Energy Resources of Australia Ltd

5. Knowledge base

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GLOSSARY

Below are key terms that are used in this section.

Key term	Definition
Bioregion	An ecologically and geographically defined area that is smaller than a biogeographical realm ,but larger than ecoregion or an ecosystem, in the World Wildlife Fund classification scheme.
Becquerels	The Becquerel (Bq) is the SI derived unit of radioactivity. One Becquerel is defined as the activity of a quantity of radioactive material in which one nucleus decays per second.
Constituents of Potential Concern	Chemical elements identified by the Supervising Scientist Division as being of potential concern to the receiving environment
Electrical conductivity	Abbreviated to EC. Electrical conductivity is a measure of how well a material accommodates the transport of electric charge.
Gamma Radiation	Ionizing electromagnetic radiation emitted by a radionuclide during radioactive decay
Gray	The Gray (Gy) is a SI derived unit of ionizing radiation dose. One Gray is defined as the adsorption of one joule of radiation energy per kilogram of matter.
Hydrolithologic Unit	A grouping of soil or rock units or zones based on common hydraulic properties.
Georgetown Billabong	The statutory surface water monitoring point for Georgetown Billabong, which is located downstream of Corridor Creek and the Corridor Creek wetland filter.
Groundwater conceptual model	Calibrated numerical groundwater flow model encompassing all hydrogeologic elements governing groundwater flow and transport at the Ranger Mine to provide the foundation for simulating groundwater flow and transport from all mine sources to potential receptors under post-closure conditions.
Land Application Area(s)	Abbreviated to LAA. An area on the RPA used as an evapotranspiration disposal method polished and unpolished pond water from the constructed wetlands filters and, more recently, permeates from the water treatment plants. However, irrigation of unpolished pond water ceased at the end of 2009. The concept of land application is to retain metals and radionuclides in the near-surface soil profile.
Land Disturbance Permit	An ERA permit required prior to undertaking any work on the RPA that may lead to surface disturbance, for example ground breaking, surface disturbance, clearing etc.
Long Lived Alpha Activity	Abbreviated to LLAA. The presence, generally in airborne dust, of any of the alpha emitting radionuclides in uranium ore, except for the short-lived alpha emitting radon decay products.
MBL Zone	A hydrolithologic zone of relatively higher permeability to the south east of Pit 1 identified through testing and pumping of bore MB_L.
Magela Creek downstream	Abbreviated to MG009. MG009 is Ranger downstream statutory or compliance surface water monitoring point. It is located on the Magela Creek, downstream of Ranger operations.
Magela Creek upstream	Abbreviated to MCUS. MCUS is the upstream statutory surface water monitoring point, location on the RPA.

Key term	Definition
Mirarr	Mirarr is a patrilineal descent group. Descent groups are often called 'clans' in English and kunmokurrkurr in Kundjeyhmi language. There are several Mirarr clans with each one distinguished by the language they historically spoke (e.g. Mirarr Kundjeyhmi, Mirarr Urningangk, Mirarr Erre). The Mirarr are the Traditional Owners of the land encompassing the RPA.
Minesite Technical Committee (MTC)	A of the Working Arrangements for the Regulation of Uranium Mining in the Northern Territory dated 30 May 2005, is tasked with: Reviewing proposed and existing approvals and decisions under NT legislation Reviewing technical information in relation to Ranger Mine, including monitoring data and environmental performance Collaboratively developing standards for the protection of the environment Developing strategies to address emerging issues The MTC consists of the representatives of the Department of Industry, Tourism and Trade, the Supervising Scientist, ERA and the Northern Land Council. Representatives of the Commonwealth Department of Industry, Science, Energy and Resources may also attend MTC meetings.
Pit 1	The mined out pit of the Ranger #1 orebody, which is used as a tailings repository. Mining in Pit 1 commenced in May 1980 and was completed in December 1994, after recovering 19.78 million tonnes of ore at an average grade of 0.321%.
Pit 3	The mined out pit of the Ranger #3 orebody, which is currently being backfilled with tailings. Open cut mining in Pit 3 commenced in July 1997 and ceased in November 2012.
Plant Available Water	Abbreviated to PAW. The amount of water that can be stored in a soil and be available for growing crops.
Processing	Processing is the mining term to describe all phases of the ore treatment from milling through to the final product packaging of uranium oxide.
Radon decay products or radon progeny	The short-lived radioactive decay products of radon-222. This includes the decay chain up to, but not including lead-210, namely polonium-218 (sometimes called radium A), lead-214 (radium B), bismuth-214 (radium C) an dpolonium-214 (radium C).
Ranger Project Area	Abbreviated to RPA. The Ranger Project Area means the land described in Schedule 2 to the Commonwealth *Aboriginal Land Rights (Northern Territory) Act 1976.*
Reference level	Abbreviated to RL. Denotes a specific elevation relative to mean sea level and is regularly used to identify the height or depth of plan or mine infrastructure – e.g. the height of the TSF or depth of Pit 3.
Retention Pond	A large constructed storage facility that collects runoff and stores pond water for treatment (RP2 & RP6) or release water post-treatment (RP1).
Sievert	The Sievert is the unit of absorbed radiation dose, taking into account the differing biological effects of different types of radiation.
Tailings dam	Surface dam used to hold tailings and process water at Ranger. Commonly referred to as "tailings storage facility" or "TSF" in other ERA material. The tailings dam is one of currently three tailings storage facilities at Ranger, the others being Pit 1 and Pit 3.

Key term	Definition
U_3O_8	The most stable form of uranium oxide and the form most commonly found in nature. Uranium oxide concentrate is sometimes loosely referred to as yellowcake. It is khaki in colour and is usually represented by the empirical formula U_3O_8. Uranium is normally sold in this form.
Waste rock	The mineral waste produced in the mine but is stockpiled due to its low grade i.e. material which does not enter the processing plant. For example, 1s waste rock is typically material that has a grade of less than 0.02% U_3O_8; 2s waste rock (or low-grade ore) is typically material that has between 0.02% and 0.12% U_3O_8.
Wetland filter	A constructed biological filter system that is designed for final treatment of release water and is monitored to ensure water quality meets regulatory criteria for disposal.

ABBREVIATIONS & ACRONYMS

Below are abbreviations and acronyms that are used in this section.

Abbreviation/ Acronym	Description
AHD	Australian Height Datum
ALARA	As low as reasonably achievable
ARRAC	Alligator Rivers Region Advisory Committee
ARRTC	Alligator Rivers Region Technical Committee
BC	Brine Concentrator
BOM	Bureau of Meteorology
BTV	Background Threshold Value
CCWLF	Corridor Creek Wetland Filter
COPC/COPCs	Constituent of Potential Concern/ Constituents of Potential Concern
CPT	Cone Penetration Test
CSM	Conceptual Site Model
DEM	Digital Elevation Model
DITT	Department of Industry, Tourism and Trade
DPIR	Department of Primary Industry and Resources (now DITT)
EC	Electrical conductivity
ECVs	Environmental and Community Values
EDZ	Excavation-damaged zone
EIS	Environmental Impact Statement
EPBC Act	*Environment Protection and Biodiversity Conservation Act 1999*
EPIP Act	*Environmental Protection (Impact of Proposal) Act 1974*
ER	Environmental Requirements
ERA	Energy Resources of Australia Ltd
ERISS	Environmental Research Institute of the Supervising Scientist
ET	Evapotranspiration
GAC	Gundjeihmi Aboriginal Corporation
GCBR	Georgetown Creek Brockman Road
GCMBL	Georgetown Creek Mine Bund Leveline
GDE	Groundwater Dependent Ecosystem
GTB	Georgetown Billabong
HDS	High Density Sludge
HLU	Hydrolithologic Unit
HDPE	High-density Polyethylene

Abbreviation/ Acronym	Description
ISWWG	Independent Surface Water Working Group
ITWC PFS	Integrated Tailings, Water and Closure Prefeasibility Studies
KKNs	Key Knowledge Needs
LAA	Land Application Area
LAI	Leaf Area Index
LEM	Landform Elevation Model
MCP	Mine Closure Plan
MTC	Minesite Technical Committee
NAQS	Northern Australia Quarantine Strategy
NLC	Northern Land Council
NSMC	Null space Monte Carlo
NP	National Park
NT	Northern Territory
OBS	Osmoflow Brine Squeezer
QQ plot	Quantile-quantile Plot
R3D	Ranger 3 Deeps
RCM	Ranger Conceptual Model
RL	Reference Level
RP1	Retention Pond 1 – also denotes other retention ponds used on site – e.g. RP2, RP3, RP6
RPA	Ranger Project Area
RPC	Release Plan Calculator
PAW	Plant Available Water
PEST	Parameter Estimation Tool
PDF	Probability Distribution Function
PTF	Pit Tailing Flux
RSWM	Ranger Surface Water Model
SAQP	Sampling Analysis Quality Plan
SSB	Supervising Scientist Branch
TAN	Total Ammoniacal Nitrogen
TLF	Trial Landform
TPM	Total Particulate Metals
TPWS Act	*Territory Parks and Wildlife Conservation Act 1978 (NT)*
TSF	Tailings Storage Facility
TSS	Total Suspended Solids

Abbreviation/ Acronym	Description
UNESCO	United Nations Educational, Scientific and Cultural Organisation
VAF	Vulnerability Assessment Framework
WRD	Water Resources Division
WTP	Water Treatment Plant

5 KNOWLEDGE BASE

The following section provides an overview of the environmental setting of the Ranger Mine, and a summary of completed and planned studies informing the closure implementation strategy. The section provides the context to planning mine closure and is a summary of a substantial knowledge base that has been accumulated by Energy Resources of Australia Ltd (ERA) and stakeholders from more than 30 years of monitoring and research investigations of the site and surrounding environment.

5.1 Social setting

5.1.1 Aboriginal culture and heritage

There is recent evidence of Aboriginal occupancy of the Kakadu region dating back more than 65,000 years.[2] Central to closure planning are the Mirarr people who are the Traditional Owners of the land encompassing the Ranger and Jabiluka mineral leases. In addition to the mineral leases, Mirarr country extends to the town of Jabiru and parts of Kakadu National Park (NP), including the wetlands of the Jabiluka billabong country and the sandstone escarpment of Mount Brockman.

Prior to the 19th Century, the Kakadu region had a population of approximately 2,000. However, the population experienced a rapid decline from the late 19th Century to the early decades of the 20th Century (Taylor, 1999). This was, in part, as a result of European missionary activity, which encouraged a dispersal of the population, and large-scale military activities during the Second World War. At the time of initial uranium exploration at the Ranger deposit in the 1970s, only 44 indigenous Australians were counted as residing in the area in the 1976 Australian Bureau of Statistics Census (cited in Taylor, 1999).

The establishment of the town of Jabiru to service the uranium mining industry was, and remains, a significant factor in the increase in population in the region since the late 1970s. The extent to which the indigenous population has varied during this period is difficult to ascertain due to a paucity of reliable data.

The RPA contains several significant Aboriginal sites, including two recorded sacred sites which lie within designated 'restricted work areas'. One site is located approximately 5 kilometres north of the mine. The second sacred site, Tree Snake Dreaming, is situated north of Pit 3 and access into the vicinity for operational activity is required on very infrequent occasions. Both sites are listed with the Aboriginal Areas Protection Authority and a Site Management Plan is in place to ensure ongoing protection.

A third site of indigenous cultural heritage significance in the RPA is a cemetery where a small number of local Aboriginal people are buried; this was established prior to mining exploration. This is not a gazetted cemetery and the burials were contemporary for the period rather than

[2] ABC News, 20 July 2017: http://www.abc.net.au/news/science/2017-07-20/aboriginal-shelter-pushes-human-history-back-to-65,000-years/8719314

being Traditional Aboriginal burials. There are also restricted work areas on the RPA boundary for two sacred sites that occur outside, but adjacent to, the RPA.

Cultural heritage surveys over the RPA since 2006 have covered 73 percent of the RPA and recorded 99 archaeological sites and 69 archaeological background scatters. There are a total of 171 recorded places of indigenous cultural heritage significance in the RPA. One such site (R34), is located adjacent to Pit 3 and is protected within a fenced exclusion zone.

5.1.2 World heritage listing attributes

The attributes of the Kakadu NP must not be compromised by the closure and rehabilitation of the RPA. The Kakadu NP was listed under the World Heritage Convention for five of a possible ten criteria, incorporating both cultural and natural attributes (UNESCO 2019). Criterion (i) and (iv) related to the cultural attributes and are discussed in Section 5.3.2.1.

5.2 Physical environment

With increasing contact between the region's Aboriginal people and other cultures from around the 17th century and a more permanent non-indigenous presence evident from the late 1800s (ERA 2014b). Historical land use within the Alligator Rivers Region has included indigenous occupation, buffalo hunting, missions, pastoral grazing, agriculture, mining exploration, uranium mining and tourism (Levitus 1995). The Magela catchment within the region (Figure 5-1) currently contains several land use types, including Kakadu NP, mining and native title lands. The catchment is largely within Kakadu NP, a World Heritage listed area and Ramsar site (Figure 5-2).

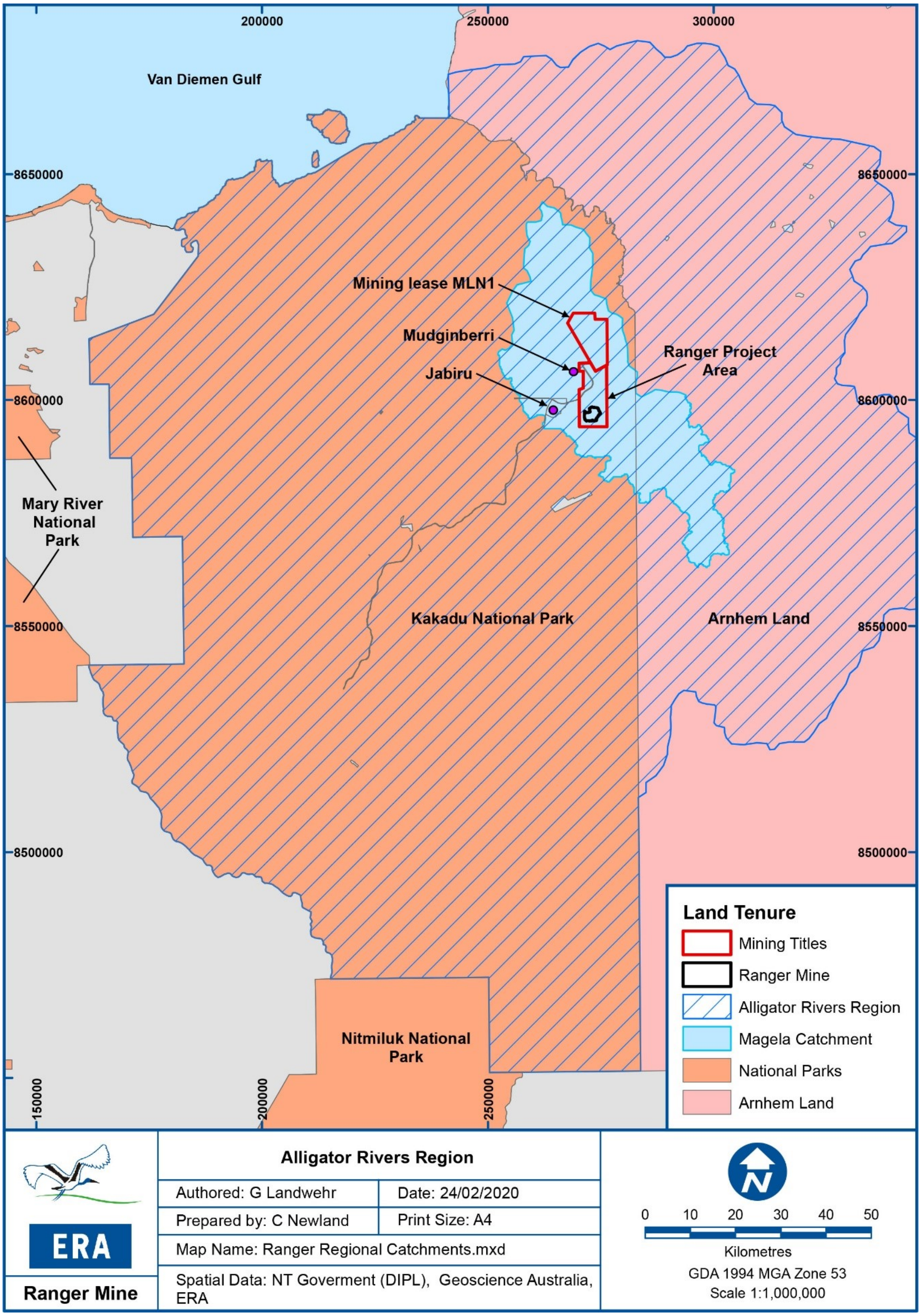


Figure 5-1 Geographic context for closure activities

ERA

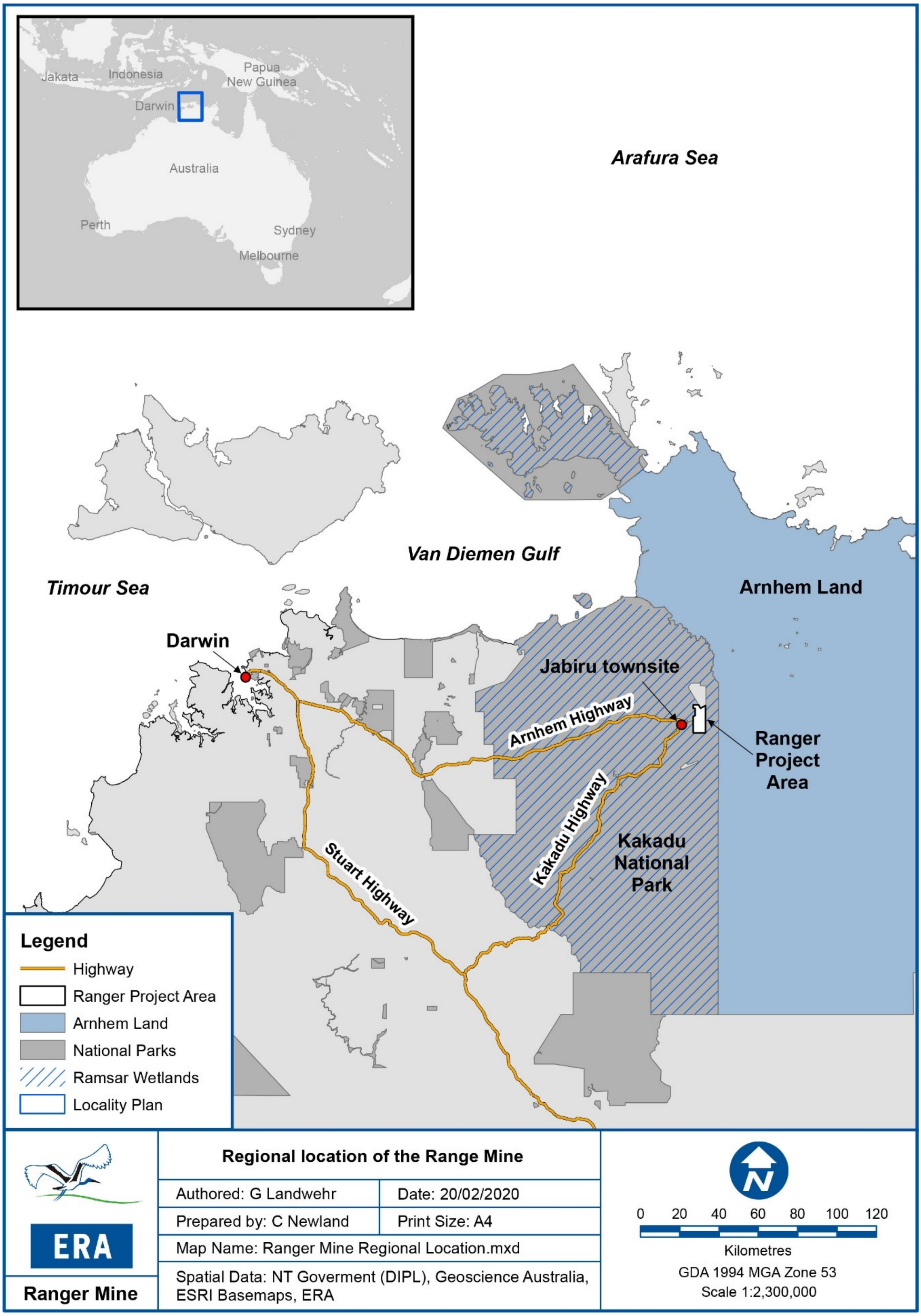


Figure 5-2 Regional location of the Ranger Mine

5.2.1 Climate

The climate of the Alligator Rivers Region, within which the Ranger Mine is located, is dominated by a seasonal wet-dry monsoon cycle with the large inter-annual and intra-seasonal variability largely associated with the effects of the El Niño Southern Oscillation, the Madden-Julian Oscillation and tropical cyclone activity (Trenberth *et al.* 2007). The wet season generally extends from late October to early April with predominantly westerly winds, whilst the dry season is dominated by easterly to south-easterly winds and extends from May to September. Historical climatic conditions for the Ranger Mine area are presented in Table 5-1.

The tropical cyclone season in northern Australia typically extends from November to April, averaging around two cyclones a year, with peak activity from December to March (BOM 2019a). Increased cyclone activity in the Australian region has been associated with La Niña years, whilst below normal activity has occurred during El Niño years (Kuleshov & de Hoedt 2003, Plummer *et al.* 1999). When cyclones and tropical lows are present, the Alligator Rivers Region can experience high winds and rainfall.

The region has a hot climate, with mean maximum temperatures ranging from just under 32 °C in June and July to just under 38 °C in October (BOM 2019b). Average monthly pan evaporation ranges from 295 mm in October to 160 mm in February (Chiew & Wang 1999). Annual pan evaporation exceeds rainfall by approximately 1,000 mm.

