MCDS collected during the 2005–06 wet season were used to fit a relationship of the form (Moliere & Evans 2009, 2010):

$$\text{Total FSS load} = K(Q_T)^a Ri^b \qquad (1)$$

where Q_T is total discharge during the FSS pulse, Ri is maximum periodic rise in discharge over 10 minutes and a, b and K are fitted parameters.

Observed event FSS loads measured for rainfall/discharge events for each site were then plotted against the predicted FSS loads, derived using equation (1) where Q_T and Ri were applied to the observed discharge for the event for which observed FSS load was measured (Figure 3). The fitted line (1:1) represents the ratio between observed and fitted event loads.

Event FSS load data observed during 2006–07 and 2007–08 at each station were plotted against the loads predicted using the fitted relationships derived for pre-cyclone conditions (Figure 3). Data points that plot above the 1:1 line indicate an observed event load greater than the expected (predicted) load for a particular discharge event and data points below the 1:1 line represent observed FSS loads less than expected for that discharge event. Observed event loads that plot above the +2 standard deviation (SD) line have a significantly higher FSS load than would be expected for the corresponding event discharge characteristics for the pre-cyclone condition.

All FSS load events observed during the first wet season after the cyclone (2006–07), at MCUS and MCDS respectively, plotted above the 1:1 line (Figure 3) even though only a few events at both stations fell outside +2 SD of the fitted relationship. An *F*-test showed that for both sites, a relationship fitted using 2006–07 data was significantly different to that fitted using 2005–06 event data (at the 95% level). This indicates a significant increase in FSS loads relative to discharge characteristics along Magela Creek during 2006–07 compared with that observed prior to the cyclone.

All observed events in 2007–08 (except for one at each station) plotted within +2 SD and around the 1:1 line for the 2005–06 fitted relationship (Figure 3). An *F*-test showed that a relationship fitted using 2007–08 data was not significantly different from the relationship fitted using pre-cyclone (2005–06) event data (at the 95% level). Therefore, event FSS loads for 2007–08 relative to the event discharge characteristics are not significantly different to pre-cyclone conditions at both stations.