Table 5-1: Historical weather data, Jabiru Airport

Parameter	Value	Month
Mean maximum temperature	37.7 °C	October
Mean minimum temperature	18.7 °C	July
Maximum average daily evaporation*	9.5 mm	October
Minimum average daily evaporation*	5.6 mm	March
Annual average daily evaporation*	7.2 mm	-
Annual rainfall	1,565 mm	-
Annual evaporation*	2,628 mm	-

Source BOM 2019b

**data available for 1973-1990 only*

5.2.2 Land systems

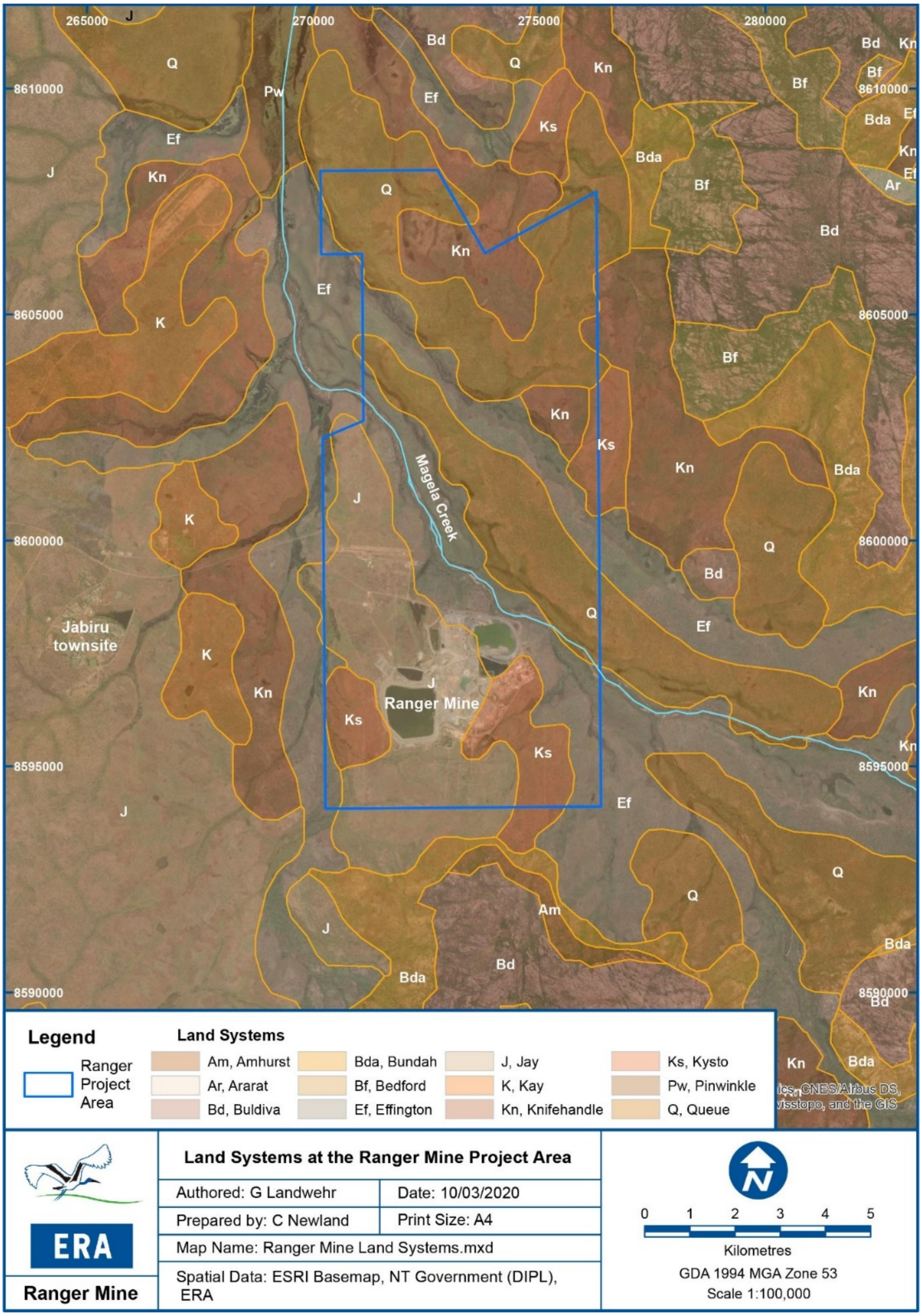


Figure 5-3 Land Systems at the RPA

5.2.3 Topography

The Ranger Mine lies on plains to the north of the Mount Brockman Massif, which is an outlier of the Arnhem Land Plateau. These plains are generally flat with numerous swamps and are rarely more than 45 m above sea level. South and east of Ranger Mine, the Arnhem Land Plateau escarpment rises to between 200 and 300 m above sea level. A major feature of the landscape is Mount Brockman, which rises 170 m above the plain, approximately 3.5 km south of Ranger Mine (Figure 5-4).

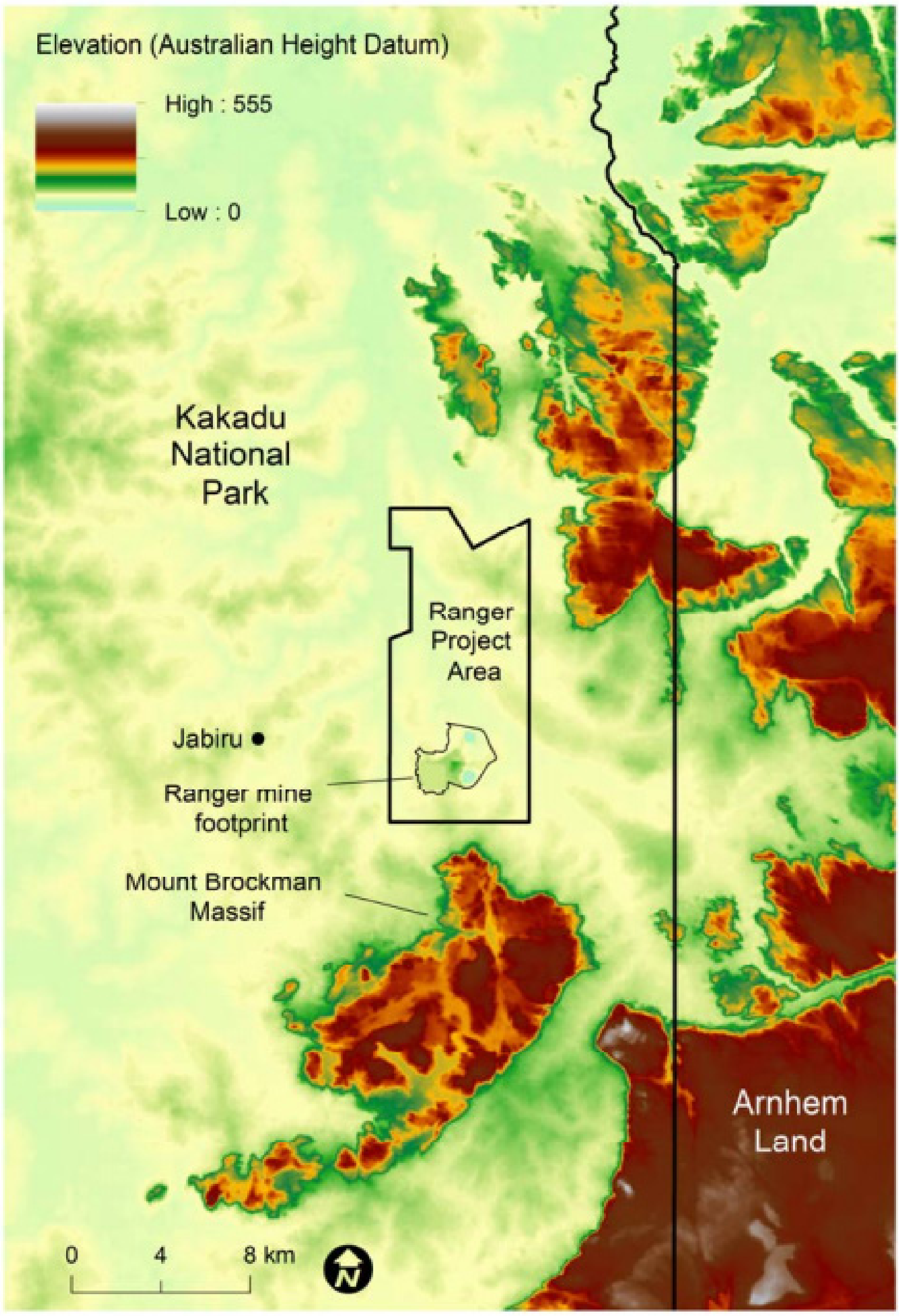


Figure 5-4: Elevation of RPA and the surrounding region

The Ranger Mine is influenced to varying degrees by the following four land surface categories:

- The Mount Brockman Massif is a massive quartz sandstone outlier. Its steep escarpment and skeletal soils constitute part of the watershed of the Magela and Gulungul creek systems. Due to its resistance to erosion and low soil moisture retaining capacity, a large volume of localised rainfall is readily accumulated in the surface drainage networks and causes rapid flood responses in creeks and drainage lines. Water infiltrates joints and fissures, contributing to groundwater recharge and the formation of springs and swamps, some of which continue to discharge well into the dry part of the year, many months after the last rainfall.
- The Koolpinyah Surface, corresponding to the plains on which the Ranger Mine is located, is characterised by level, rolling or dissected lowlands. The surface consists of deeply weathered bedrock partly overlain by Late Tertiary to Recent sediments derived from the erosion of Cretaceous, Middle Proterozoic and Lower Proterozoic formations. These are mantled by ferruginous soils and ferricrete crusts.
- Alluvial plains have been formed by the flow of numerous rivers across the Koolpinyah Surface. The Magela and Gulungul Creeks flow in a northerly direction from the Mount Brockman Massif and dissect the RPA. Alluvial materials have been deposited by these creek systems to form the flat Magela floodplains to the northwest. Coarse, sandy Late Tertiary and Quaternary alluvial deposits cover part of the plains. These occupy channels of diverted streams and anabranches.
- Coastal plains extend north of the Koolpinyah Surface. These are flat, poorly drained and penetrate far inland along the broader river valleys.

5.2.4 Soils

The type (class) and distribution of soils across the land surfaces of the Ranger Project Area (RPA) are influenced by geology, topographic position and seasonal changes to the amount of moisture in the ground (Story *et al.* 1969, Chartres *et al.* 1991 and Hollingsworth *et al.* 2005). The four main geomorphic units have particular associated soil types, which in turn influence vegetation assemblages.

Colour variation in the soils is primarily a product of differential drainage and the resulting mineralogy of the component iron oxyhydroxides. Stony layers within the soil profile may represent the boundary between residual and non-residual (e.g. transported) materials.

Soils are non-saline and non-sodic and can be gravelly, with clasts of quartz, ferricrete and ferruginised rock. Kaolinitic minerals are common and illite, together with minor chlorite, can be inherited from underlying Cahill Formation schists (see also Section 5.2.5). The cation exchange capacity is generally moderate to low in the near-surface horizons and there are low levels of organic materials and nutrients. Table 5-2 provides a brief description of the soil characteristics associated with the Ranger Mine, which are also depicted in Figure 5-6.

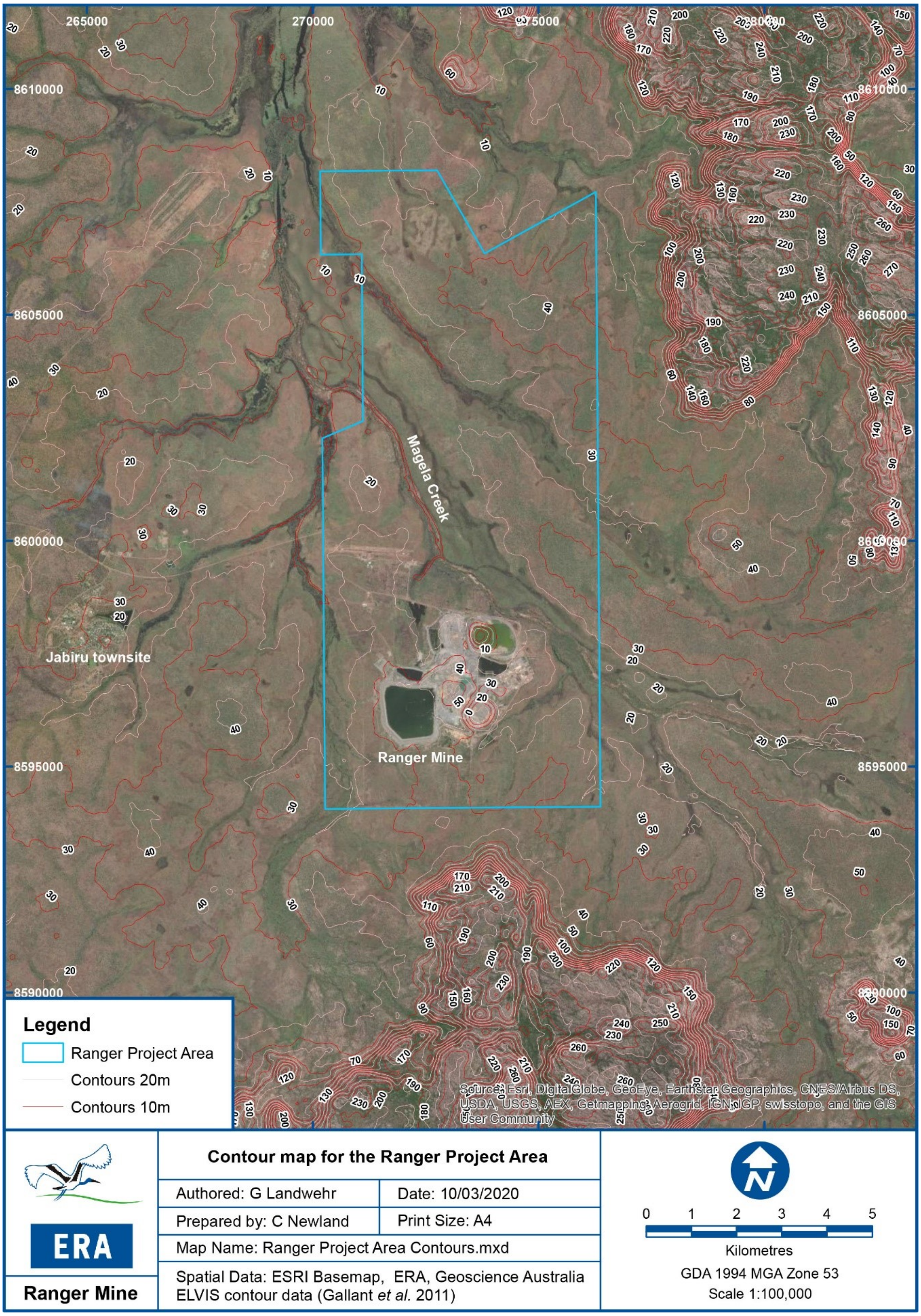


Figure 5-5 Contour map of the RPA and surrounds

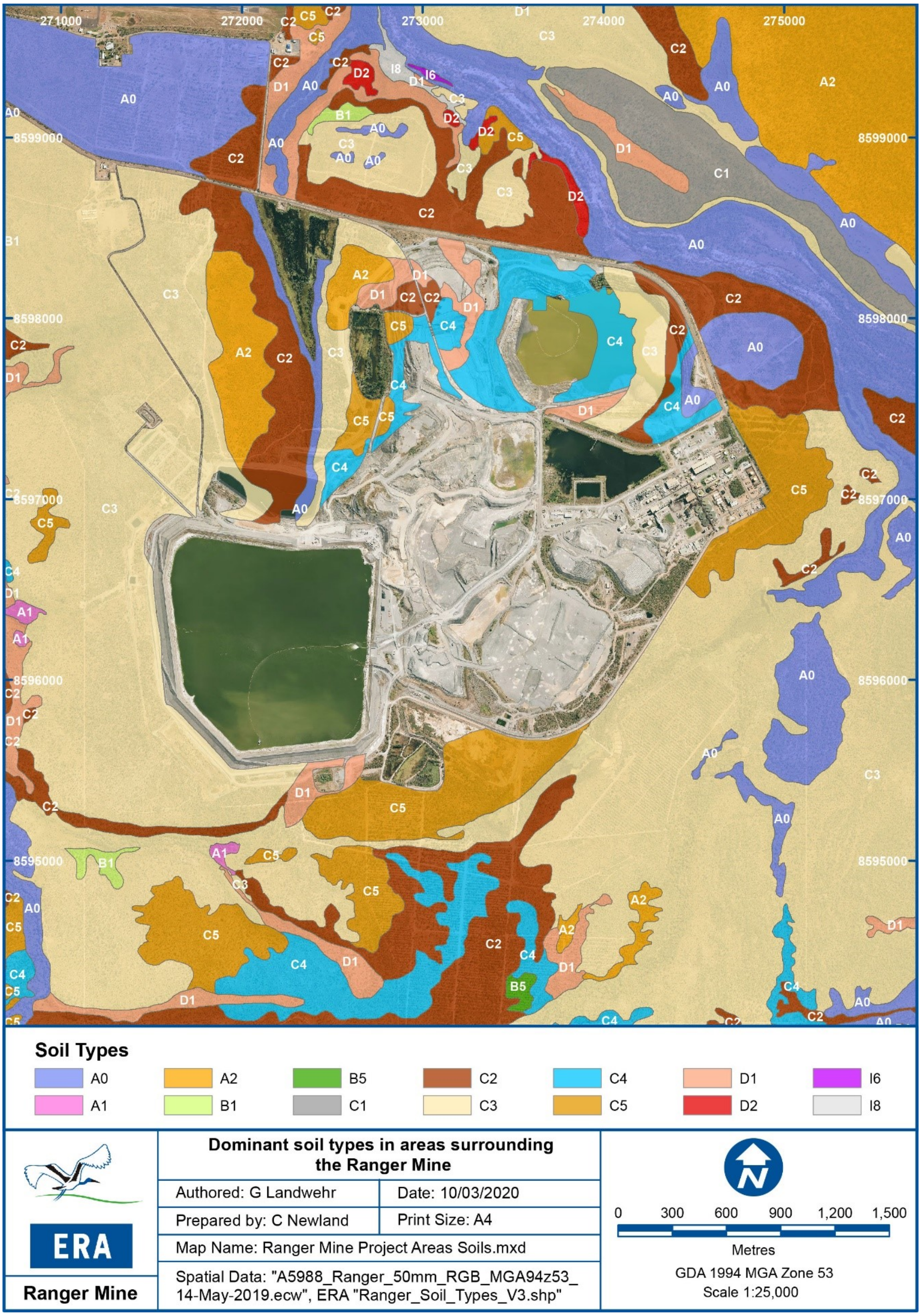


Figure 5-6: Dominant soil types in areas surrounding the Ranger Mine

Table 5-2: Brief description of soil characteristics around the Ranger Mine (Figure 5-6)

Map unit (Hollingsworth, 1999) (refer Figure 5-6)	Map unit description
A0	Organic horizon, sand/loamy surface.
A1	Deep pale brown, yellow and yellowish brown sands, sand/loamy sand surface and generally non-mottled single grained and sandy throughout. Variations include: light yellowish brown and dark brown; and yellow brown, yellow and faint red brown mottles.
A2	Deep yellowish brown to very pale brown; highly permeable, generally non-coherent sand, bottoming onto ferruginous and quartz gravel and stone. Profiles may vary: depths may extend from 100 cm; *in situ* gravels may occur within the lower horizons and the firm clay clod nodules may become hard; 10-15 mm, prominent, red mottles.
B1	Deep brownish yellow to yellowish brown massive gravel-free earthy sands with minor mottles common at depth. Profile variations include different degrees of mottles at depth, and on rare occasions, overlie a buried zone.
B5	Shallow, gravelly, brown to yellowish brown, massive, earthy sands. Variations may have light brownish yellow and minor light grey horizons at depth, textures may not be heavier than loamy sands.
C1	Moderately deep to deep yellowish brown to light yellowish brown, sandy earths with no gravel present. No profiles bottom onto laterite pavement and gravel pans. Profiles may be deeper, lighter in chroma and increasing in texture to sandy light clay.
C2	Moderately deep to deep sandy loams over a gravel pan.
C3	Moderately deep to deep, dark yellowish brown to yellowish brown, sandy earths with gravel throughout, bottoming onto ferruginous gravel.
C4	Shallow yellowish brown to brownish yellow sandy earths bottoming onto dense ferruginous gravel and stone. Mottles may occur. Variations include distinct, grey and prominent, red mottles in B-horizon.
C5	Shallow brown to yellowish brown gravelly sandy earths over a ferruginous and quartz gravel pan. Variations include colours to yellowish brown; depth varying to 30 cm; and gravel contents ranging between 5% and 50% within the profile.
D1	Deep light brownish grey to grey loamy earths, massive.
D2	Deep to moderately deep yellowish brown to pale brown gravel-free loamy earths over a gravel/stone hardpan. Variations include textures to coarse sandy clay at depth; colours from pale brown to grey; and mottles where sites are ponded.
I6	Deep profiles of grey to brown sands and earthy sands over a generally mottled light grey to pale brown clay and sandy clays.
I8	Profiles are very dark grey to greyish brown loamy earths and sandy earths over a brown to pale brown earthy sand, with mottles common. Considerable variation was found with all soil characteristics.

Field investigations of soil hydraulic conductivity (Table 5-3) have identified that individual soil horizons range from very permeable, on account of the presence of naturally occurring piping, to impervious. The A and B horizons support a shallow, unconfined surface aquifer that overlays a low conductivity C horizon (Hollingsworth 1999). This unit is underlain by an impervious unfractured bedrock D horizon. The unconfined aquifer is observed to recharge both the A and B horizons during the wet season, to the point where water expresses as baseflow in lower areas of the topography and drainage lines. During the dry season, the upper A and B soil horizons can be entirely dry down to the confining C horizon.

Hydraulic conductivities in the A and B horizons can range from 0.01 to 10 m/day (Chartres *et al.* 1991), whilst the range of hydraulic conductivities of underlying confining C and D horizons are indicative of low transmissive hydrolithologic units (INTERA 2016).

Table 5-3: Soil hydraulic conductivity

Horizon	Hydraulic conductivity, K
Alluvial sands and 'A' horizon	10 to 1 m/day
Bleached zone 'B' horizons	1 to 0.1 m/day
Saprolite 'B' horizon	2 to 0.01 m/day
Fractured rock 'C' horizon	0.1 to 0.001 m/day
Unfractured rock 'D' horizon	0.05 to 0.001 m/day

Depending on vegetation cover and the presence or absence of a surface rock lag, erosion is highly seasonal and is dominated by sheet erosion in the wet season. At the beginning of the wet season, understorey cover can be sparse due to preceding dry season conditions and vegetation loss due to fire. The variability of vegetation cover contributes to the impact of rain splash erosion. Where grasses and leaf litter remain, these assist in protecting the soil from early wet season rain splash erosion. However, as rainfall intensifies with the development of monsoonal troughs, other erosion processes become dominant including floods, sheet flow runoff, high winds and cyclones. Overland sheet flow, and gully erosion by streams increase and are particularly severe in areas where vegetation is disturbed. Further detail on these erosion processes are provided in Table 5-4.

5.2.5 Geology and mineralisation

The Ranger uranium deposits are located in the East Alligator region of the Paleoproterozoic Pine Creek Inlier. Mineralisation is contained in chlorite-altered metasediments of the Lower Cahill Formation (age approximately 1,870 million years) which overlie an older basement complex of Archaean granitoid gneisses and schists known as the Nanambu Complex (age approximately 2,470 million years). Unconformably overlying rocks of both the Lower Cahill Formation and the Nanambu Complex are sandstones and conglomerates of the Kombolgie Sandstone (age approximately 1,650 million years) which forms part of the Katherine River Group of the McArthur Basin.

Uranium mineralisation occurs within a northerly trending and gently easterly-dipping belt of Lower Cahill metasediments, directly east of the Nanambu Complex (Figure 5-7). The Lower Cahill Formation has been informally subdivided into three units. All uranium ore occurs in chlorite schists referred to as the Upper Mine Sequence schists. These overlie a sedimentary sequence dominated by carbonates and dolomites (Lower Mine Sequence) and are themselves overlain by mica schists with local horizons of amphibolite (Hanging Wall Schists), as shown in Figure 5-7

Table 5-4: Typical erosion susceptibility of soils

Soil type	Erosion potential
Deep siliceous sands lacking structure	Vulnerable to rain splash and overland flow erosion but are less vulnerable if covered by vegetation
Red earths well drained with good structure	Characteristic of areas with minimal erosion
Yellow earths less well drained than the red earths	More erodible, particularly if dispersive
Duplex soils with texture contrast and massive impermeable B horizons which form aquicludes when saturated, weakly structured topsoils	Most erodible, very vulnerable to slope wash and gully type erosion, due to dispersive nature
Alluvial soils	Generally, recipients of other soils but prone to erosion along breaks of slope
Shallow skeletal soils	Protected by surface layer of gravel but, if this is disturbed, erosion can be rapid

5.2.6 Geomorphology

The Magela floodplain, which lies 15 km downstream of the Ranger Mine, represents a catchment of 815 km^2 and joins with the floodplain of the East Alligator River.