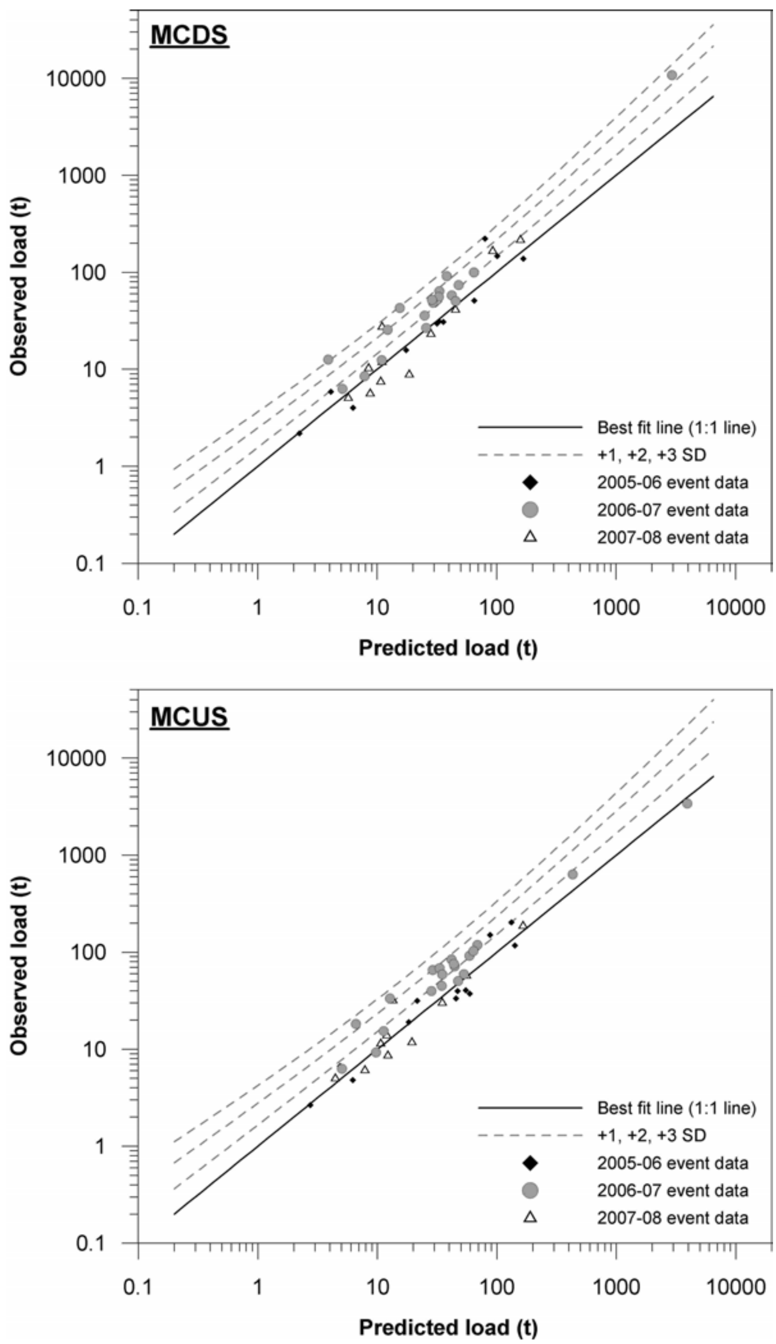

Figure 3 Fitted lines for relationship between observed and predicted event FSS loads and associated +1, +2, and +3 standard deviation lines for MCDS (Top) and MCUS (Bottom). Discrete event data collected during 2005–06 (used to fit the relationships) and the post cyclone (2006–07 and 2007–08) event data are also shown.

This analysis indicates that the effects of Cyclone Monica caused a significant impact on sediment transport characteristics within the catchment during the first year after the cyclone. There was a general increase in stream suspended sediment concentrations and suspended sediment loads relative to discharge measured at both stations along Magela Creek. By the second year after the cyclone (2007–08) the system had returned to pre-cyclone FSS transport conditions.

Comparison of approaches

The student *t*-test using BACIP results shows that there was no significant change in the relative differences in event FSS loads between the two stations as a result of the cyclone. However, using this approach it is not possible to distinguish if there has been a similar change in sediment transport characteristics at both sites, which was indicated to have occurred by the regression model. It is considered that while the BACIP approach may be a good method to assess an impact that occurs *between* the two stations (i.e. a mining-related impact), it is not an appropriate method for assessing the impact of a catchment-wide impact (such as cyclone, fire or extreme flood event) where both upstream and downstream sites are significantly and similarly impacted.

The regression approach detected a drainage system (ie broad scale catchment) impact since suspended sediment concentrations and loads significantly increased along Magela Creek as a result of the cyclone. This approach also clearly illustrated the subsequent recovery of the system by returning of the fitted regression line (for 2007–08 data) to its original position, indicating that sediment transport in the creek system had returned to pre-cyclone conditions. It is likely that much of sediment made available by treefall was washed away during the major flood event in late February and early March 2007 (Moliere et al 2008). This event completely inundated the terraces and palaeochannels alongside the main channel where most of the treefall occurred.

Application of the regression analysis method has shown that riparian zone treefall associated with a Category 2 cyclonic event can have a significant, although relatively short-lived. effect on sediment transport characteristics within a catchment,.

This was the only progress made on analysis of the Cyclone Monica data base this year due to changes in ***eriss*** staff structure and competing priorities with the trial landform. However, all field survey data and soil survey data were published as SSD Internal Reports in March 2009 (Saynor et al 2009a, b).

Future work

Future work will include finalising the statistical analysis of impacts on vegetation and sediment transport rates in the Gulungul Creek catchment, and other sites where data were collected such as Ranger and Nabarlek. No further work will be specifically undertaken to assess impacts of Cyclone Monica on sediment loads in Magela Creek. Monitoring will continue in the ‘Turbidity and suspended sediment management guidelines and trigger values for Magela Creek’ project under KKN 2.5.2 (Off-site monitoring during and following rehabilitation) – and ongoing BACIP and regression analysis analyses undertaken to detect any minesite impacts and to track any systematic changes in the overall sediment delivery behaviour of the catchment.

References

Humphrey CL, Faith DP & Dostine PL 1995. Baseline requirements for assessment of mining impact using biological monitoring. *Australian Journal of Ecology* 20, 150–166.