The Magela floodplain is very flat with elevation changes of less than 0.7 m over more than 40 km. Although the inflow to the floodplain is well defined, waters continue to disperse across poorly or undefined channels until eventually discharging into the meandering channel of the East Alligator River. Average flow rates during a wet season, depending on channel definition, have been estimated at 0.02 – 0.05 m per second (Roos & Williams 1992). Wet season vegetative growth within the floodplain proper accelerates quickly with the onset of the wet season and has a significant effect upon flow rates. Roos & Williams (1992) demonstrated that the aquatic vegetation retained flood waters in the lead up to, and in the period immediately after, the highest wet season flow.

The pattern of sediments accumulated in the Magela floodplain has been examined using radionuclide analysis. Wasson (1992) found that 90 percent of the sediments transported by Magela Creek were deposited within the first 18 km of the floodplain. The rest of the floodplain sediments are sourced from smaller catchments that enter the floodplain further down the Magela Creek catchment. It was also found that Magela Creek has had no significant influence on sediment deposition below Jabiluka Billabong for the last 3,000 to 4,000 years.

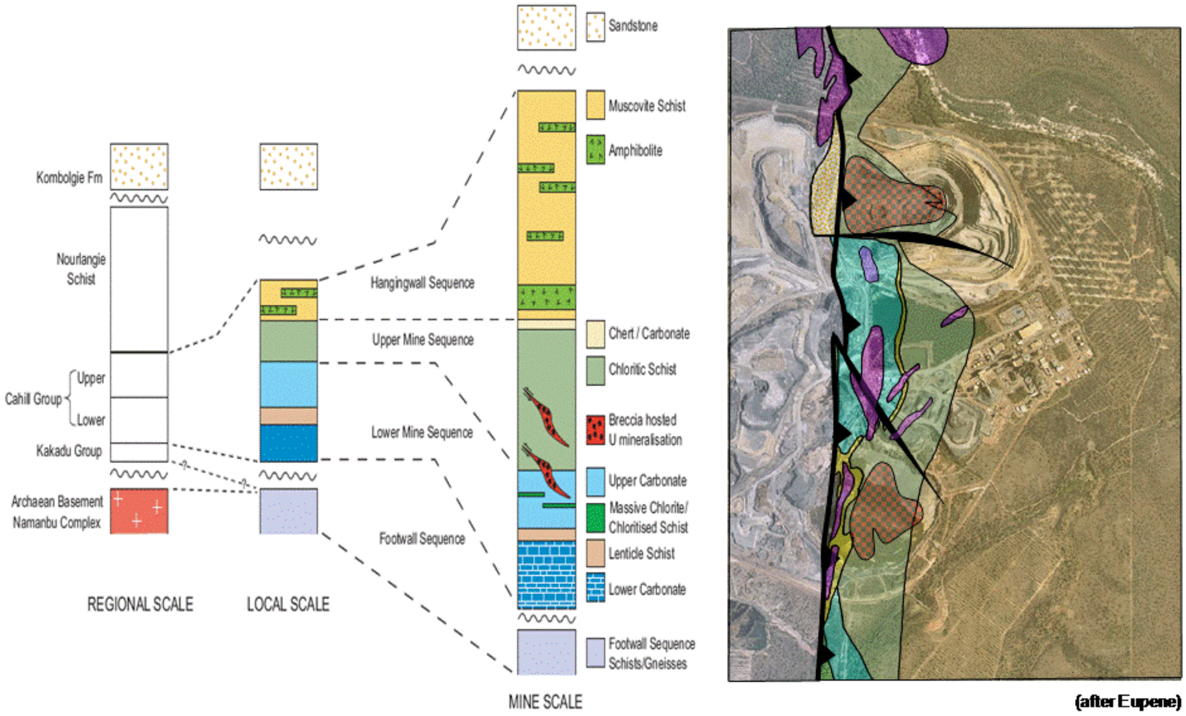


Figure 5-7: Stratigraphic sequence from regional to mine scale and corresponding geological map of the immediate area of the Ranger Mine orebodies

5.2.7 Groundwater and background constituents

The tropical, monsoon climate of the Northern Territory (NT) creates seasonal changes that drive groundwater flow into and out of the Ranger Mine area. Groundwater occurrence and flow through the RPA consists of a shallow groundwater flow system, within the relatively permeable alluvium and weathered rock, and a deeper bedrock groundwater flow system with relatively low permeability, in which groundwater is encountered within faulted, sheared, cracked and brecciated rocks1F3 Groundwater also occurs in intermediate layers of weathered bedrock between the shallow and deeper groundwater flow systems.

The alluvial and weathered rock aquifers are more connected to each other than to the deeper, fractured rock aquifer, and show similar seasonal variations in groundwater levels and quality (INTERA 2016). Groundwater within the fractured rock aquifer is weakly connected to near-surface processes, particularly rainfall-recharge, and there is limited mixing of groundwater between the shallow and deep aquifer units.

Groundwater generally flows northward across the minesite towards Magela Creek (Salama & Foley 1997, Weaver *et al.* 2010). Figure 5-8 shows the annual groundwater level behaviour

[3] Brecciated means rock that has been mechanically broken by faulting and shearing, resulting in angular fragments.

illustrating fluctuations that follow a similar, distinctive wet season – dry season oscillation akin to, but in a more subdued form than the typical surface water flow hydrograph, typically peaking following wet season recharge and declining during the dry season recession (INTERA 2019a).

In general, groundwater heads appear to increase several metres during the first one to two months of the wet season and then decrease several metres within the first two to three months of the dry season. Along Magela Creek, water exchange between the subsurface and flowing creek depends on groundwater and surface water dynamics (INTERA 2016). When surface water flow ceases in Magela Creek and Corridor Creek, subsurface groundwater flow continues through the deeper alluvial sediments of the creek beds throughout the dry season (Ahmad *et al*. 1982).

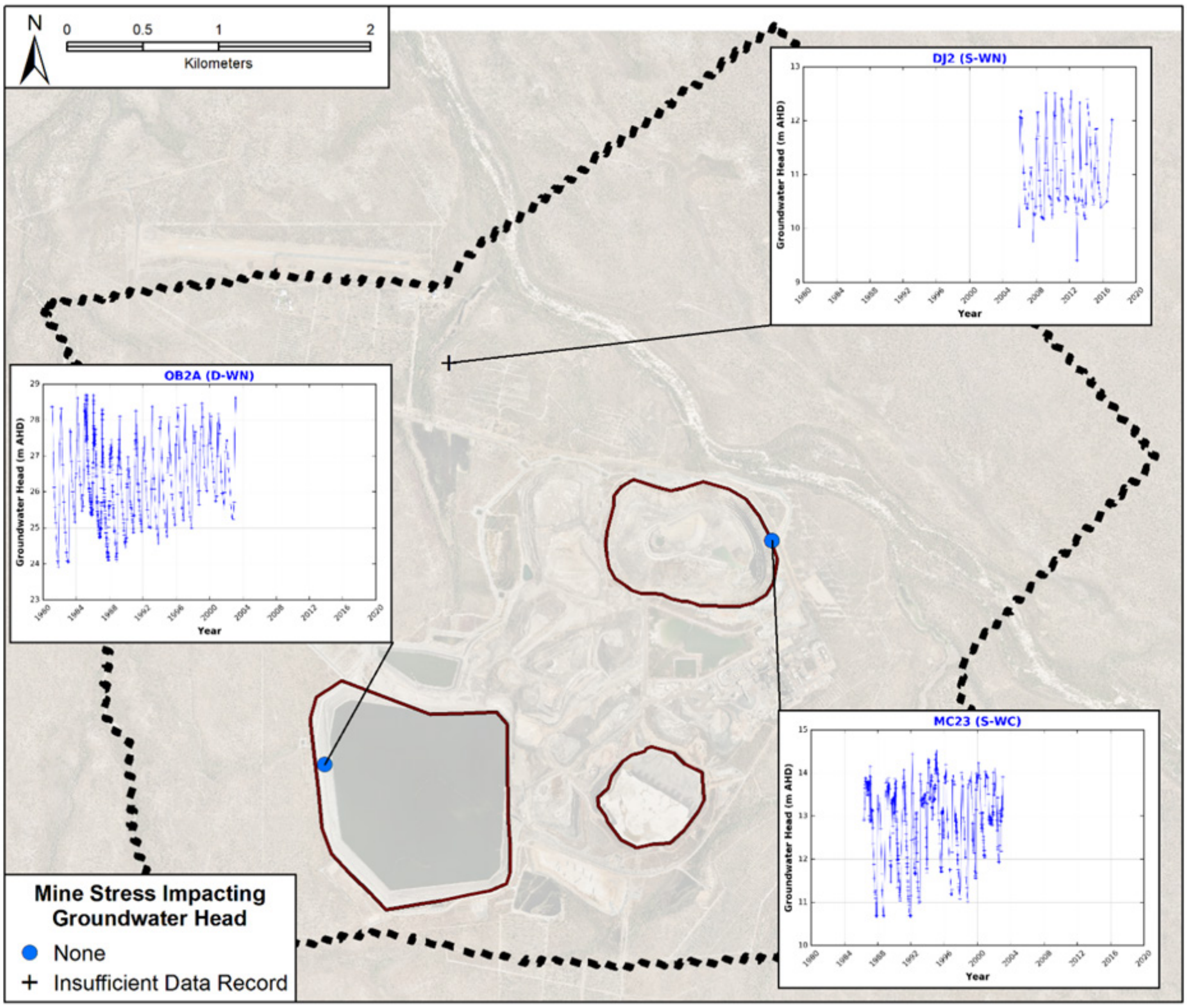


Figure 5-8 Hydrograph showing examples of seasonal groundwater head fluctuations (INTERA 2019a)

The RPA contains three distinct regional HLU zones: alluvial, weathered and bedrock. These HLU zones are discretised into specific HLUs, which describe the geological, groundwater flow and transport characteristics of that unit. A HLU can consist of a single geologic unit, part of a geologic unit, cross geologic units and mining related units in the subsurface that will be in contact with groundwater. HLUs can be aquifers or aquitards depending on their permeability. All material in which groundwater flows is assigned to an HLU, and the HLUs are the building blocks for the material components of the groundwater flow model. A breakdown of the Ranger Mine HLUs is shown in Table 5-5.

The HLUs were reviewed and updated as part of the Ranger Conceptual Model update (INTERA 2019a). The HLUs are being further reviewed and refined as part of the solute transport modelling with uncertainty analysis currently underway to support Key Knowledge Need (KKN) WS2.

The natural background hydrochemistry of groundwater of the RPA typically exhibits relatively low concentrations of total dissolved constituents. However, because of the slow passage (compared to surface water flow rates) of groundwater through the rocks, the longer contact time allows a greater degree of mineralisation of the bedrock to occur.

Baseline groundwater quality had been previously reported to ARRTC in November 2013 (ERA 2013) and November 2014 (ERA 2014c). The 2013 report described groundwater quality in six HLUs (aquifer components partitioned by hydraulic characteristics and rock type) for the five constituents of potential concern (COPCs) discussed at ARRTC in April 2012 (ERA 2012). The 2014 report described an additional COPC (radium-226), the geochemical behaviour of uranium and manganese in groundwater, the reactions of uranium and manganese with the fracture minerals that line aquifer wall-rocks and modelling work done to support the knowledge base of background concentrations of COPCs at the Ranger Mine.

In 2015, Esslemont reviewed the datasets with the geology team, which resulted in changes to the spatial assignment of groundwater to some HLUs (Esslemont 2015). Selected groundwater concentrations assigned to HLUs in November 2013 were recalculated, and the multivariate statistical analysis completed in November 2014 was revised. Following update of the Ranger conceptual mode (INTERA 2019), collection of a further 4 years of groundwater chemistry data and the increased list of COPCs to be assessed against closure criteria, the project to determine the background concentrations of COPCs in groundwater was undertaken again to inform KKN WS1.

Commencing in 2019, Environmental Resources Management (ERM) were engaged to establish a background data set for a broader suite of analytes in groundwater from HLUs identified in the Ranger Conceptual Model Update (INTERA 2019a). The evaluation was conducted with the premise that concentrations of COPCs in samples collected in potentially impacted areas comprise both mining-derived concentrations and background concentrations. This premise is used as a basis for 'extracting' an anthropogenic, site-specific background dataset from a dataset obtained from impacted areas at a site (USEPA 2014b). In the case that analyte concentrations in a sample derive only from background conditions (i.e. are not related to mining activities), the analyte is not considered to be a COPCs. Background threshold values (BTVs) were developed for the background concentration to facilitate use of the background datasets in decision making.

Table 5-5 Ranger Conceptual Model HLUs (INTERA 2019a)

HLUs	HLU Abbreviation
Alluvial HLUs	
Magela Creek sediments	MCS
other creek sediments	OCS
Djalkmarra sands	DS
Shallow Weathered HLUs	
shallow weathered Cahill	S-WC
deep weathered Cahill	D-WC
Zone C weathered carbonate (weathered Cahill subunit)	ZCWC
Pit 1 permeable zone (weathered Cahill subunit)	Pit1-P
depressurised UMS confining unit (weathered Cahill subunit)	D-UMS-C
shallow weathered Nanambu	S-WN
deep weathered Nanambu	D-WN
Deeper Bedrock HLUs	
shallow bedrock Cahill	S-BC
shallow bedrock Nanambu	S-BN
HWS	HWS
UMS	UMS
MBL zone (UMS subunit)	MBL
depressurised UMS (UMS subunit)	D-UMS
Zone C shallow bedrock (UMS subunit)	ZCSB
LMS	LMS
lower-K Deeps Water Producing Zone (DWPZ) (LMS subunit)	DWPZ-L
higher-K DWPZ (LMS subunit)	DWPZ-H
Nanambu Complex	Nam
Mine Backfill HLUs	
waste rock	NA
tailings	NA

Extraction of a background dataset from a larger site investigation dataset has support from various guidance documents (US Navy 2004; ITRC 2013; USEPA 2014b) and although no prescriptive approach is suggested, most guidance recommends a combination of a population partitioning approach followed by a weight of evidence evaluation. This is the approach that was implemented in this assessment.

Nearly a quarter of a million records from the Ranger site database were compiled and reviewed in the background assessment database to ensure that the data met the data quality and usability standards. Although some HLUs and analytes had limited spatial and/or temporal coverage, 64 HLU-analyte combinations across eight HLUs were able to undergo a full background evaluation. A robust and objective approach was taken to extract background values from the dataset. The dataset was reviewed for the number of reported results for each fraction. In all but one HLU, the dissolved fraction accounted for more than 75% of available metal data, with 9 HLUs consisting entirely of dissolved fraction metal data. Because of this, for aluminium, arsenic, beryllium, boron, cadmium, chromium, copper, iron, lead, nickel, radium, selenium, uranium, and vanadium only the dissolved fraction was retained for the background analysis. All of the available magnesium data was reported in total fraction, therefore the total fraction was used for this analyte.

In Phases 1 and 2, a data screening framework was developed to off-ramp HLUs and analytes that did not meet the minimum data requirements for the further background evaluation. Where supported, surrogate background values were developed for those HLUs and analytes with low detection frequencies, poor spatial coverage, and/or substantial data gaps. For HLUs and analytes with sufficient data, the dataset was progressed to a full background evaluation (Phase 3) that was conducted based on the following approach.

First, an iterative population partitioning approach was used to identify a breakpoint in the data using QQ plots (USEPA 2014a). This initial determination was made independently of site qualifying information. The breakpoint was then refined based on the data characteristics, in the context of the conceptual site model (CSM) and with consideration of site history, sources and known impacts. Refining the breakpoint relied on multiple lines of evidence including temporal trends in concentrations, covariance with known site sources (sulfate concentrations and SO4:Mg weight:weight ratios) and spatial patterns in impacts in the context of the CSM. Almost without exception, the final breakpoint was supported by at least one additional line of evidence; where support for the breakpoint was limited this was typically due to the dataset size and characteristics, such as concentrations approaching analytical limits. A schematic of the decision framework for data screening and the further background dataset evaluation is provided in Figure 5-9, Figure 5-10, and Figure 5-11. The background dataset was validated using multiple statistical validation methods that further strengthened the breakpoint determination by identifying additional lines of supporting evidence across COPCs and/or HLUs.

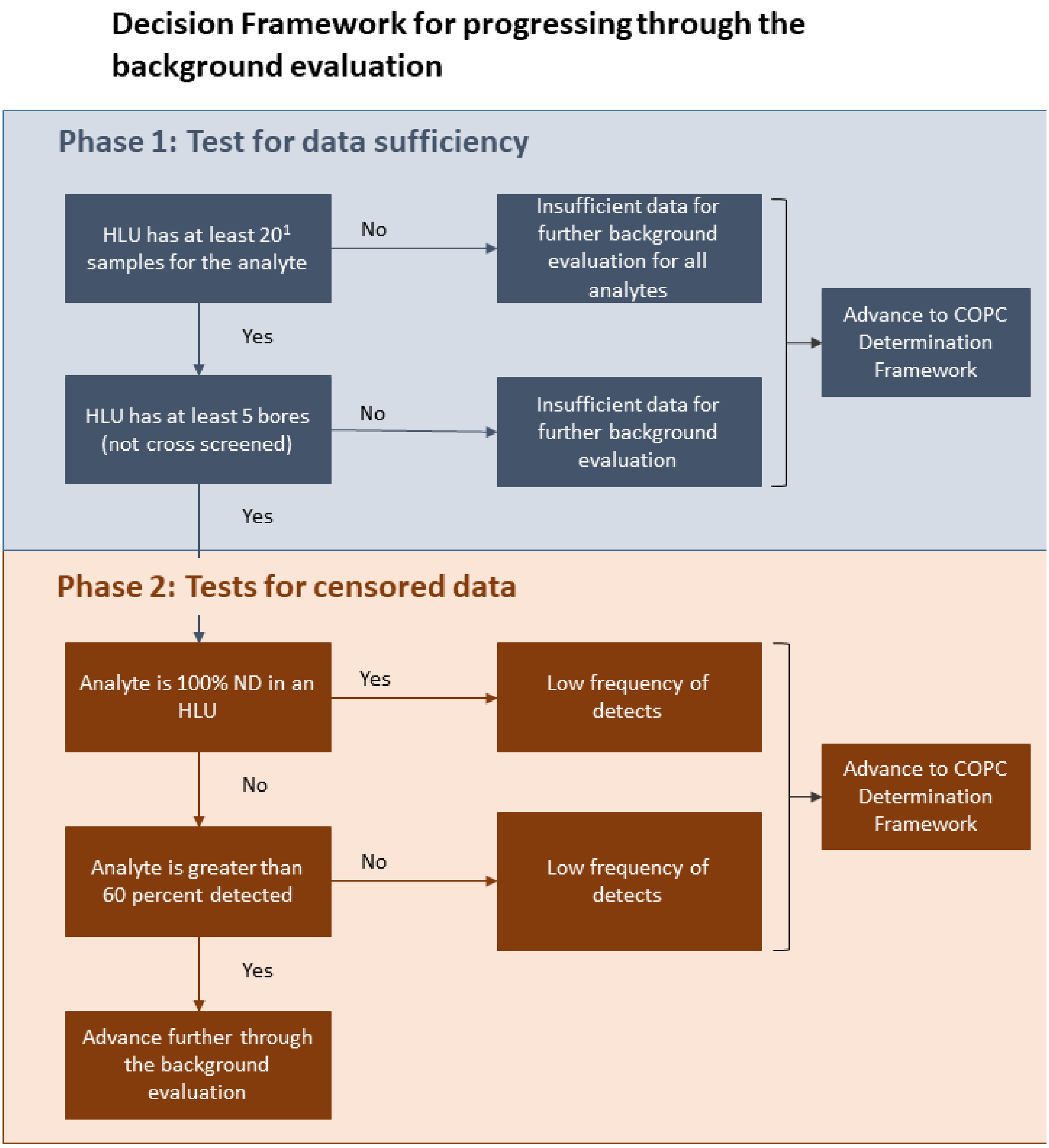
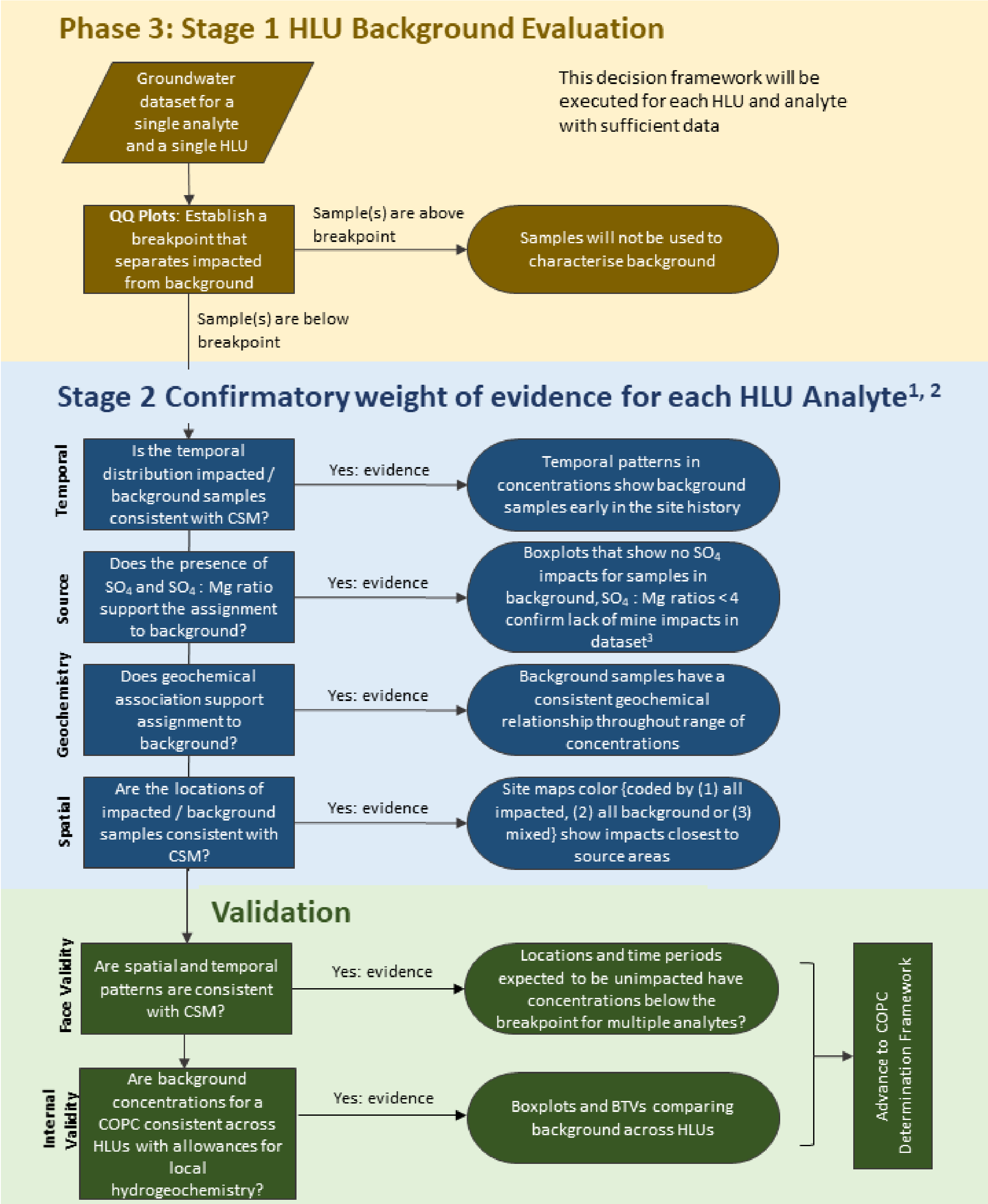


Figure 5-9 Background COPC decision framework for data screening (ERM 2020a)



Notes:

[1] Some iteration back to the QQ plot may be needed based on findings from confirmatory evaluation

[2] To aid with interpretation, data points in the visuals will be coded as impacted / background for subsequent evaluations

[3] SO_4 : Mg ratios ≥ 4 may be used as another sulfate-related line of evidence

Figure 5-10 Background COPC decision framework for weight of evidence background evaluation (ERM 2020a)

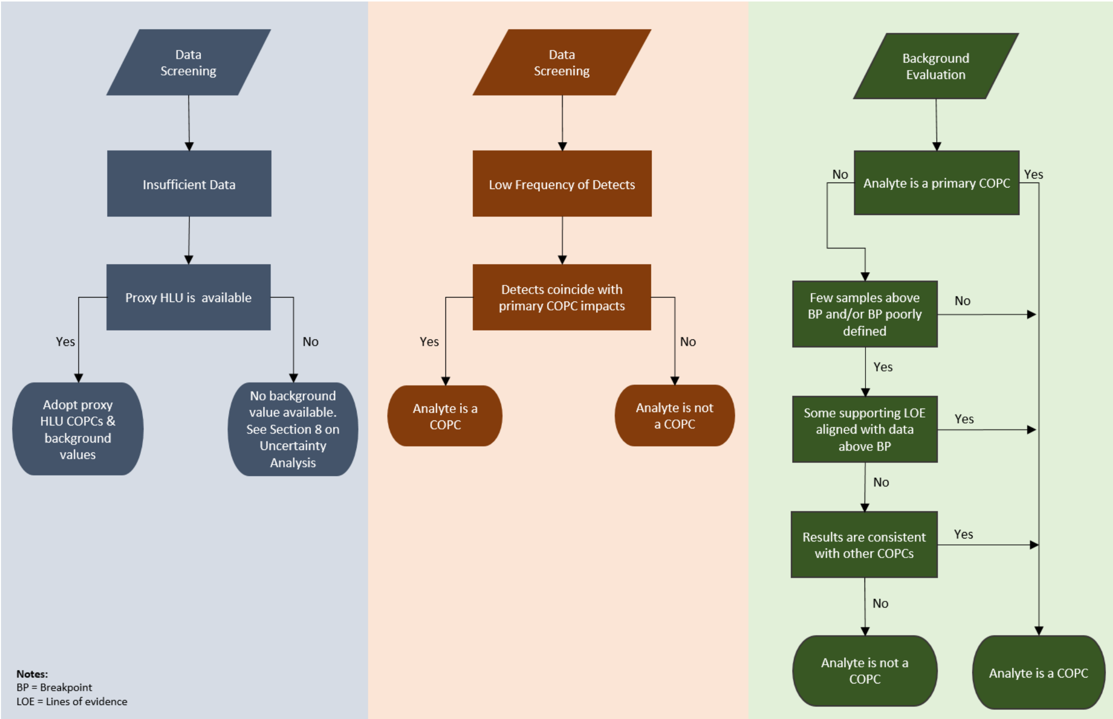


Figure 5-11 Background COPC decision framework for identifying COPC (ERM 2020a)

The initial dataset included a broader suite of analytes than had been considered previously, and the lines of evidence were used to refine the COPC list for each HLU based on evidence of impacts in the data. Primary COPCs were all retained, including uranium, radium, magnesium, manganese, and sulfate; however, the background radium dataset did not indicate that radium was a COPC in the Shallow Weathered Cahill, Shallow Bedrock Nanambu and the MBL Zone. Ammonia (NH_3-N), nitrate (NO_3-N), aluminium, arsenic, boron, nickel, and zinc were also retained as COPCs on an HLU-by-HLU basis. Several other metals did not show evidence of impacts and were ultimately removed from the COPC list. These included beryllium, cadmium, chromium, copper, lead, mercury, selenium and vanadium. The final COPC list by analyte and HLU is presented in Table 5-6.

BTVs were developed for each HLU and analyte for which there was data to support development of a BTV, even in the case that the analyte was not a COPC. The Pit 1 Permeable Zone HLU was determined to be entirely impacted at the available sampling locations and no BTVs were developed for this HLU. Calculated BTVs are presented in Table 5-6; background concentrations, which were adopted as BTVs for data with a low frequency of detects, are presented in Table 5-7. In this project 95/95 upper tolerance limits (UTLs) were used as BTVs for the background datasets. BTVs are advantageous because they are simple to implement and understand, they do not need to be recalculated over time, and point comparisons (single data points) can be made to the BTV. However, the application of BTVs can be problematic, because the more comparisons are made to the BTV, the more likely false positives become (i.e. the chance of falsely concluding that a sample or bore is impacted). Therefore statistical hypothesis testing is recommended to control for false positive rates in those cases where COPC concentrations are above the BTV.

Table 5-6 Background Threshold Value (BTV) from data rich HLUs from the background evaluation 95/95 Upper Tolerance Limit (ERM 2020c)

Analyte	Units	Shallow Bedrock Cahill	Deep Weathered Cahill	Shallow Weathered Cahill	Shallow Bedrock Nanambu	Deep Weathered Nanambu	Shallow Weathered Nanambu	MBL Zone (UMS subunit)
Aluminium	µg/L			27.6	14.4[a]	34.9	19.3	
Ammonia (NH_3-N)	mg/L				0.88	0.312	0.43	
Arsenic	µg/L				2.5	8	4.5	
Boron	µg/L				30	55	25	
Copper	µg/L			3.8		4	6.15	
Lead	µg/L			0.9			2.05	
Magnesium	mg/L	21.7	57.9	11.1	39.8	26.7	52.3	40.5
Manganese[b]	µg/L	190	87.5	483	1420	401	890	18
Nickel	µg/L				2.3	4.9	11.5	
Nitrates (NO_3-N)	mg/L		0.554	3.17				0.554
Radium	mBq/L	130	50	27.3	130	90	30	37.3
Sulfate	mg/L	1.5	4.3	1.88	2.5	7.6	1.6	1.6
Uranium	µg/L	7.74	21.9	3.03	5.76	5.7	3.37	1.92
Vanadium	µg/L					3		
Zinc	µg/L			13	3	16.5	11.5	

Table 5-7 Background COPC concentrations HLUs for analytes with low frequency of detects (ERM 2020c)

HLU	Analytes	Adopted Background Concentration	Basis for Selection
Deep Weathered Cahill	Ammonia	0.005 mg/L	Detection limit reported in all samples available.
Deep Weathered Nanambu	Beryllium	0.5 µg/L	Detection limit reported in all samples available.
	Cadmium	0.1 µg/L	Detection limit reported in all samples available.
	Chromium	0.5 µg/L	The lowest and most frequently detection limit reported from samples available.
	Lead	0.1 µg/L	Based on detectable lead concentrations in groundwater at bores located away from mine activities and considered to be background (22138_D and 23931_DEEP).

HLU	Analytes	Adopted Background Concentration	Basis for Selection
	Mercury	0.1 µg/L	100% of concentrations were reported below detection limit.
	Nitrates	0.022 mg/L	Detection limits ranged from 0.005 mg/L to 0.1mg/L. The selected background concentration was the most frequent detection limit reported and was also from the most recent analyses (after 2010).
	Selenium	1.0 µg/L	The lowest and most frequent detection limit reported from samples available.
MBL Zone (UMS subunit)	Ammonia	0.005 mg/L	The lowest and most frequent detection limit reported from samples available.
Pit 1 Permeable Zone	Ammonia	0.005 mg/L	Detection limit reported in all samples available. Other background concentrations not able to be assessed for this HLU.
Shallow Bedrock Cahill	Nitrates	0.022 mg/L	Detection limits ranged from 0.01 mg/L to 0.1 mg/L. The selected background concentration was the most frequent detection limit reported and was also from the most recent analyses (after 2010).
Shallow Bedrock Nanambu	Beryllium	0.5 µg/L	100% of concentrations were reported below detection limit.
	Cadmium	0.1 µg/L	100% of concentrations were reported below detection limit.
	Chromium	0.5 µg/L	100% of concentrations were reported below detection limit.
	Copper	0.05 µg/L	The lowest and most frequent detection limit reported from samples available.
	Lead	0.05 µg/L	The lowest and most frequent detection limit reported from samples available.
	Mercury	0.1 µg/L	100% of concentrations were reported below detection limit.
	Nitrate	0.022 mg/L	Detection limits ranged from 0.01 mg/L to 0.1 mg/L. The selected background concentration was the most frequent detection limit reported and was also from the most recent analyses (after 2010).
	Selenium	1 µg/L	100% of concentrations were reported below detection limit.
	Vanadium	0.5 µg/L	100% of concentrations were reported below detection limit.

HLU	Analytes	Adopted Background Concentration	Basis for Selection
Shallow Weathered Cahill	Ammonia	0.005 mg/L	100% of concentrations were reported below detection limit.
Shallow Weathered Nanambu	Beryllium	0.5 µg/L	The lowest and most frequent detection limit reported from samples available. .
	Cadmium	0.1 µg/L	Most frequent detection limit reported from samples available.
	Chromium	0.5 µg/L	Most frequent detection limit reported from samples available.
	Mercury	0.1 µg/L	Most frequent detection limit reported from samples available.
	Nitrates	0.022 mg/L	Detection limits ranged from 0.005 mg/L to 0.1 mg/L. The selected background concentration was the most frequent detection limit reported and was also from the most recent analyses (after 2010).
	Selenium	1 µg/L	Most frequent detection limit reported from samples available.
	Vanadium	0.5 µg/L	Most frequent detection limit reported from samples available.

This background evaluation has refined the COPC list for the site, established background datasets for HLUs and analytes, and calculated BTVs for analytes and COPCs on an HLU-by-HLU basis. The BTVs were established using an objective decision framework that supported a defined process that was generalisable and repeatable across analytes and HLUs. This resulted in a transparent and defensible process. The results were supported by multiple forms of validation that help to create a high level of confidence in the conclusions.

The approach allowed the data to dictate the background concentrations and then supported this with multiple lines of evidence and site knowledge to develop BTVs and to identify COPCs for the HLUs at the site. The statistical methodology used to establish the background dataset and develop the supporting lines of evidence is well established and reproducible, and the uncertainty evaluation did not identify material inconsistencies in the data or the approach that would need to be considered when using the resulting BTVs to inform site closure decisions.

5.2.8 Surface water

5.2.8.1 Hydrology

Surface water management will be a key focus of rehabilitation and closure, as it is one of the pathways for COPCs to enter the environment.

The Ranger Mine is located within the 1,600 km^2 of the Magela catchment and adjacent to Magela Creek (Figure 5-12). Two tributaries of Magela Creek are also located in close proximity to the mine: Gulungul Creek to the west and Corridor Creek to the south. Magela Creek is a seasonally flowing tributary of the East Alligator River, with a catchment originating from headwaters on the Arnhem Land Plateau.

The seasonal pulse of the wet season monsoon controls regional hydrology (Wasson 1992) with flows beginning in an average year in mid-December, after the onset of the monsoonal wet season which usually occurs in November. During the wet season, creeks become sheets of water that extend beyond the low banks. This water is reduced to a series of isolated backflow billabongs and swampy depressions in the dry season winter months. Poor drainage makes access to surrounding areas difficult and roads and tracks are frequently cut off by flood waters for extended periods in the wet season. The sand aquifers in the channel of Magela Creek, in the middle catchment fill, with shallow groundwater and begin flowing as interflow within the creek channel, before surface flow commences in the creek. Average annual runoff for the Magela Creek system has been estimated at 420 GL (Moliere 2005, Salama & Foley 1997, Vardavas 1988).

Magela Creek and its tributaries flow north from the extensive sandstone Arnhem Plateau. In more specific terms, Magela Creek comprises four sections:

- escarpment channels that flow through deep narrow gorges, which make up around one third of the Magela catchment. These systems are fed by groundwater seeping into the fractured rock of the escarpment and can flow practically all year round. Escarpment rainforest vegetation species (dominated by Allosyncarpia ternate (a Kakadu hardwood tree species)) are found in the gullies due to year-round water supply.

- sand bed anabranching channels (Jansen & Nanson 2004) with sandy levees. Magela Creek flows through sandy soils that may be more than five metres deep along the creek channels. This is the section in which the Ranger Mine is located.

- a series of billabongs and connecting channels at Mudginberri (termed the Mudginberri Corridor)

- a 200 km2, seasonally inundated black-clay floodplain, at two to five metres above sea level, with permanent billabongs, and a single channel that discharges into the East Alligator River approximately 40 km to the north of the RPA and, ultimately, Van Diemen Gulf

Gulungul Creek, on the western boundary of the RPA, drains runoff from the catchment to the west and south of the Ranger Tailings Storage Facility (TSF) and from relatively undisturbed bushland to the west of Retention Pond 1 (RP1). The main stream of the Gulungul Creek has a length of around 12.5 km. The Gulungul sub-catchment has an area of approximately 98.4 km^2.

Moliere (2005) reviewed historical stream flow data for Gulungul Creek in order to provide confidence in the flow and flood frequency estimations. Despite data gaps, an annual runoff of 25.5 GL at G8210012, immediately west of Ranger Mine (as shown on Figure 5-13)[4] was determined, with a general flow period for Gulungul Creek of approximately six months between December and May. Observations from Ranger Mine operations have noted that the general flow period can, however, extend through to June or July in above average wet seasons. Stream flows are highly variable throughout the wet season and reach peak discharge during the months of February to March (Salama & Foley 1997).

Antecedent rainfall in the Gulungul sub-catchment that is required prior to overland flow in Gulungul Creek is similar to that for Magela Creek at approximately 295 mm (Moliere 2005).

Corridor Creek drains the southern side of the Ranger Mine. The natural catchment has been modified in the vicinity of the mine, with mine drainage water being redirected to water treatment areas. There is also a series of natural and artificial water bodies within the creek line that modulate the effects of storms and rainfall events. Corridor Creek runs into Georgetown Creek at Georgetown Billabong. The main water bodies in Corridor Creek include the pre-mining Georgetown Billabong and the constructed Corridor Creek wetland filter (CCWLF), the Georgetown Creek Brockman Road (GCBR) bund, Georgetown Creek Mine Bund Leveline (GCMBL) and Sleepy Cod Dam.

Prior to mining, the local hydrology included four separate sub-catchments, namely Gulungul to the west and southwest, Coonjimba in the centre west, Djalkmarra in the centre east and Corridor Creek in the east and south. Within the sub-catchments, backflow billabongs sit on the margins of Magela Creek creating complex localised hydrological relationships.

[4] Government agency gauging stations shown in Figure 5-13 correspond with stations listed on the NT Government, Natural Resource Maps website: https://nt.gov.au/environment/environment-data-maps/water-data

Figure 5-12: Regional extent of Magela catchment

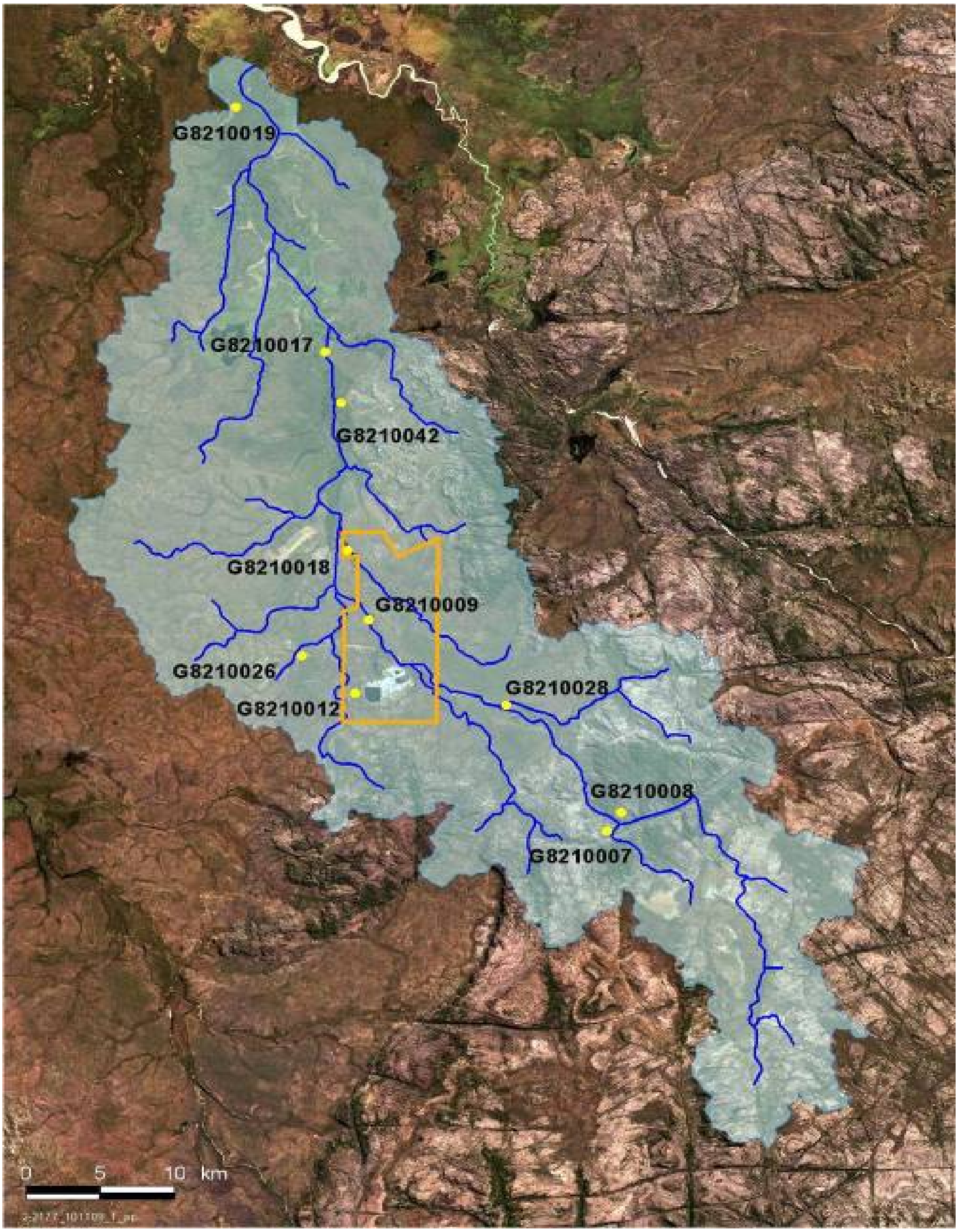


Figure 5-13: Magela catchment showing government agency gauging stations

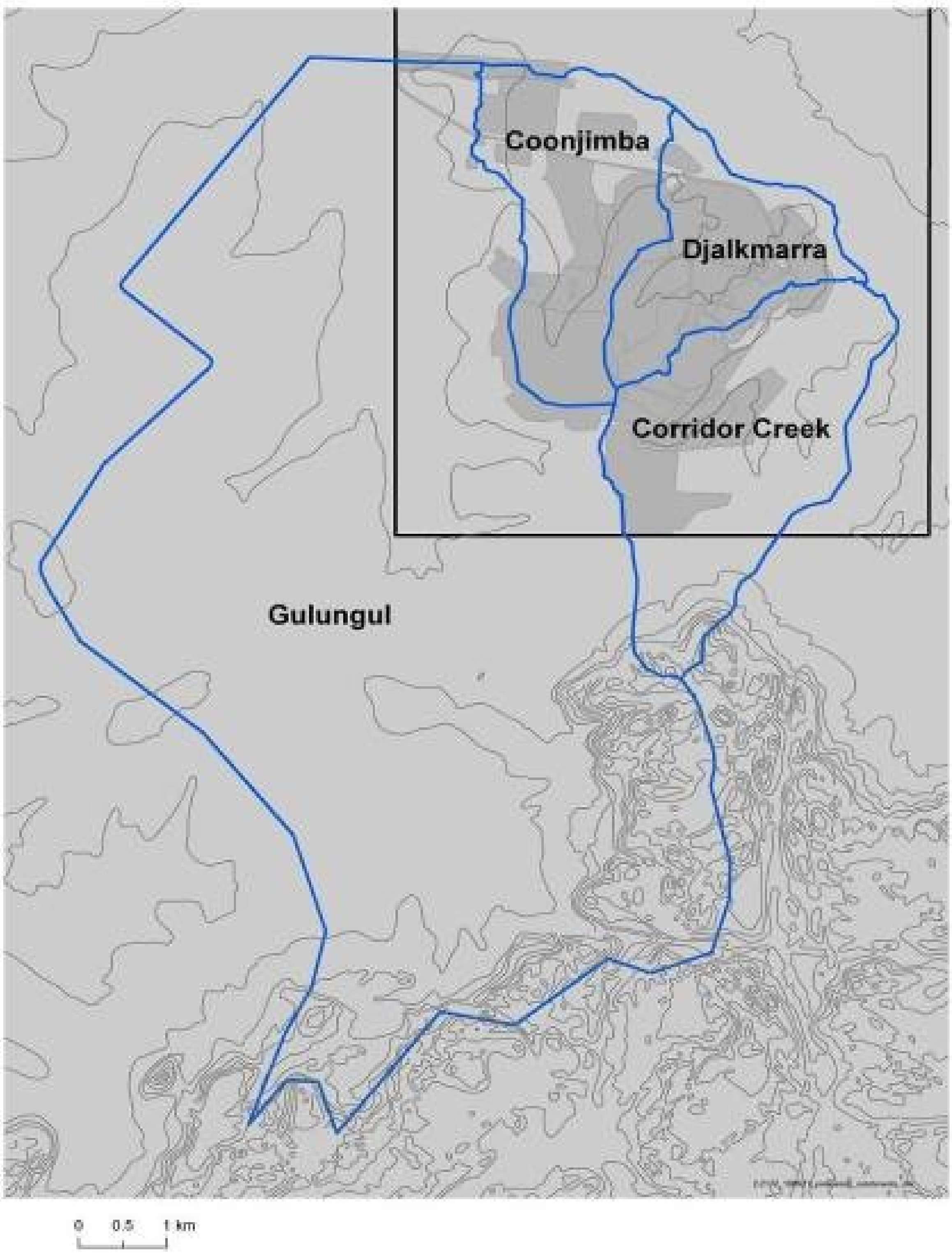


Figure 5-14 Pre-mining catchments in relation to the Ranger Mine

5.2.8.2 Water quality

Water quality monitoring has been ongoing at Ranger Mine and in the surrounding environment for several decades providing a significant volume of reference data for surface water quality within the creeks and billabongs. Several studies conducted before, or shortly after, mining commenced describe the background conditions in billabongs and creeks in the Magela Creek catchment (e.g. Hart and McGregor 1980, 1982, Walker & Tyler 1982, 1983, Office of the Supervising Scientist 2002, Hart *et al.* 1987a, Hart *et al.* 1987b, Hart *et al.* 1981, Hart *et al.* 1986b, Hart *et al.* 1986a).