Moliere DR & Evans 2009. Cyclone Monica and the resulting impacts of treefall on stream sediment transport. In *Adapting to change*, Proceedings of the 32nd Hydrology and Water Resources Symposium, 30 November – 3 December 2009, Newcastle, Engineers Australia (CD).

Moliere DR & Evans KG 2010. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research* DOI: 10.1111/j.1745-5871.2010.00641.x

Moliere DR, Evans KG & Turner KF 2008. Effect of an extreme storm event on catchment hydrology and sediment transport in the Magela Creek catchment, Northern Territory. In *Water Down Under 2008, Proceedings of the 31st Hydrology and Water Resources Symposium and 4th International Conference on Water Resources*, 15–17 April 2008. Adelaide, South Australia, Engineers Australia (CD).

Moliere DR, Saynor MJ & Evans KG 2005. Suspended sediment concentration-turbidity relationships for Ngarradj – a seasonal stream in the wet-dry tropics. *Australian Journal of Water Resources* 9(1), 37–48.

Saynor MJ, Houghton R, Hancock G, Staben G, Smith B & Lee N 2009b. Soil sample descriptions – Gulungul Creek, Ranger mine site and Nabarlek: Cyclone Monica fieldwork. Internal Report 558, Supervising Scientist, Darwin. Unpublished paper.

Saynor MJ, Staben G, Hancock G, Fox G, Calvert G, Smith B, Moliere DR & Evans KG 2009a. Impact of Cyclone Monica on catchments within the Alligator Rivers Region – Data. Internal Report 557, Supervising Scientist, Darwin. Unpublished paper.

Staben GW & Evans KG 2008. Estimates of tree canopy loss as a result of Cyclone Monica, in the Magela Creek catchment northern Australia. *Austral Ecology* 33, 562–569.

Stewart-Oaten A, Bence JR & Osenberg CW 1992. Assessing effects of unreplicated perturbations: No simple solutions. *Ecology* 73, 1396–1404.

Stewart-Oaten A, Murdoch WW & Parker KR 1986. Environmental impact assessment: ‘Pseudoreplication’ in time? *Ecology* 67, 929–940.

Assessment of suspended sediment movement upstream and downstream of Ranger

K Evans & D Moliere[1]

Background

Event loads of fine suspended sediment (FSS) obtained during the three wet seasons from 2005–06 to 2007–08 from upstream (MCUS) and downstream (MCDS) of Ranger along Magela Creek were used to derive proposed guideline trigger values for suspended sediment. The trigger values were based upon attributes of the statistical distribution of FSS data derived from the creek in periods of no net inputs from the mine (ie a 'reference' condition), using two methods (1) using a Before-After-Control-Impact-Paired differences (BACIP) design and (2) using regression analysis. The results were reported at ARRTC 22 (see Moliere et al 2007).

Progress

In 2008–09 the above analysis was written up as an SSD internal report and a journal paper. The internal report is being reviewed and the paper has been accepted for publication in *Geographical Research* (Moliere & Evans 2010).

The 2005–08 data set has also been used to analyse the likely impact that Cyclone Monica (2005–06 wet season) had on suspended sediment transport in Magela Creek in the two wet seasons following this event. The findings from this work are reported in this volume (see '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', pp 179–184, in this volume).

Continuous rainfall, stream discharge, electrical conductivity and turbidity data were collected for the 2008-09 wet season (Figure 1) but have not yet been analysed or reported due to staffing changes and competing priorities.

Future work

Continuous monitoring data will continue to be collected. The data are used to update and revise the trigger values, and to detect and assess possible mine-associated changes in FSS in Magela Creek, using a combination of BACIP and regression model analysis. Work to date has identified one possible mine-related change in FSS event loads measured downstream of Ranger since 2005. It is recommended that impact assessment of FSS loads continue to use both statistical approaches, given their complementary strengths and weaknesses (see 'impact of Cyclone Monica' paper, pp 179–184, in this volume, for a description of how the two approaches may be applied).