Klessa (2000) derived baseline water quality data for Magela Creek against which change in water quality could be determined, based on:

- Ranger Mine water quality data base
- Northern Territory Department of Primary Industry and Resources (DPIR) check monitoring water quality database, and
- Northern Territory Water Resources Division (WRD).

The majority of water samples were taken upstream of Ranger Mine from site GS8210067. In addition, the DPIR data is independent of the Ranger Mine data. The WRD data from the downstream site GS009 collected before the 1976-77 wet season is pre-mining data. The Klessa (2000) baseline data (provided within Klessa (2005) analysed the Magela Creek monitoring data to produce a balance sheet over 4 wet seasons (1999 to 2003) to account for magnesium sulfate entering Magela Creek from the Ranger Minesite.

Upstream Magela Creek data (from 1993 to 2003) showed magnesium concentrations varied from approximately 1 mg/L at low flow to less than 0.1 mg/L flow rates that exceeded 100 m^3/s. Corresponding sulfate concentrations ranged from approximately 0.1 to 1 mg/L but did not show the same negative correlation with flow rate. EC showed that same trend as magnesium with EC decreasing with flow rates approximately 20 microSemens/cm to 5 microSemens/cm. At the end of the wet season, upstream of Ranger Mine, waters have elevated magnesium and EC. This implies a base-flow water source with higher ionic strength than the predominantly allogenic surface water flow observed earlier in the wet season.

Generally EC and magnesium variation follows the hydrological phases of flow, which is a decrease in concentration from start of wet season to a minimum near mid-wet season, followed by a subsequent increase to end of wet season. The EC and magnesium concentrations in surface water at the start and end of the wet season are similar. This observation by Klessa (2005) is consistent with the results of the ERA and SSB monitoring programs.

Table 5-8 was derived from Ranger and the DPIR datasets from sites GS028 and GS067 and WRD site GS009.The results in Klessa (2000) are compared to the 1992 – 2018 Magela Creek upstream reference site (MCUS) data, collected by the ERA (predominantly weekly) monitoring program (Table 5-8). The Klessa (2000) dataset contains MCUS data from 1991, which is considered to be affected by Georgetown Billabong (GTB) outflows (Hart *et al.* 1982). Some data from this location have high uranium in the early part of the year. However, the dataset contains greater than 200 data points and the statistics shown are percentiles rather than an average, so the influence of these points is considered to be small. Data from that time is not included in the MCUS 1992 – 2018 dataset.

A review conducted a decade after Klessa (2005) describes similar water quality and seasonal treands. INTERA (2016) describe Magela Creek surface water chemistry as being generally slightly acidic pH (~6.2) with very low electrical conductivity (EC) (~15 to 16 micro Siemens per centimetre; up to 30 micro Siemens per centimetre during low flow conditions), and low turbidity (7 Nephelometric Turbidity Unit) and metal concentrations, reflecting rainfall chemistry more closely than groundwater chemistry.

During the wet season, EC and concentrations of magnesium (Mg) and calcium (Ca) upstream of the Ranger Mine are highest during initial flows, lowest during high flows and increase during the recessional flow limb (late wet season, when stream flow is decreasing). Sulfate (SO_4) and manganese (Mn) concentrations are highest with the start of flow, but then decrease to steady levels; whereas turbidity is high during the accessional limb (early wet season, when stream flow is increasing), but decreases to a steady low during the recessional limb. Only pH appears to increase during the period of flow, although it is highly variable over the entire period. Uranium (U), total ammonia nitrogen (TAN) and radium-226 (^{226}Ra) remain essentially constant throughout the period of flow (INTERA 2016).

A comparison of the Magela Creek water chemistry upstream and downstream of the Ranger Mine indicates that generally:

- turbidity is lower downstream than upstream
- pH and Mg and SO4 concentrations are higher downstream than upstream
- Mn, U, Ca, 226Ra and TAN concentrations are similar downstream and upstream, with the following exceptions:
    - Mn concentrations are higher downstream than upstream during the recessional limb, and
    - U concentrations are very occasionally slightly higher downstream than upstream (INTERA 2016).

ERA

Table 5-8: Baseline values from Klessa (2000) and ERA Laboratory Information Management System (LIMS) database 1992-2018; results are for filtered fraction except for ^{226}Ra

Parameter	Unit	Source	*n*	Minimum	Percentiles		Maximum
					50th	80th	
pH	-	Klessa 2000	*366*	4.20	6.20	6.45	7.00
		MCUS 1992-2018	*880*	3.97	6.15	6.44	8.04
EC	(µS/cm)	Klessa 2000	*493*	5	16	21	75
		MCUS 1992-2018	*885*	3.4	13	16	47
Turbidity	(NTU)	Klessa 2000	*356*	0.5	5	9.9	82
		MCUS 1992-2018	*718*	<1	3	5	46
SO_4	(mg/L)	Klessa 2000	*232*	0.03	0.27	0.78	9.3
		MCUS 1992-2018	*805*	0.03	0.20	0.40	3.5
Mg	(mg/L)	Klessa 2000	*266*	0.05	0.64	0.88	8.1
		MCUS 1992-2018	*806*	0.05	0.55	0.80	1.7
Ca	(mg/L)	Klessa 2000	*214*	0.05	0.52	0.8	6
		MCUS 1992-2018	*682*	0.05	0.30	0.50	1.3
Na	(mg/L)	Klessa 2000	*150*	0.05	1.3	1.7	5.5
		MCUS 1992-2018	*379*	0.05	1.2	1.4	2.5
K	(mg/L)	Klessa 2000	*149*	0.05	0.22	0.4	1.8
		MCUS 1992-2018	*379*	0.05	0.12	0.20	1.00
Cl	(mg/L)	Klessa 2000	*125*	0.8	2.1	3	24
		MCUS 1992-2018	*324*	0.3	1.8	2.2	3.4
NO_3	(mg/L)	Klessa 2000	*122*	0.002	0.03	0.05	0.43
		MCUS 1992-2018	*163*	0.011	0.011	0.050	0.841
NH_3	(mg/L)	Klessa 2000	*76*	0.01	0.01	0.025	0.18
		MCUS 2013-2019	*179*	0.003	0.012	0.021	0.068
Cu	(µg/L)	Klessa 2000	*105*	0.1	0.6	1	3.49
		MCUS 1992-2018	*78*	0.0	1.00	1.00	3.49
Mn	(µg/L)	Klessa 2000	*224*	0.5	5.6		180
		MCUS 1992-2018	*807*	0.22	4.93	7.35	41.5
Pb	(µg/L)	Klessa 2000	*122*	0.01	0.5		22
		MCUS 1992-2018	54	0.020	0.025	0.124	0.530
U	(µg/L)	Klessa 2000	*260*	0.013	0.10	0.30	24.95
		MCUS 1992-2018	*853*	0.003	0.030	0.050	3.50
Zn	(µg/L)	Klessa 2000	*93*	0.5	2.5	13.0	81
		MCUS 1992-2018	88	0.25	1.00	1.72	141
^{226}Ra Total	(mBq/L)	Klessa 2000	*101*	0.6	3	18.0	43.2
		MCUS 1992-2017	137	0.5	1.94	3.94	58.4
Al	(µg/L)	Klessa 2000	NR	NR	NR	NR	NR
		MCUS 1992-2018	*43*	0.5	51.5	99.8	187
Fe	(µg/L)	Klessa 2000	NR	NR	NR	NR	NR
		MCUS 1992-2018	48	28	97	130	544

5.2.9 Radiation

To determine the achievement of criteria for both human health and environmental protection, the pre-mining radiation baseline is required. All assessments against radiation criteria will be made based on the above-background mine-sourced radiation dose. This section details the pre-mining baseline.

5.2.9.1 Terrestrial baseline radiation

The pre-mining radiological conditions for the Ranger Mine have been investigated and reported by the Supervising Scientist (Bollhöfer *et al*. 2014). The study was based on pre-mining aerial surveys, with extensive ground measurements to provide calibration of the final external gamma radiation dose rates. Ground measurements taken for soil radon concentrations and radon exhalation rates were then correlated to the airborne gamma results to obtain averages for the area. The summary of results from this study is provided in Table 5-9.

The results show that the average external gamma dose rate in areas removed from uranium mineralisation ranges between 0.10 and 0.20 microgray per hour, with the overall average for the RPA being 0.11 microgray per hour. Dose rates above the orebodies were, as expected, much higher, reaching an average of 0.87 microgray per hour above Pit 1.

Similar patterns to the gamma dose rates were observed for both average soil radium concentrations and average radon exhalation. Average radium concentrations over the orebodies (880 – 1,800 Becquerels (Bq)/kg) were much higher than for the surrounding area (110 Bq/kg), as were the average radon flux densities over the orebodies (1.3 -2.7 Bq/kg per square metre per second) relative to the surrounding area (0.15 Bq per square metre per second).

5.2.9.2 Aquatic baseline radiation

The RPA contains three distinct regional HLU zones: alluvial, weathered and bedrock.The derivation of the background threshold values for uranium and radium is discussed in 5.2.7. The results for uranium and radium baseline groundwater concentrations are presented in Table 5-10. Radionuclide concentrations in Magela Creek, upstream of the Ranger Mine, are routinely monitored throughout the wet season by both ERA and the SSB. Water quality at this location is considered to be unaffected by mining and therefore representative of baseline conditions. The statistical results of Magela Creek upstream monitoring conducted by ERA for the 2010 to 2014 wet seasons are presented in Table 5-11.

Table 5-9: Pre-mining radiological baseline determined by the Supervising Scientist (Bollhöfer *et al.*, 2014)

Location	Average gamma dose rate ($\mu Gy\ h^{-1}$) *	Average radium concentration ($Bq\ kg^{-1}$)*	Average radon exhalation ($Bq\ m^{-2} s^{-1}$)*
Pit 1	0.87 ± 0.18	1,880 ± 430	2.7 ± 0.8
Pit 3	0.44 ± 0.09	880 ± 200	1.3 ± 0.4
Djalkmarra land application area	0.20 ± 0.03	310 ± 70	0.46 ± 0.14
Corridor Creek land application area	0.14 ± 0.02	170 ± 40	0.25 ± 0.08
TSF	0.11 ± 0.01	110 ± 30	0.16 ± 0.05
Magela land application area	0.12 ± 0.01	110 ± 30	0.17 ± 0.05
RP1	0.11 ± 0.01	90 ± 20	0.14 ± 0.04
RP1 land application area	0.11 ± 0.01	90 ± 20	0.13 ± 0.04
Jabiru East land application area	0.10 ± 0.01	90 ± 20	0.13 ± 0.04
Jabiru	0.11 ± 0.01	90 ± 20	0.14 ± 0.04
Ranger Project Area	0.11 ± 0.01	110 ± 20	0.15 ± 0.05
* ± 95% confidence			

Table 5-10 Estimated baseline groundwater radionuclide concentrations

Analyte	Units	Shallow Bedrock Cahill	Deep Weathered Cahill	Shallow Weathered Cahill	Shallow Bedrock Nanambu	Deep Weathered Nanambu	Shallow Weathered Nanambu	MBL Zone (UMS subunit)
Radium	mBq/L	130	50	27.3	130	90	30	37.3
Uranium	µg/L	7.74	21.9	3.03	5.76	5.7	3.37	1.92

Table 5-11: Magela Creek upstream radionuclide concentrations (2010 – 2014 average)

Magela Creek upstream	Total radium-226 ($mBq\ L^{-1}$)	Total uranium ($mBq\ L^{-1}$)
Average	2.1	0.70
Minimum	1.2	0.16
Maximum	4.0	2.6
Standard deviation	0.9	0.48

5.2.9.3 Bushfood baseline radiation

Radiation work to date has focused on radiation exposure of people living a traditional lifestyle in the area, and downstream of the RPA, along with radiation exposure of plants and animals inside and downstream of the RPA. This work has included extensive monitoring to determine pre-mining, area-wide radiological conditions, as a first step to assessing post-mining changes and the success of rehabilitation from a radiological perspective (e.g. Bollhöfer *et al*. 2014, Bollhöfer *et al*. 2011, Esparon *et al*. 2009).

Aboriginal people living a traditional lifestyle in Kakadu NP consume bush foods that contain natural background concentrations of radionuclides. A summary of the available data on the uptake of radionuclides into aquatic and terrestrial foodstuffs was completed by ERISS and published in its annual research summary (Ryan *et al*. 2009).

A model diet for local Aboriginal people was obtained from the following sources:

- a questionnaire developed by ERISS and distributed to local Aboriginal people in 2006
- information provided by a local supplier of meats to Aboriginal outstations, and
- data gained from ERISS Kakadu bush food project over the last 11 years.

ERISS collated all available data on radionuclide activity concentrations in bush foods (from natural sources) and used this to determine a baseline radiation dose to Aboriginal people living in the region from ingestion of foodstuffs of 0.84 mSv/year. This radiation dose is irrespective of the mining activity and reflects the natural state for Aboriginal people living in Kakadu NP.

ERISS has compiled this data, along with more recently collected information, into a database (Doering 2013). The database can be used to determine bush food concentration ratios, from which the ingestion dose from various parameter inputs and a variety of situations can be calculated (Ryan *et al*. 2011). The database contains more than 1,500 individual records of radionuclide activity concentrations in various plants, animal tissues and environmental media. All information in the database has associated geospatial information to allow for spatial analysis. ERISS has also developed a bush foods geospatial information system called the "bushtucker database" (Walden 2011). This contains 30 years of data on radionuclide concentrations in traditional bush foods and is available to the public.

A summary of radionuclide concentrations published by ERISS for key flora and fauna of the Alligator Rivers Region is provided in Table 5-12 (Bollhöfer *et al*. 2011, Martin & Ryan 2004, Ryan *et al*. 2009, Ryan *et al*. 2005). Since completion of the baseline data assessment ERISS have since published updated radionuclide activity concentrations (Doering and Bollhöfer, 2016b, Doering et al., 2017). This data will be used in any further radiation dose assessments.

Table 5-12: Radionuclide concentrations in local bush foods

Bush food	Radionuclide activity concentrations (mBq g^{-1} fresh weight)[1]		
	Uranium	Radium	Lead
Wallaby flesh[2]	0.025	1.9	0.7
Magpie goose[3]	0.004	0.03	0.05
Mussels[1, 4]	2.7 – 7.6	450 – 2,500	360 – 800
Turtle flesh[2]	0.007	0.16	0.098
Fish[2]	0.005 – 0.085	0.22 – 3.5	0.043 – 0.20
File snake[2]	0.021	0.031	0.037
Cheeky yams[3]	0.06	0.26	0.042
Various fruits[5]	0.020 - 0.028	0.26 – 71	0.042 – 11
Water lily[2]	0.96	5.1	4.3

Notes:

[1] Mussels from Mudginberri Billabong, data provided are dry weights; [2] Source (Ryan *et al*. 2009); [3] Source (Martin & Ryan 2004); [4] Source (Bollhöfer *et al*. 2011); [5] Source (Ryan *et al*. 2005)

5.2.10 Sediment

Aquatic sediments at Ranger Mine and the Magela catchment have been studied since the late 1970s. This includes research projects as well as a routine monitoring to understand metal concentrations and bio-geochemical pathways, spatial distribution (vertically and within and between catchments), changes over time, and potential bioavailability.

1970 - 2001

A number of studies of sediment quality from billabongs along the Magela Floodplain were carried out in the late 1970's and early 1980's. The earlier work was done by Pancontinental in 1978 and 1979 as baseline studies, but did not include uranium data (Pancontinental, 1981).

Johnston and Milnes (2007) lists a number of reports from the 1980s that assessed the fate of chemical species with respect to deposition as sediment and quantities stored in floodplain sediments and described the physico-chemical properties of sediments in billabongs. They describe the geochemical behaviour of sediments and their interactions with water and the use of sediment monitoring as a method for early detection of potential ecological effects.

Jones *et al*. (2001) collected sediment samples from the Magela Creek Floodplain billabongs in November and December, 1997, at the end of the dry season as part of the Jabiluka baseline data collection.

Monitoring of sediments in selected billabongs on and adjoining the Ranger Project Area (RPA) formed part of the regulatory framework governing the authority to operate between 1981 and 2002. In 2002, the Supervising Authorities accepted a recommendation (Milnes *et al*. 2002) to cease the prescriptive statutory routine monitoring which they said was not a good basis for assessment of environmental protection. Instead performance-based monitoring using a project based approach was to be undertaken.

Iles and Klessa (2010) provides a characterisation of sediments in billabongs on and off the Ranger site, based on a review of literature and a comprehensive summary of all the sediment data from Ranger wetlands and billabongs, collected by ERA from 1981 to 2002. Uranium was confirmed as the contaminant of concern. The uranium concentrations in Coonjimba, Gulungul and Mudginberri Billabongs were similar throughout this period, with an increase in concentration in Coonjimba Billabong from 1999.

2003 - 2015

Performance-based monitoring of the sediments in Retention Pond 1(RP1), Georgetown Billabong (GTB) and the RP1 and Corridor Creek constructed wetland filters (CCWLF) was undertaken by ERA in 2003 – 2006 to assess the current status of those sediments, in terms of spatial and temporal distribution of contaminants.

The results are reported in Iles *et al.* 2010 who describe the metal concentrations and relationships in surface and core sediments for different digestion methods and compares the measured concentrations in both to earlier data and to sediment quality guidelines. Based on total and bioavailable U concentrations in the surface sediments the ecological risk associated with the sediments at the onsite water bodies was ranked (from highest to lowest) as RP1 wetland filter > Corridor Creek wetland filter (CCWLF) > RP1 > GTB ≈ Coonjimba.

The Supervising Scientist conducted a sediment sampling and analysis program from billabongs in the Alligator Rivers Region in 2007, 2011 and 2013. The three data sets had comparable sampling and analysis methods and were designed to assess the different sampling, sediment fractions, and extraction methods. Results are reported in Parry 2016.

In 2013 an Independent Surface Water Working Group (ISWWG) was established by ERA and the GAC to review surface water management and monitoring at Ranger. Hart and Taylor (2013) reported that the Traditional Owners were concerned that sediments were no longer routinely monitored and recommended that a sediment monitoring program be reintroduced to:

> *"…reliably evaluate possible adverse environmental impacts during the operational phase of the mine, while providing benchmark data to detect possible impacts after closure."*

2015 onward

To address the ISWWG recommendations, Parry (2016) reviewed past sediment studies, data and monitoring guidelines to:

- Identify, collate and document the available information.
- Design a sediment monitoring program that could identify mine related changes in sediment.
- Assess if any such changes had occurred.

- Provide a pre-closure baseline dataset.

Parry (2016) reported:

The historic dataset includes results from a variety of methods but are still useful with statistical analyses demonstrating comparable results. Analysis of the data sets showed the overall metal concentrations generally follow the order: nitric/perchloric (63 µm) > reverse aqua regia (63 µm) greater than 1 Molar HCl (63 µm) > nitric/perchloric (whole) > reverse aqua regia (whole) > 1 Molar HCl (whole).

Whilst the data sets from these variable sources could not readily be normalised, a consistent data set was identified from the ERA monitoring program and analysed using principal coordinate analysis. The principal coordinate analysis showed that for the majority of years Georgetown, Coonjimba, Gulungul and Djalkmarra billabongs (excluding radium-226) had similar compositions, with Mudginberri Billabong separated by higher concentrations of zinc and manganese, non-Ranger Mine sources. The results from this analysis demonstrated that with suitable data bases this type of statistical analysis can be used to determine any patterns of change spatially and/or temporally.

Jones *et al* (2001) 1997 sediment U data represents one of the best background sediment data sets, albeit based on the <63 µm fraction. It also demonstrated no change in metal concentrations in the floodplain billabongs since 1977-78.

The Supervising Scientist billabong sediment sampling in 2007, 2011 and 2013 provides a robust data set, especially for control water bodies in the Magela Creek and Nourlangie Creek catchments. The data clearly shows the distinction between on-site (within the Ranger Project Area) water bodies and unimpacted off-site (outside the Ranger Project Area) water bodies. The 2013 Control Billabongs' data had lower concentrations than in the historic Mudginberri Billabong dataset.

Assessment of all available sediment data from 1982 to 2013 (ERA and Supervising Scientist) showed the following order of billabongs in terms of uranium concentrations: Mudginberri = Gulungul < Coonjimba ≈ Georgetown.

Sinclair (2015) showed that uranium, thorium and metal concentrations in the majority of the Ranger surface samples and sediment cores were low and comparable with concentrations at other creeks within the Alligator Rivers Region.

Lead isotope ratios showed sediments from Georgetown Billabong and the Gulungul Creek tributary in close proximity to the TSF, and to a much smaller degree the younger sections of the MCDS (Magela Creek downstream) core contain some mine derived material. This demonstrated the usefulness of the isotope method for determining the source of erosion products being transported albeit at low concentrations (equivalent to only about 1.1 mg/kg of lead at MCDS).

The Supervising Scientists biological monitoring program provides an indirect assessment of any potential sediment impacts.

Determination of uranium and radium levels in mussels from Mudginberri Billabong has shown consistently low levels with lack of any increase in concentration of U and analysis of isotope ratios in mussel tissues through time (2000 to present) indicating absence of any mining influence on the water and sediment in Mudginberri Billabong[5].

The biological monitoring results from 1988 to present across multiple sites in the Magela catchment have shown that biological communities (fish and macroinvertebrates) have not been adversely impacted as would be expected if sediments were adversely impacted.

Parry (2016) concluded that sediment concentrations in billabongs off the RPA had not increased due to mining and recommended a routine sampling and analyses program based on leading practice.

The recommendations, agreed to by a stakeholder working group, were trialled in 2015 and implemented and refined in 2016. The billabongs sampled in 2016 were Wirnmuyr, and Buba (control sites), Gulungul (exposed site), and Coonjimba and Georgetown (potentially mine affected). Corndorl (a control site) and Mudginberri Billabongs were not able to be sampled due to early rains. However, as noted above the SSB mussel monitoring program indicates the absence of any mining influence on the water and sediment in Mudginberri Billabong.

Esslemont and Iles (2017) compared the metal concentrations at these billabongs with historic data and used stable lead isotope ratios, principal component analysis, and associations with iron and aluminium to interpret the results. The updated dataset was also used to derive background concentrations for metals in sediment based the 80th, 95th and 99.7th percentiles of data from un-impacted sites (control and unimpacted exposed sites, and data from potentially impacted sites prior to any identifiable change shown by time series data for each site). This follows the approach to derive background concentrations in Magela and Gulungul Creek waters (Turner *et al.* 2016). Regional background sediment concentrations based on this information are shown in Table 5-13.

Table 5-13 Regional background values and datasets

Element (mg/kg dry wt. <0.63mm)	**Percentiles**				**Data sets**
	50	**80**	**95**	**99.7**	
Copper	29	37	43	55	Metal concentration data from non mine-affected sediments were evenly represented from the billabongs, and percentiles developed from the pooled data.
Lead	21	30	40	68	
Zinc	18	27	41	73	
Manganese	84	119	174	247	
Uranium	6	9	20	25	
Based on 12 samples from Buba (2007-16), Wirnmuyurr (2007-16), Corndorl (2007-13), Coonjimba (pre 1999), Georgetown (pre 1999), Gulungul (pre 1999), and Mudginberri (pre 1999; Cu, Pb, U only)					

5 Concentrations of other metals in mussels from Mudginberri Billabong were also reported to be low and between 5 – 100 times lower than national food standards in the SSB Annual Report for 2014.

Esslemont and Iles (2017) compared the 2016 and previous sediment-bound metal concentrations against the derived background dataset, national sediment quality guideline values or the site specific uranium guideline value derived by the SSB. The results are shown in Figure 5-15,Figure 5-16,Figure 5-17,Figure 5-18 and Figure 5-19.

In general, sediment concentration in 2016 were generally below the sediment quality guideline values, or historical concentrations, in billabongs where sediment guidelines were lacking except for Buba Billabong.

Concentrations of metals had not increased in sediments in the offsite billabongs in the Magela catchment with concentrations within natural variation (at the low end of the range). Comparisons with historical data show that sediment concentrations of manganese were the lowest, and uranium close to the lowest, recorded for all sites except Buba Billabong.

All uranium concentrations were well below the site-specific guideline value of 94 µg/kg developed by the SSB, with the highest values for 2016 at Georgetown Billabong being less than one fifth of this and Buba Billabong being less than a tenth of this value.

Copper, lead and zinc concentrations in billabong sediments were below the national sediment quality guideline values, and with the exception of one zinc result in Buba Billabong were low relative to historical concentrations. Historical concentrations were consistently below the sediment quality guideline high values (SQG-H), and usually below the sediment quality guideline values (SQGV). As such the results show these are not metals of concern.

Elevated uranium, zinc and manganese concentrations at Buba Billabong, a control billabong not in the Magela Catchment, were not related to mining operation. However, understanding the reasons behind these elevations can help to determine if elevations that may occur at a mine exposed site in future are mining related. The associations of these metals with iron and aluminium were reviewed along with principal component and stable lead isotope analysis. These analyses showed these elevated concentrations are a result of natural accumulation of uranium with iron and aluminium oxides in alluvium, and a possible localised weathering anomaly (hydromorphic anomaly) of manganese and zinc.

Coonjimba Billabong data from the late dry season in 2015 showed some high uranium concentrations compared with historic data, in contrast with 2016 data that showed low concentrations compared with historic data. The 2015 conditions allowed aquatic sediments to be sampled from the dry central channel of the billabong which is usually submerged. In 2016 sediments were collected from the wetted edge of the billabong when the billabong still contained a substantial volume of water, and consequently samples were collected from a relatively high position up the bank and more similar to historic sampling locations. Therefore during 2015, there was a larger dataset and more spatial variation represented from across the billabong than in 2016, and the 2015 dataset identified replicate samples with concentrations above the control range as well as replicate samples with concentrations below the control range.

The 2015 dataset from Coonjimba identified that leachable (1M HCl) sediment-bound uranium concentrations within 460 meters of the RP1 release point were higher than background concentrations derived by Parry (2016), and total uranium concentrations in the billabong channel were in excess of ambient associations with bog-iron and aluminium oxides. Lead isotope ratios from 2016 and 2015 showed that uraniferous (206/207Pb) and thoriferous (208/207Pb) signatures of the sub-clay (<63 µm) sediment fraction were consistent with sediment from a uranium mineralised source. However, the thoriferous (208/207Pb) signature of the sub-sand (<2mm) sediment fraction in 2016 indicated that sand from a non-mineralised source had also contributed to the samples. As such the 2015 Coonjimba Billabong samples contained sediment from a mineralised source mixed with sediment from a non-mineralised source.

In summary the spatial variation of the sediment samples within Coonjimba Billabong are consistent with potential sources of sediment from the minesite, which had mixed with sediment from non-mineralised sources. This is expected to be observed during mine operation in a billabong located within a kilometre of the RP1 release point.

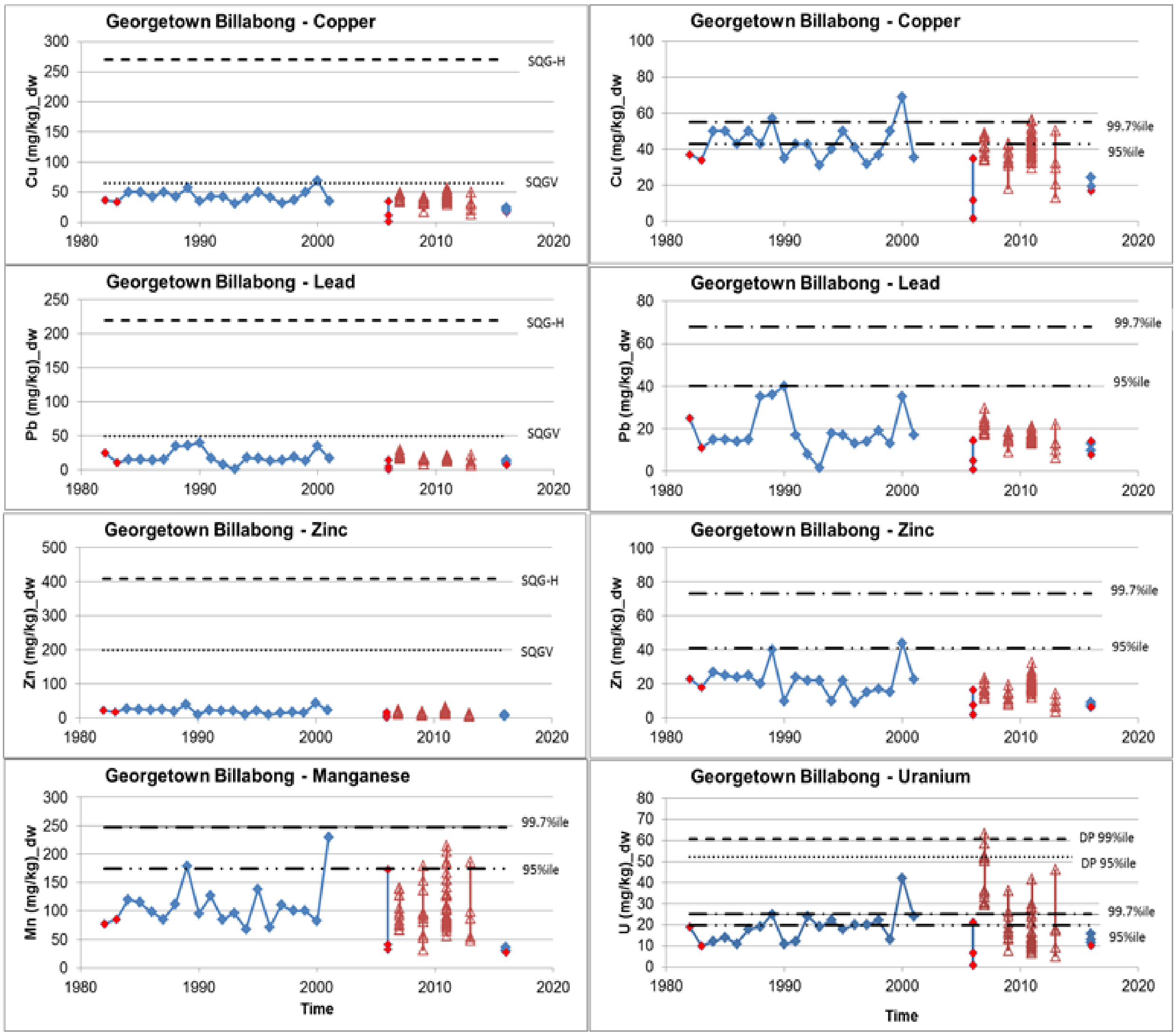


Figure 5-15: Control Charts of TPM concentrations in surface sediments of Georgetown Billabong. ◆ sub-clay (<63 µm) ERA samples, ◆ sub-sand (<2mm) ERA samples, △ sub-clay (<63 µm) SSB samples. Digests before 2006 were by reverse aqua regia and after 2006 were by nitric/perchloric acid.

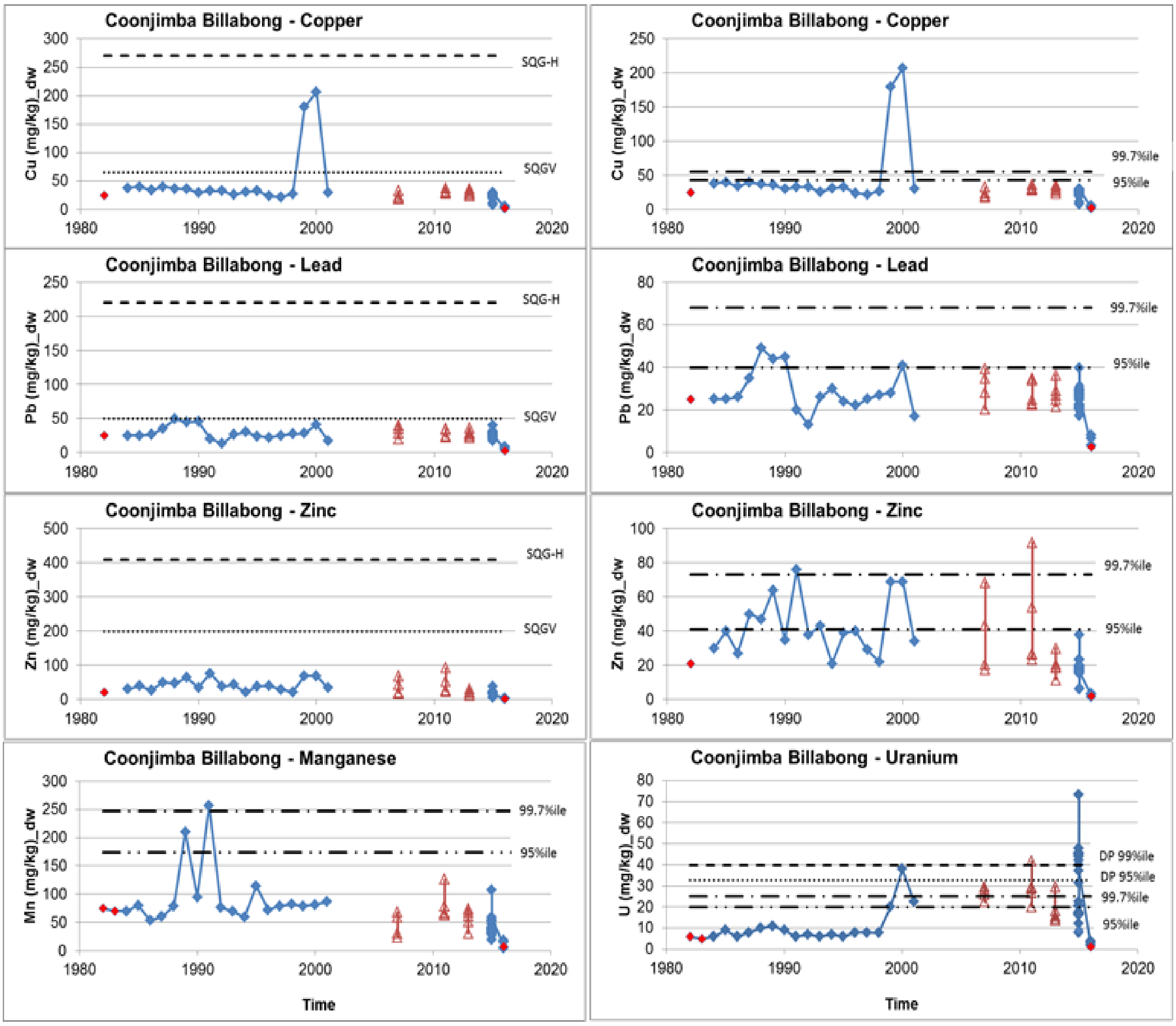


Figure 5-16: Control Charts of TPM concentrations in surface sediments of Coonjimba Billabong. ◆ sub-clay (<63 µm) ERA samples, ◆ sub-sand (<2mm) ERA samples, △ sub-clay (<63 µm) SSB samples. Digests before 2006 were by reverse aqua regia and after 2006 were by nitric/perchloric acid.

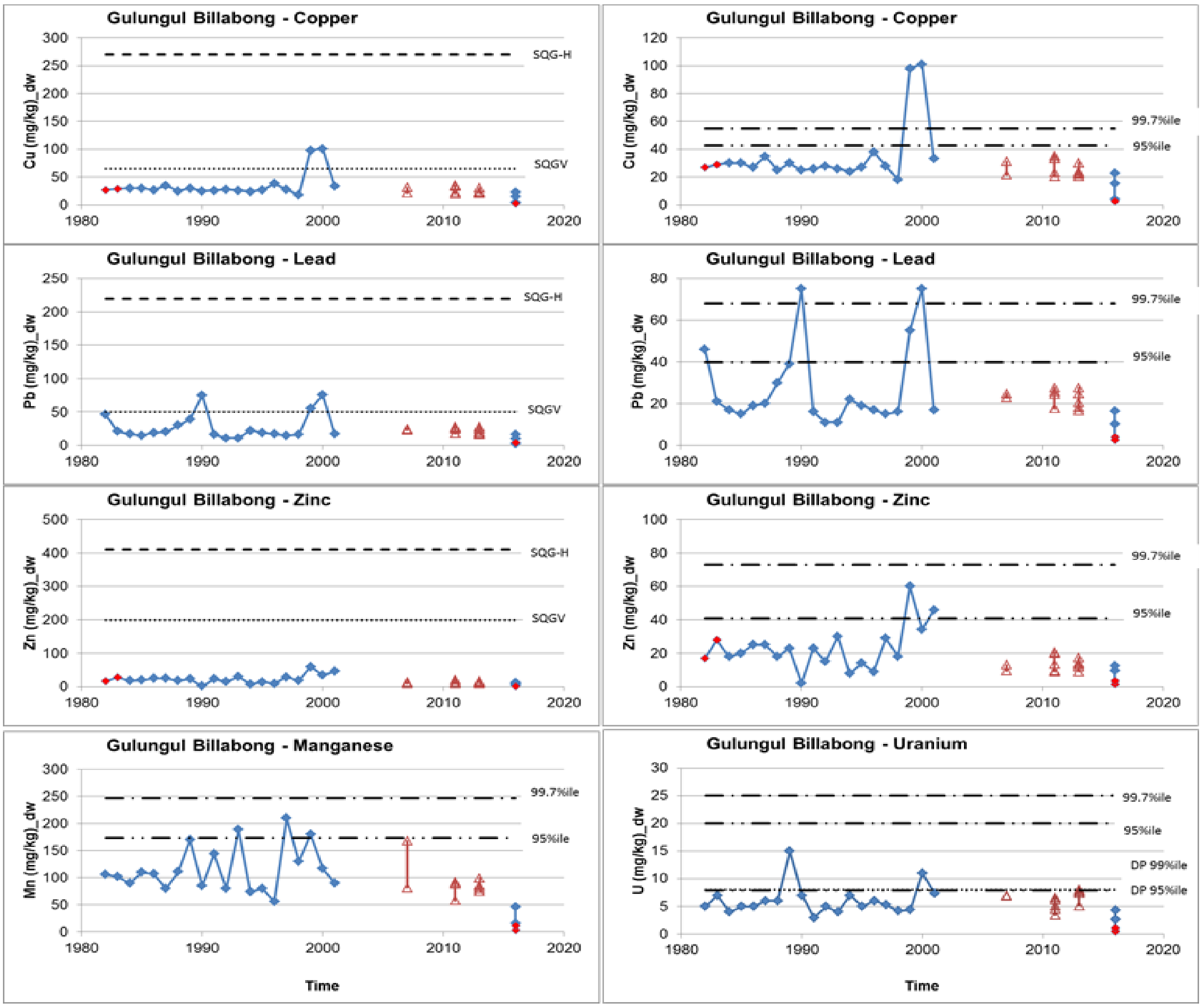


Figure 5-17: Control Charts of TPM concentrations in surface sediments of Gulungul Billabong. Symbols as for Figure 5-13. Digests before 2001 were by reverse aqua regia and after 2001 were by nitric/perchloric acid.

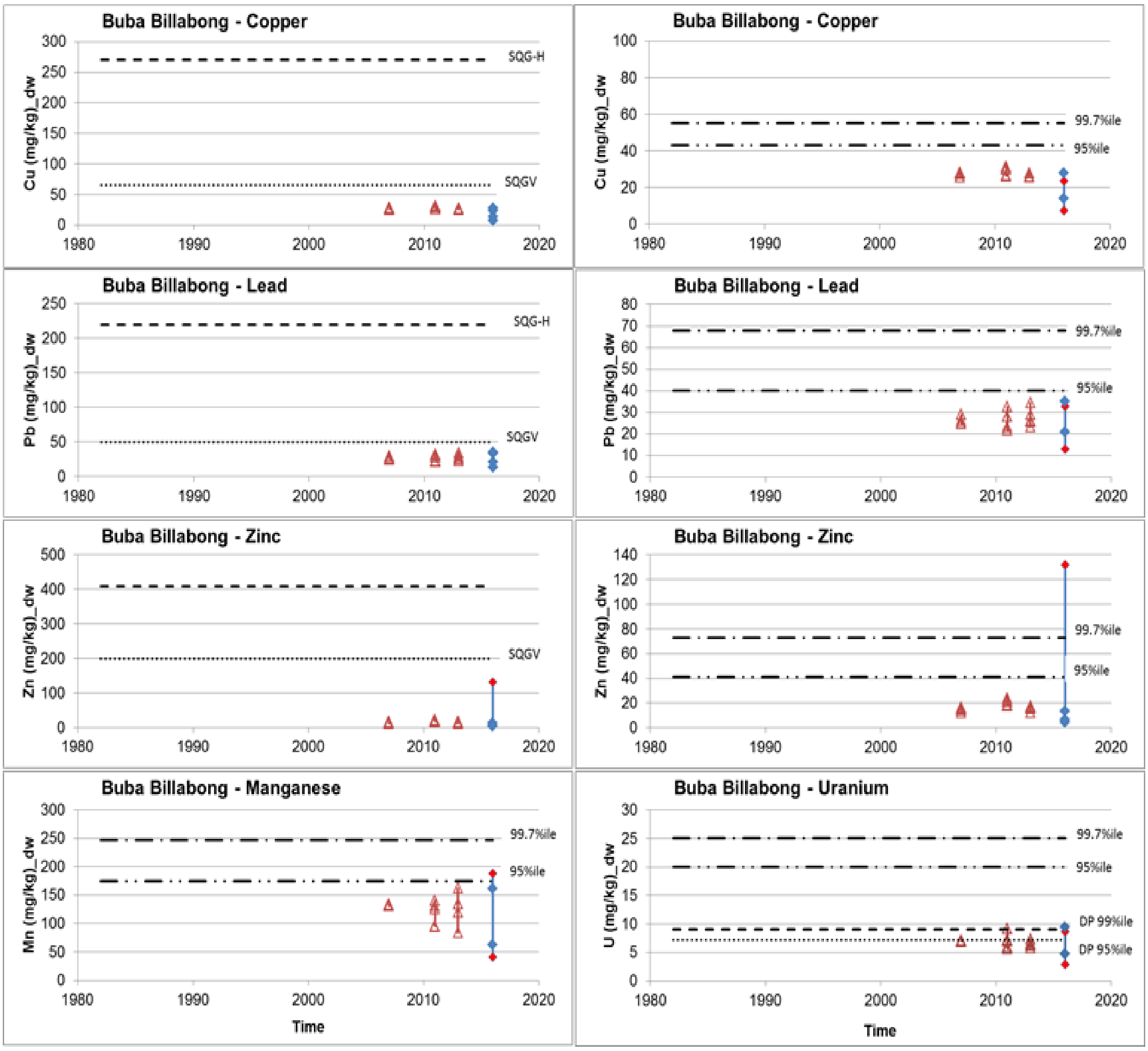


Figure 5-18 Control Charts of TPM concentrations in surface sediments of Gulungul and Buba billabongs. Symbols as for Figure 5-13. Digests before 2001 were by reverse aqua regia and after 2001 were by nitric/perchloric acid.

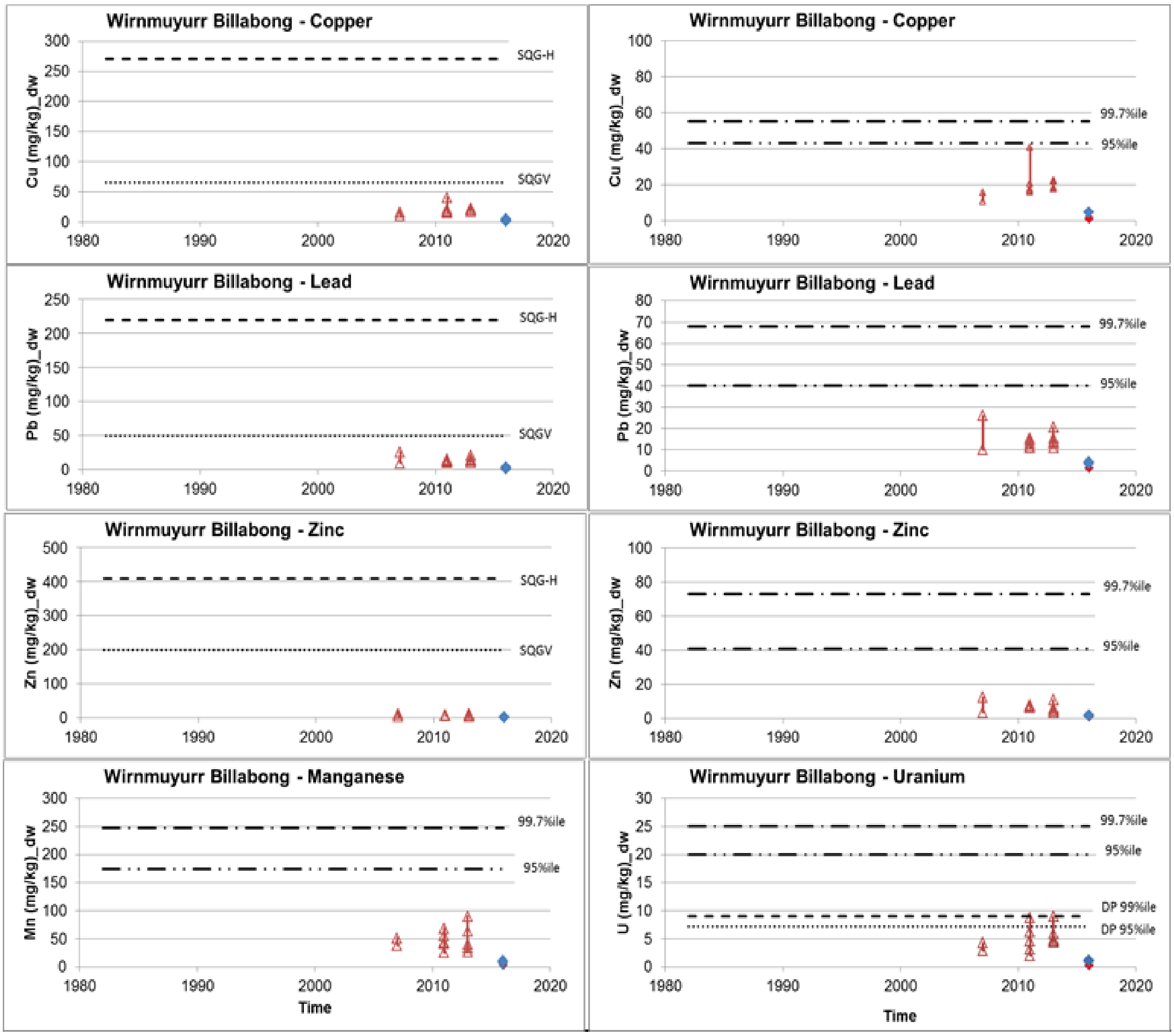


Figure 5-19 Control Charts of TPM concentrations in surface sediments of Wirnmuyurr billabong. Symbols as for Figure 5-13. Digests were by nitric/perchloric acid.