[1] Water Division – Bureau of Meterology, NT Regional Office

Future measured event load data which plot above the 'action' trigger levels derived from these methods of sediment load analysis should prompt further investigation and management action, if required. The investigation should include chemical analysis of the suspended sediment to ascertain if it has come primarily from the minesite or from adjacent un-impacted subcatchment areas. This distinction needs to be made since elevated sediment loads could arise from an intense local storm event on land, unassociated with mining activity itself, yet lying within the mine lease boundary. Water samples for this analysis will be provided by the event-triggered autosamplers located at the downstream site in Magela Creek.

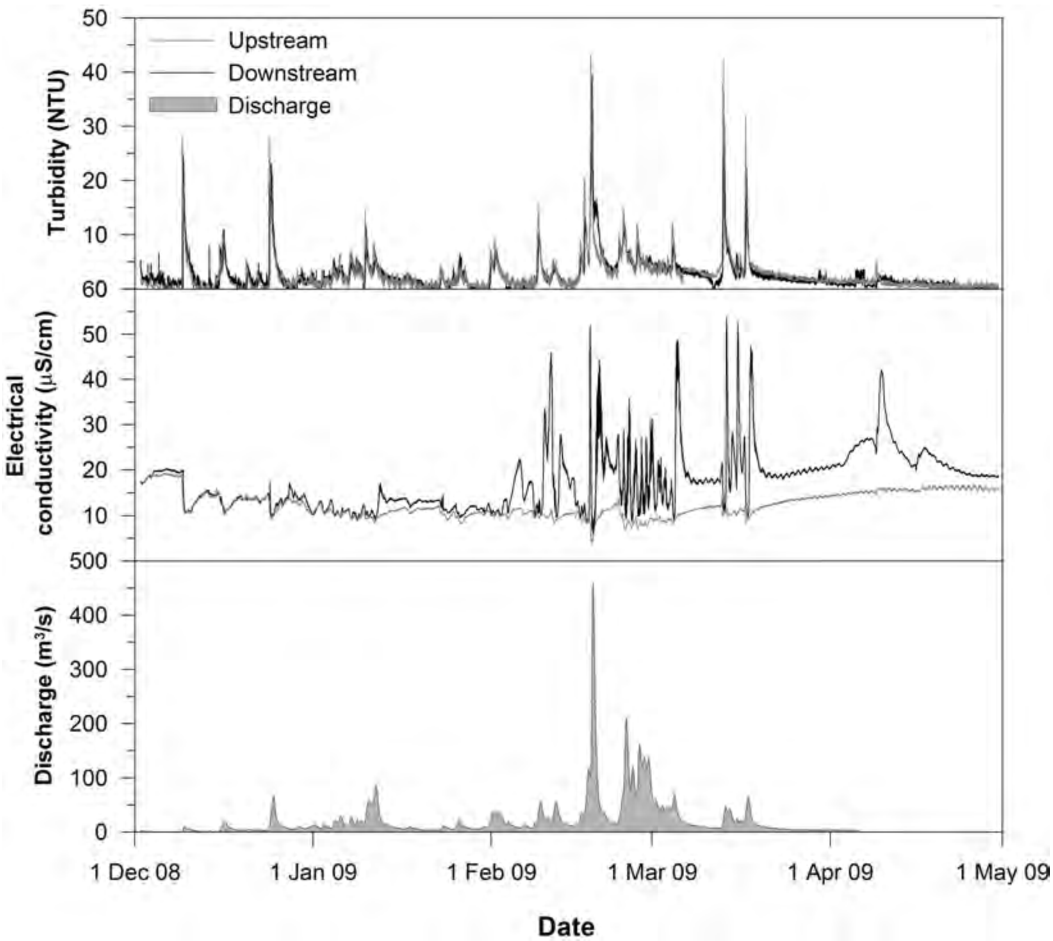

Figure 1 Magela Creek turbidity and electrical conductivity measured upstream and downstream of the mine and discharge measured at G8210009

References

Moliere DR & Evans KG 2010a. Development of trigger levels to assess catchment disturbance on stream suspended sediment loads in the Magela Creek, Northern Territory, Australia. *Geographical Research* DOI: 10.1111/j.1745-5871.2010.00641.x

Moliere D, Evans K & Turner K 2007. Assessment of continuous Magela Creek turbidity data upstream and downstream of Ranger. In ***eriss*** *research summary 2006–2007*. Supervising Scientist Report 196, Supervising Scientist, Darwin NT, 124–130.