The next sediment sampling program is planned for 2020 and will focus on acid sulfate soil potential and confirming metal concentrations in the onsite waterbodies and creeks and the closest offsite billabong, Gulungul Billabong, refer to section 5.5.2.2.

5.3 Biological environment

5.3.1 Bioregions

Bioregions for the Australian continent have been created as part of a national classification of ecosystems. There are currently 89 bioregions and 419 sub-regions in Australia. Each region is based on similarities in climate, geology, landform, native vegetation and species information. Most of the RPA lies within the northeast section of the 28,520 km^2 Pine Creek Bioregion. Features of the Pine Creek Bioregion include:

- a landscape broadly consisting of hilly to rugged ridges with undulating plains
- vegetation communities that include eucalypt woodland, with patches of monsoon forest
- major land uses that include conservation, pastoralism, intensive rural freehold blocks, horticulture, mining and indigenous freehold, and
- major population centres at Batchelor, Adelaide River, Pine Creek and Jabiru.

The Pine Creek Bioregion, in the Top End of the NT, comprises hilly ridges with undulating plains within the foothills of the Arnhem Land Massif (ERA 2014b, DNREA 2005). Typical vegetation types consist broadly of tall eucalypt woodlands, dominated by Darwin woollybutt (*Eucalyptus miniata*) and Darwin stringybark (*E. tetrodonta*) with patches of monsoon forests, riparian vegetation and tussock grasslands (DNREA 2005). The bioregion supports a high diversity of flora and fauna, with 279 bird species, 100 reptile species and approximately 2,300 plant taxa recorded in 2005. Of those, a total of six plant species and 14 fauna species are threatened. During the wet season (November to March) approximately 90 percent of annual rainfall occurs in this tropical monsoonal bioregion (DEE 2005).

5.3.2 National parks and protected areas

The RPA is surrounded by Kakadu NP, which is an internationally recognised area of natural and cultural importance, and is inscribed on the United Nations Educational, Scientific and Cultural Organisation (UNESCO) World Heritage Register. The RPA is also within 150 km of three other national parks: Warddeken Indigenous Protected Area (approximately 10 km east of the RPA and adjacent to the eastern boundary of Kakadu NP), Mary River National Park (115 km west of the RPA) and Nitmiluk (Katherine Gorge) National Park (approximately 123 km south of the RPA)

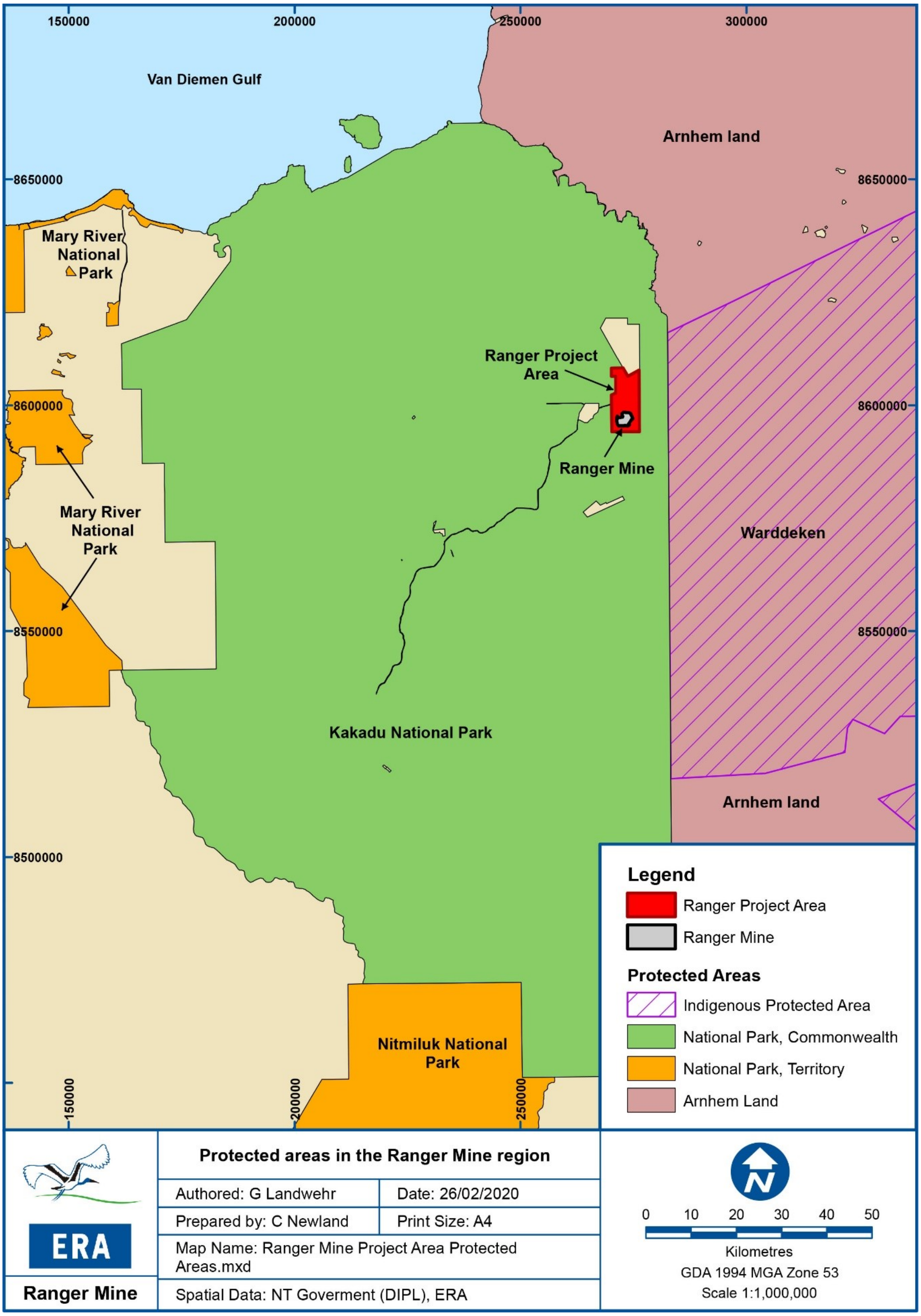


Figure 5-20 Protected areas in the Ranger Mine region

5.3.2.1 Kakadu National Park

The area of Kakadu was established as a national park in April 1979, with construction of Ranger Mine commencing in January 1979. Since the original proclamation, the park has been extended to cover an area of almost 20,000 km^2 of the Alligator Rivers Region; the Alligator Rivers Region is as defined in the *Environment Protection (Alligator Rivers Region) Act 1978*). Over half of the Kakadu NP is held by Aboriginal Land Trusts on behalf of the Traditional Owners and has been leased to the Director of Parks Australia North. Kakadu NP is of great significance for its landforms, its variety of fauna and flora and its rich legacy of Aboriginal art.

5.3.2.2 Ramsar wetlands and sensitive habitat

The entire Kakadu NP is listed as a wetland of international importance under the Ramsar Convention, due to its adherence to the selection of the criteria defining wetlands of international importance (BMT WBM 2010).

Criteria defining Kakadu NP as a site containing Ramsar wetlands of international significance (BMT WBM 2010) are:

- a wetland should be considered internationally important if it contains a representative, rare, or unique example of a natural or near natural wetland type found within the appropriate biogeographic region
- a wetland should be considered internationally important if it supports vulnerable, endangered, or critically endangered species or threatened ecological communities
- a wetland should be considered internationally important if it supports populations of plant and/or animal species important for maintaining the biological diversity of a particular biogeographic region
- a wetland should be considered internationally important if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions
- a wetland should be considered internationally important if it regularly supports 20 000 or more waterbirds
- a wetland should be considered internationally important if it regularly supports one percent of the individuals in a population of one species or subspecies of waterbird
- a wetland should be considered internationally important if it supports a significant proportion of indigenous fish subspecies, species or families, life-history stages, species interactions and/or populations that are representative of wetland benefits and/or values and thereby contributes to global biological diversity
- a wetland should be considered internationally important if it is an important source of food for fishes, spawning ground, nursery and/or migration path on which fish stocks, either within the wetland or elsewhere, depend

- a wetland should be considered internationally important if it regularly supports one percent of the individuals in a population of one species or subspecies of wetland-dependent non-avian animal species

The wetlands of Kakadu NP are also part of an East Asian-Australasian Flyway established to protect areas used by migratory shorebirds (BMT WBM 2010). Due to this international recognition of wetlands in the Kakadu NP these wetlands must not be negatively affected by the closure and rehabilitation of the RPA. However, no environments of special significance (such as significant breeding sites, seasonal habitats or wetlands areas) occur within the RPA or the footprint of the Ranger Mine.

One ecological community in the Alligator Rivers Region is listed as Endangered under the (Commonwealth) *Environment Protection and Biodiversity Conservation Act 1999* (*EPBC Act*). However, this Arnhem Plateau Sandstone Shrubland Complex is restricted to stone country and the nearest suitable habitat occurs approximately 1.5 km from the eastern boundary of the RPA.

World Heritage listing attributes

In June 2013, the World Heritage Committee adopted the retrospective Statements of Outstanding Universal Value for all World Heritage properties inscribed between 1978 and 2006, prior to the launching of the Second Cycle of Periodic reporting in each region (UNESCO 2013). World Heritage criteria that apply to Kakadu NP, include:

World Heritage criterion (i): The Kakadu art sites represent a unique artistic achievement because of the wide range of styles used, the large number and density of sites and the delicate and detailed depiction of a wide range of human figures and identifiable animal species, including animals long-extinct.

World Heritage criterion (vi): The rock art and archaeological record is an exceptional source of evidence for social and ritual activities associated with hunting and gathering traditions of Aboriginal people from the Pleistocene era until the present day.

World Heritage criterion (vii): Kakadu NP contains a remarkable contrast between the internationally recognised Ramsar–listed wetlands and the spectacular rocky escarpment and its outliers. The vast expanse of wetlands to the north of the park extends over tens of kilometres and provides habitat for millions of waterbirds. The escarpment consists of vertical and stepped cliff faces up to 330 m high and extends in a jagged and unbroken line for hundreds of kilometres. The plateau areas behind the escarpment are inaccessible by vehicle and contain large areas with no human infrastructure and limited public access. The views from the plateau are breathtaking.

World Heritage criterion (ix): The property incorporates significant elements of four major river systems of tropical Australia. The Kakadu NP ancient escarpment and stone country span more than two billion years of geological history, whereas the floodplains are recent, dynamic environments, shaped by changing sea levels and big floods every wet season. These floodplains illustrate the ecological and geomorphological effects that have accompanied Holocene climate change and sea level rise.

The Kakadu region has had relatively little impact from European settlement, in comparison with much of the Australian continent. With extensive and relatively unmodified natural vegetation and largely intact faunal composition, the Kakadu NP provides a unique opportunity to investigate large-scale evolutionary processes in a relatively intact landscape.

World Heritage criterion (x): The Kakadu NP is unique in protecting almost the entire catchment of a large tropical river and has one of the widest ranges of habitats and greatest number of species documented of any comparable area in tropical northern Australia. The large size, diversity of habitats and limited impact from European settlement of the Kakadu NP has resulted in the protection and conservation of many significant habitats and species.

The park protects an extraordinary number of plant and animal species including over one third of Australia's bird species, one quarter of Australia's land mammals and an exceptionally high number of reptile, frog and fish species. Huge concentrations of waterbirds make seasonal use of the park's extensive coastal floodplains.

5.3.3 Terrestrial ecology

This section provides an overview of the terrestrial ecosystems of the RPA and surrounding region. Discussion on ecosystem establishment, including revegetation trials and seed provenance is provided in Appendix 5.1. This also includes a fine scale assessment, including plant species composition and relative abundance in the RPA, and surrounding natural analogue sites.

5.3.3.1 Vegetation communities

Schodde *et al*. (1987) described four vegetation types in the RPA dominated by eucalypt open forest and/or woodland (Figure 5 14). Similarly, Firth (2012) described the main vegetation/ habitat types on the RPA as comprising of woodland and open forest, mostly co-dominated by *E. tetrodonta* and/or *Eucalyptus (E) miniata*. The RPA is surrounded for the most part by vast unbroken and undeveloped tracts of the same eucalypt woodlands and open forest savannas that cover at least 180,000 km2 in the NT alone (Hart & Jones 1984). The topography of the RPA is relatively simple and as with vegetation, mirrors that of the region as a whole.

Vegetation types are described below and the area and proportion of each vegetation type on the RPA and in Kakadu NP are given in Table 5-14

Habitat 1: Myrtle-Pandanus Savanna/Paperbark Forest/Coastal Deciduous Rainforest

Paperbark forests line freshwater creek systems and the edges of billabongs and are dominated by Melaleuca spp. The canopy can be 15 to 20 m in height and can vary greatly from open to almost closed. The shrub layer varies from sparse to dense and comprises *Acacia* spp., *Ficus* spp. on marginal areas and the ubiquitous freshwater mangrove *Barringtonia acutangula*. *Pandanus aquaticus* and *B. acutangula* line streams and channels. In zones edging woodland (which is often the case in the RPA), the trees are wider spaced and often form an ecotone with myrtle-pandanus savanna. In this ecotone area other eucalypts, bloodwoods and other savanna trees co-dominate with the paperbarks. Coastal deciduous rainforest habitat is not present in the RPA according to the description of Schodde *et al*. (1987).

Habitat 2: Myrtle-Pandanus Savanna

Consists of grassland with small open pockets of woodland, mixed shrubland and rainforest trees, interspersed with strips of Pandanus (*Pandanus spiralis*) along the edges of floodplains and with paperbarks (*Melaleuca* spp.) along creeks and streams. Tall trees from genera such as *Corymbia* and *Eucalyptus* are sparingly present. A very patchy shrub layer of *Melaleuca viridiflora*, *M. nervosa* and *P. spiralis* occur. Common grasses include annuals from genera such as *Digitaria, Ectrosia, Panicum, Schizachyrium* and *Sorghum* and perennials grasses including those from genera such as *Eriachne* and *Themeda*. Sedges (Cyperaceae) are also a common component of the ground cover.

Habitat 3: Open Forest

Tall (12 to 20 m) open forest dominated by *E. miniata* and *E. tetrodonta* and with other species of eucalypts present in the canopy. The only frequent non-eucalypt that occurs in the canopy is Ironwood (*Erythrophleum chlorostachys*). The shrub layer consists of Acacia spp., *Calytrix exstipulata, Croton arnhemicus, Gardenia* spp., *Livistona humilis, Petalostigma quadriloculare, Planchonia careya, Terminalia* spp. and *Xanthostemon paradoxus*. Ground cover is usually sparse, inconspicuous and comprises mostly annual grasses of Sorghum spp. and other herbaceous plants.

Habitat 4: Woodland

This habitat typically lacks a distinct canopy and is more stunted (usually less than 12 m) than open forest, being dominated by bloodwoods (*Corymbia* spp.), but also contains eucalypts such as *E. miniata, E. tetrodonta* and *E. tectifica*. However, it is quite variable in structure and can be tall on slopes to the point where it grades into open forest. The shrub layer is the same as in open forest but much sparser. The palm *Livistonia humilis* is common and pockets of

P. spiralis may also be present. The ground cover is much denser than in open forest, containing mainly annual grasses, e.g. Sorghum spp. In stunted woodlands perennial grasses *Heteropogon triticeus* and *Sehima* sp. dominate.

5.3.3.2 Flora species

Native flora species

There has been a substantial survey and monitoring of the terrestrial flora across the RPA over the past 15 years. In a 2013 survey of lowland riparian and woodland areas within the RPA, 292 flora species from 30 families were identified (Eco Logical Australia 2014). These species are common in surrounding Kakadu NP and did not include any threatened or rare species. Approximately 1,600 terrestrial and aquatic flora species have been recorded in Kakadu, including 15 species considered rare or threatened (Director of National Parks 2016). These conservation significant species have not been recorded within the RPA.

On the basis of previous studies integrated from previous studies near the RPA a total of 461 flora taxa from 80 families and 195 genera have been recorded and identified to a minimum of genus level if not species and subspecies (see Appendix B). The flora is representative of a range of underlying environments ranging from riparian, seasonally wetter lowlands and a range of forests and woodlands on the slopes and ridges. There are a few local restricted communities associated with extreme site conditions including outcrops and shallow soils. The lifeforms summarized in Appendix C have been extracted from the NT Flora database (Northern Territory, 2020), the WA Florabase (Western Australian Herbarium, 1998–) and key references such as Brock (2001) and provides observations on site preferences of the respective species in relation to underlying landforms, soils and soil moisture records.

Conservation significant species

No terrestrial or aquatic flora species of conservation significance listed under the *Territory Parks and Wildlife Conservation Act 1978* (NT) (*TPWC Act*) or the *EPBC Act* have been recorded in the RPA.

Table 5-14 Area and proportion of vegetation communities on the RPA and Kakadu NP

Community (Schodde *et al.* 1987)	**RPA[1] (ha)**	**RPA[1] (%)**	**Kakadu NP (ha)**	**Kakadu NP (%)**	**RPA community as a percentage of equivalent habitat in Kakadu NP (by area)**
Myrtle-pandanus savanna/ paperbark/coastal rainforest	434	6	39,487	4	1.1
Myrtle-pandanus savanna	1,863	26	170,802	16	1.1
Open forest	3,018	42	336,269	32	0.9
Woodland	1,870	26	508,000	48	0.4

Note 1 – undisturbed (non-mine) sections only

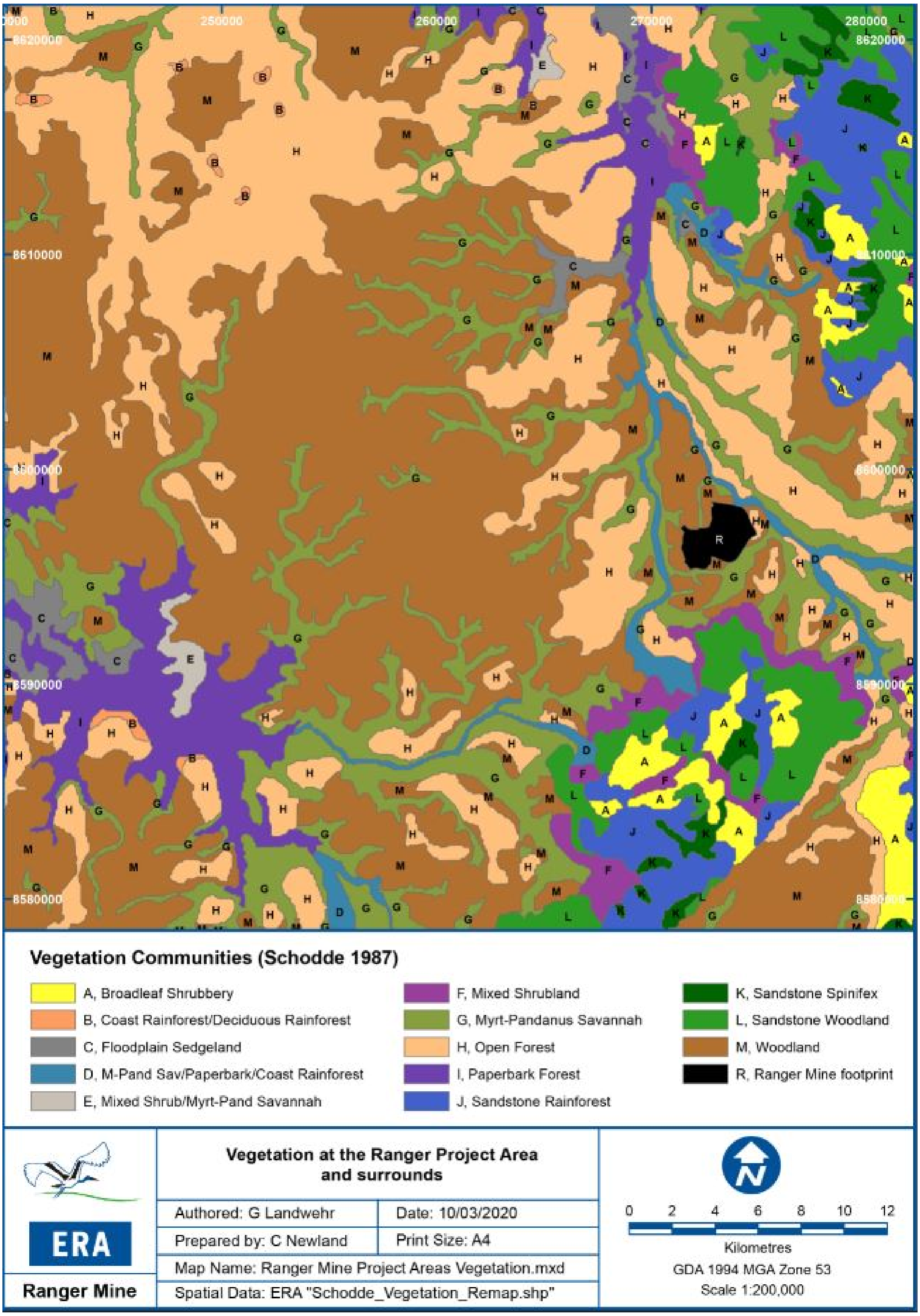


Figure 5-21 Vegetation of the RPA and surrounding Kakadu NP (Schodde *et al*. 1987)

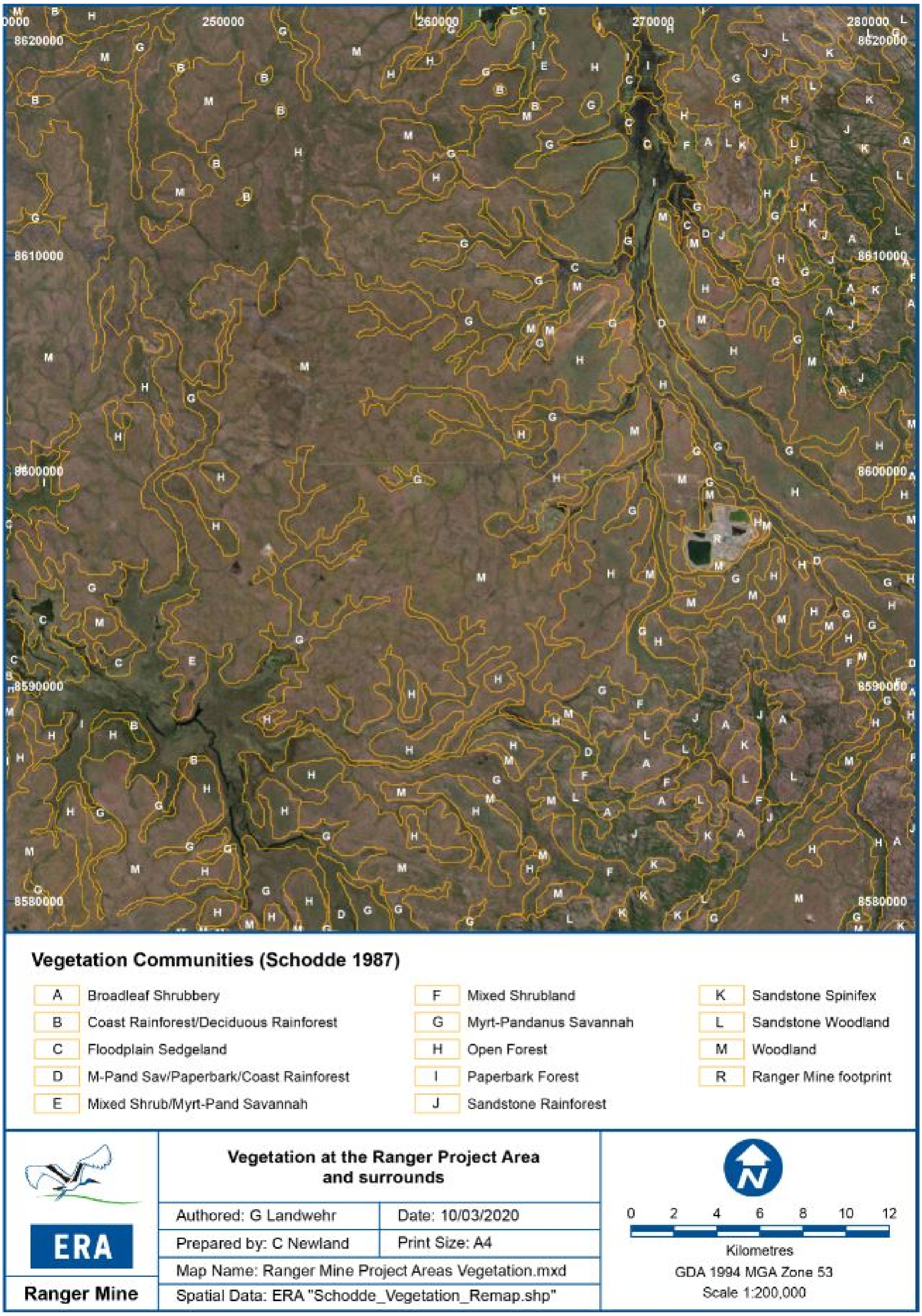


Figure 5-22 Vegetation types over aerial of the RPA and surrounding Kakadu NP

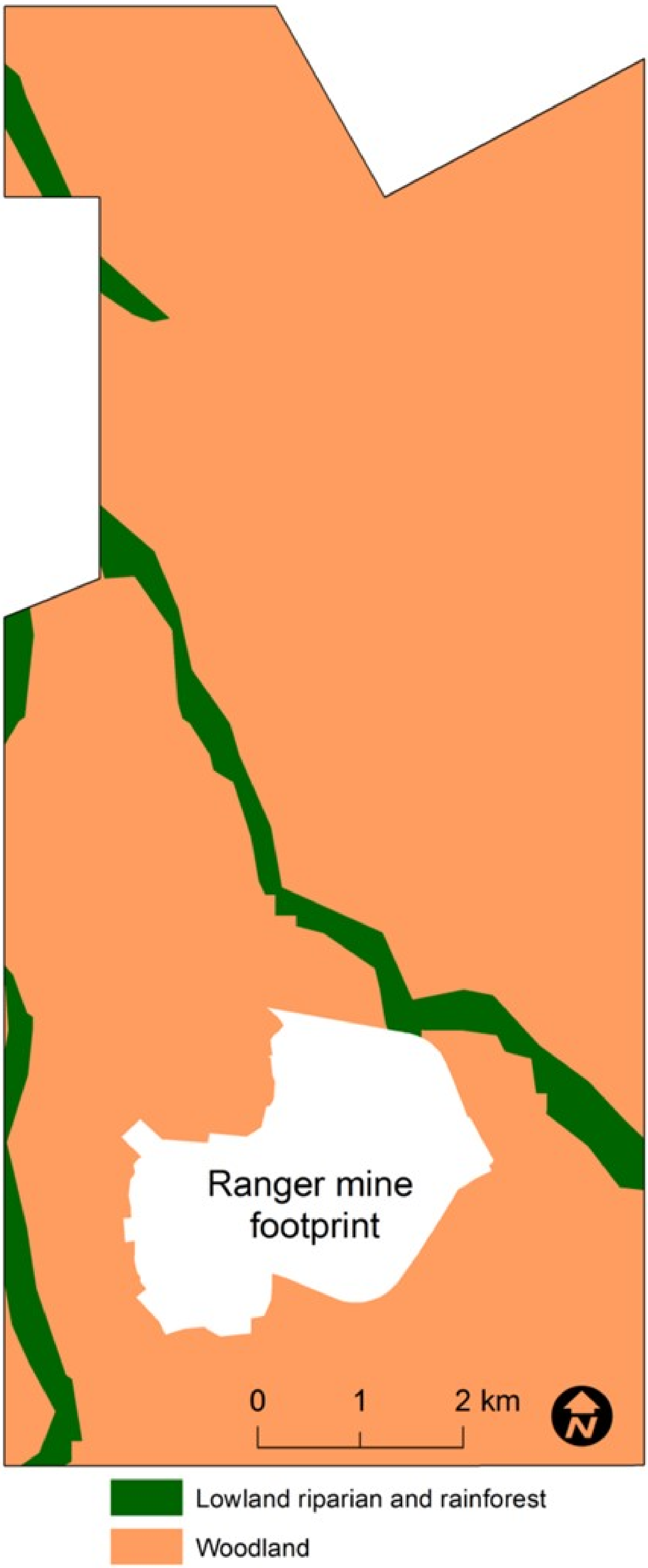


Figure 5-23 Vegetation habitat map of the RPA (based on Brady *et al.* 2007)

Weed species

A weed is an exotic or native species that colonises and persists in an ecosystem in which it did not previously exist. These invasive plants typically produce large numbers of seeds and are excellent at surviving and reproducing in disturbed environments. Weeds potentially reduce biodiversity by competing with or displacing endemic species and may also affect natural processes such as fire intensity and stream flows. The restriction to recreational movement of people may also result from weed infestations.

One of the most significant threats to the natural and cultural values of the Kakadu NP is weeds (Director of National Parks 2016). Compared to other national parks in the region, Kakadu NP has a low proportion of weeds. However, there are still significant impacts by invasive weeds to some of the landscapes within the national park.

The RPA has been surveyed by ERA annually for weeds since 2003, and approximately 80 species have been recorded during this time. Weeds of National Significance (WoNS) are categorised under the Federal *EPBC Act.* Gamba Grass (*Andropogon gayanus*) is the only WoNS previously recorded in the RPA with the recorded presence restricted to isolated plants on roadsides or in the vicinity of the Jabiru Airport. With successful weed control, there has been no plants nor viable seeds of this species detected for a number of years. There are five grass species listed as Key Threatening Processes to Australia's biodiversity also under the *EPBC Act*. Gamba Grass is one of these, whilst the other four species have not been recorded on the Ranger Minesite.

The Northern Australia Quarantine Strategy (NAQS) was established in 1989 to manage the risks of biosecurity particular to northern Australia due to the proximity to neighbouring countries. The NAQS is administered by the Federal Department of Agriculture. No weeds listed within the NAQS have been recorded within the RPA. There are also six weed species listed under the Tropical Weeds Eradication Program (DAF 2019) which, to date, have not been recorded on the RPA.

In the NT, the *Weeds Management Act 2001* is administered by the Department of Environment and Natural Resources. Six species listed under this legislation as Class A/B/C (eradicate/growth and spread to be/not to be introduced into the NT) have been recorded within the RPA (Table 5-15). In addition, there are a further nine weed species that have been identified by ERA as requiring active treatment and/or removal when detected on the RPA. The potential risk of weeds to closure success is further discussed within Section 7. Weed management strategies are discussed within Section 9.

An un-identified plant was observed, and a sample submitted to the NT Herbarium for identification was identified on 17 April 2019 as *Spigelia anthelmia* (Indian Pinkroot). The identification of *Spigelia* at the Ranger Mine is the first known occurrence of this weed in Australia. External stakeholders were notified. *Spigelia* is native to the tropical and sub-tropical Americas and is known to have spread to parts of Africa and South East Asia (including Thailand, Philippines and PNG). Since identification the Ranger Project Area has been surveyed. *Spigelia* was detected in a number of locations and all located plants were treated. ERA aims to eradicate the *Spigelia* infestation. A timeframe to achieve eradication is 5-6 years given that *Spigelia* seed may remain viable for at least 3 years.

Table 5-15 Actively Managed Weeds in the RPA

Scientific name	Common name	*Weeds Act 2001* (NT) listing
Andropogan gayanus	Gamba Grass	Class A, Class C and Weed of National Significance
Calopogonium mucunoides	Calopo	–
Cenchrus pedicellatus	Annual Pennisetum	–
Cenchrus polystachios	Mission Grass	Class B, Class C
Chamaecrista rotundifolia	Wynn's Cassia	–
Crotalaria goreensis	Rattlepod	–
Hyptis suaveolens	Hyptis	Class B, Class C
Ipomoea quamoclit	Cupid's Flower	–
Macroptilium atropurpureum	Siratro	–
Senna obtusifolia	Sicklepod	Class B, Class C
Sesamum indicum	Sesame	–
Sida acuta	Spinyhead Sida	Class B, Class C
Sida cordifolia	Flannel Weed	–
Spigelia anthelmia	Indian Pinkroot	–
Themeda quadrivalvis	Grader Grass	Class B, Class C

5.3.3.3 Vegetation ecology

At the broad scale, the distribution of the more dominant native forest and woodland communities near Ranger in the wet-dry tropics of northern Australia is controlled predominantly by three factors:

- The underlying geomorphology (which influences site hydrological features and soil fertility);
- The seasonality and predictability (inter-annual variability) of climate; and
- The frequency and intensity of fire.

These factors govern the structural complexity (e.g. height, biomass, number of strata, size class distributions, root depth and distribution patterns), species compositions and the functioning of the vegetation (e.g. water use, nutritional uptake, regeneration strategies, and phenology). These are the environmental factors that have moulded (and constrained) the native vegetation, and its responses to disturbances. Within areas with similar climate and fire regime, geomorphology plays the major role in determining vegetation communities. This is reflected in distinctive catenary sequences of forest and woodland vegetation that are found throughout the lowland parts of Kakadu NP (Bowman *et al*. 1987) and is the basis of 'land system' and other mapping that has been undertaken in the region (Story *et al*. 1969).

However, the way in which individual plant communities have been delineated and classified in these surveys has depended on factors such as the scale of the mapping (1:20,000 to 1:1,000,000) and the particular purpose for which the survey was conducted (e.g. broadscale vegetation description, fire risk management, fauna habitat mapping or mine EIS).

Vegetation dynamics and responses to disturbance

Disturbance events are the major agents of change in vegetation communities. The severity of their effects on plant community structure and composition depends upon (a) the type of disturbance, (b) its intensity, spatial extent and frequency of recurrence, and (c) the resistance and resilience of the affected plant community and its individual component species. Understanding how native vegetation responds to, and recovers from, disturbance is fundamental in designing ecologically-based revegetation programs.

Plants of forests and woodland communities of the wet dry tropics have been successful and survived because they have adapted to the disturbance events (eg fire, cyclone, El-Nino drought) that are characteristic of the region. The strategies adopted by the flora of the region fall into two broad categories, 'persistence' and 'opportunism'.

Persistence

All of the long-lived framework species rely on a 'persistence' strategy based upon the ability to resprout from lignotubers and root suckers (Lacey & Whelan 1976; Fensham & Bowman 1992). Although they produce and shed seed, seedling regeneration is considered rare in *Eucalyptus tetrodonta* and *E. miniata* (Fensham 1992). The chance of an individual seedling surviving by the end of the first dry season is extremely low, considering their slow growth and the combined pressures of a lack of water and the likelihood of fire. In their review of previous revegetation research at Ranger Mine, Reddell and Zimmermann (2002) noted that, of 5000 young seedlings of framework species observed in natural woodland plots, not one survived after 2 years. Other research in north Australian eucalypt savannas has found that seedlings of *Eucalyptus miniata* and *Acacia oncinocarpa* grown from seed were reduced by 75% and 65% respectively by the end of the first dry season, and this had further dropped to only 11% and 33% survival by the middle of the following dry season (Setterfield 2002). In contrast, woody resprouts of framework species are common components of the ground and shrub layers in these woodlands. Although often damaged or filled by the frequent low intensity fires that are characteristic of the management regime in the region, once they reach approximately 3m in height they become increasingly fire resistant and are able to 'break-out' from the fire-suppressed ground layer. To reach such a height, fire would need to be excluded from a woodland site for 3 to 5 years (Williams *et al* 2003a).

The success of the 'persistence' strategy over seed regeneration for long-lived species probably related to a number of factors including (a) the hostile environment of these woodlands (eg very infertile soils, extended annual dry periods, high fire frequency, high densities of very competitive grasses and forbs in the ground-layer) for establishment of the generally slow-growing seedling of long-lived plants, and (b) the marked competitive advantages for a root sprout of being able to access a well-established existing root system.

The persistence strategy allows long-lived species to capture and store resources, tolerate repeated low-intensity fire, and cope with other less frequent bit potentially more damaging disturbances (such as cyclones, El Nino events or high intensity wildfires). Given that the annual mortality rate of canopy trees in these woodlands is estimated to be around 1% (and up to 15% after particularly intense fires (Lonsdale & Braithwaite 1991; Williams *et al* 1999), current prescriptive fire management strategies which result in the continued suppression of woody sprouts in the ground layer could in the long-term have severe demographic consequences for the composition, structure and functioning of these plant communities.

Opportunism

The grasses and forbs that dominate the ground layer, together with some short-lived shrubs and trees (eg *Acacia holosericea* and *Grevillea pteridifolia*), rely largely on an 'opportunism' strategy for regeneration. This strategy is based on the ability to rapidly colonise a disturbed area and capture resources in the ground layer of the woodland that have been made 'available' by the disturbance event. Species with this strategy tend to produce large seed crops, some of which can form a soil seed back, and have high growth rates. The frequency and intensity of fire has a major effect on the composition of the opportunists which successfully capture a disturbed site (Andersen *et al* 1998; Fensham & Bowman 1992; Grant & Loneragan 2001; Lonsdale & Braithwaite 1991; Williams *et al* 1999; Williams *et al* 2003b). This strategy explains the significant year-to-year changes and the high spatial heterogeneity in the plant diversity in the ground layer of savanna woodlands.

The long-term dynamics of woodland vegetation in the wet-dry tropics results from the interaction between these two broad strategies. Framework species dominate the site and its resources and are very resistant and/or resilient to most natural disturbance events, including cyclones, El Nino drought and relatively intense fires. Recruitment of these species is predominantly by suckering from underground stems and they give the woodlands a high degree of long-term structural and functional stability. In contrast, 'opportunist' species form an extremely dynamic ground layer, changes in which are driven by frequent fire. Although contributing little to the overall stability of the plant community, this ground layer provides habitat and food resources for many of the native fauna. As a consequence, the predictability of the response of a woodland site to severe disturbance is linked directly to the size and dominance of the framework species (eg. Russell-Smith 1995; Williams *et al* 1999). Only when the soil profile is removed and the underground perenniating organs destroyed (eg in road cuttings, borrow pits, minesites), do the framework species lose their competitive advantage. In these situations, slow recolonisation by growth of suckers from adjacent undisturbed areas is likely the main regeneration strategy However, successful establishment of framework species from seed may occur in some of these highly disturbed areas, but only in situations where there are:

- high light conditions;
- some protected microsites for germination and early growth;
- minimal competition from aggressive, faster-growing species; and

- protection from fire for at least three to five years.

Despite the functional importance of framework species for the long-term sustainability and stability of the plant communities, they are not necessarily the major components of species diversity in these forest and woodlands. Annual and perennial grasses and forbs in the ground-layer often dominate total plant species diversity. However, these components can be very ephemeral in their nature, resulting in considerable year-to-year variation in both species diversity and composition, even at a single natural woodland site (e.g. Williams *et al* 2003b). In particular, the frequency, timing and intensity of fire can cause large changes in the composition of the ground stratum in these woodlands within a single year. As a result, measures of total species diversity and composition can be quite dynamic and variable in a manner that is largely unrelated to the overall functional performance of the plant community (which is controlled by the framework species).

5.3.3.4 Fire ecology

Fire is a major exogenous feature of Australian eucalypt-dominated ecosystems, especially subtropical savanna woodlands (e.g. Gill 1981; Bradstock *et al*. 2002). Removal of vegetation and litter by fire strongly influences nutrient cycling in savanna ecosystems of northern Australia (Cook 1994). The frequent occurrence of fire has driven the evolution and development of savanna woodland and has resulted in the fire-tolerance and reproductive adaptations that enable the range of plant and animal species found in these systems to persist.

In northern Australia, savanna forests and woodlands are often burnt due to traditional burning of country by indigenous peoples, prescribed burning for infrastructure protection and biodiversity conservation, and wildfires. Tropical savannas worldwide are intentionally burnt every 1 to 3 years (Andersen *et al*. 1998).

Intensity, frequency and timing are all important factors that impact on the influence fires have on the environment (Gill 1981; Bradstock *et al*. 2002; Woinarski *et al*. 1999). Intensity is often related to timing, for instance late dry season burns are usually more intense as fuel is very dry, but can also be influenced by the type of fuel (e.g. fire-promoting grasses such as gamba grass (*Andropogon gayanus*). Deliberately lit fires usually occur earlier in the dry season than wildfires, and therefore are generally less intense and less destructive to vegetation.

Two major research projects in the Northern Territory, Munmarlary and Kapalga, have examined savanna dynamics in relation to different fire regimes at landscape scales (e.g. Bowman and Panton 1995; Andersen *et al*. 1998, 2003, 2005). Sites at Kapalga that had been unburnt for a number of years were found to have less grass cover (7% in November and 13% in March) than sites that had been burned annually (for 5 years) in the early or late dry season (Setterfield 2002). These previously-burned sites had 11% and 15% grass cover, respectively, in November and over 25% for both by the end of the wet season in March.

The frequent dry-season fires often remove any accumulated litter or grass biomass. Nutrient cycling in tropical, fire dependent ecosystems, such as the eucalypt-dominated woodlands of Kakadu NP, is driven by this disturbance regime (Cook 1994). Annual litter accumulation can be significant (depending on vegetation composition and structure), especially due to grass, and fallen leaves and branches. In the humid wet season, this organic material is rapidly decomposed by soil micro-organisms, providing significant nutrient input, much of which is available to plants at the precise time they are growing most rapidly and require it. As the dry season progresses and soil moisture is depleted, and with the removal of the litter layer by fire, microbial activity declines (Cook 1994).

Fire management

The RPA is surrounded by the eucalypt savanna dominated landscape of Kakadu NP. High annual wet season rainfall promotes extensive vegetation growth, particularly from annual grasses dominated by Sorghum (*Sorghum intrans*). The subsequent curing of the vegetation during the long dry season (May to September) results in a highly flammable landscape, where fire is an annual event (Russell-Smith *et al*. 1997) and a major force in shaping and altering the natural landscape (Edwards *et al*. 2003). Risk of fire becomes especially severe in September to November due to a combination of low humidity, average maximum temperatures above 35 °C and low soil moisture (Gill *et al*. 1996).

Changes to fire management practices in Kakadu NP since the late 1980s have resulted in more frequent early dry season fires and fewer late dry season fires (Russell-Smith *et al*. 1997). The management approach in Kakadu NP has been to copy the indigenous burning regime by by undertaking early dry season burns which can be accomplished by using helicopter incendiary burning combined with on-ground burning (Edwards *et al*. 2003). Fire is estimated to occur over 55 percent of the park annually (Russell-Smith *et al*. 1997, Lehmann *et al*. 2008 and NAFI 2015) .

Despite the adoption of early dry season burning by management agencies, total fire frequency (which includes both early and late dry season fires) has been shown to have a deleterious impact on the environment (Andersen *et al*. 2005, Lehmann *et al*. 2008). A higher early dry season fire frequency increases grass fuel levels, which in turn encourages higher intensity fires. Such a fire regime may have a similar negative impact on flora and fauna as infrequent late dry season fires (Woinarski *et al*. 2010) and frequent fire has adversely affected sensitive flora species in sandstone escarpment habitats (Russell-Smith *et al*. 1998). Further to this, a high fire frequency has been shown to have a propensity for producing a grass-fire cycle (D'Antonio & Vitousek 1992) where trees and shrubs are replaced by annual grasses. The presence of grassy weeds such as Mission Grass (*Pennisetum polystachion*) and Gamba Grass (*Andropogon gayanus*) can exacerbate the effects of a grass-fire cycle (Rossiter *et al*. 2003)

Fire within the RPA is managed by ERA primarily for asset protection, and includes fuel reduction burns, excluding fire from certain areas and maintaining a network of graded firebreaks. Fuel reduction burns are usually undertaken in the early dry season to produce cooler fires with smaller burnt areas (patchy) and to remove fuel without damaging the over- or under- storey vegetation. Burns along the RPA boundary are typically coordinated with Parks Australia aerial burns in Kakadu NP and are designed to minimise the risk of unmanaged late dry season fires travelling into the RPA. The non-operational area of the RPA north of

Magela Creek is burned by Parks Australia (in co-operation with ERA) as part of annual burning programs.

5.3.3.5 Ecohydrology of natural tropical savanna ecosystems

Plant responses to water stress in the wet-dry tropics

A particularly strong influence on vegetation survival in the wet-dry topics is water availability. The survival of vegetation is dependent on the water balance in the dry season, especially towards the end of the dry season when the soil water stress is at its highest. Plants generally evolve to have adaptions suited to survival in their particular environment. In the seasonally wet-dry tropics, this includes strategies to survive what can be extremes of inundation or 'drought', or more-nuanced variations such as length of dry season, or timing of the wet season onset. Most plants have evolved physiological responses to cope with a broad (natural) range of scenarios. During the dry season plants resort to strategies of ever-decreasing water demand including stomatal closure, loss of leaves, and progressively developing a deeper root system.

A key adaptation is strategies to avoid a catastrophic cavitation of the water-conducting xylem system by balancing canopy water loss and root absorption. As soil moisture is reduced, trees reduce their water loss first by stomatal closure, then progress to sacrifice non-vital, peripheral organs (such as leaves, twigs, small branches to larger ones and above ground stems) to slow down water loss and soil water depletion and survive through the drought (Tyree and Sperry 1988). Vegetation, even the evergreen trees (such as *E. miniata* and *E. tetrodonta*), lose large amounts of leaves to reduce transpiration (water loss from tree canopy), to maintain a balance between root water uptake and canopy water loss (Thomas and Eamus 1999). As a result, although the amount of soil PAW is very low, it is sufficient for the survival of the trees.

Another key strategy to reduce water stress is to develop roots that can access plant available water as it retreats down the soil profile with the progress of the dry season. Root soil water extraction is energy driven; water is pulled by a tension gradient created between the leaf surface to the root tips. Roots first extract the soil water from nearer the soil surface where water is mostly readily available (water potential is high or less negative) and thereafter access water progressively deeper into the ground as the upper soil profile dries out. Plants will not generally establish roots to a depth below a layer that has already provided sufficient soil-water. That is, if soil-water is available in the top four or five metres of the soil profile, plants should not need to root any deeper than this. However, if water is more readily available below that depth, i.e. if a plant can spend less energy to access that water from depth than from an upper dry soil layer, then the root will go and reach that layer, as long as the level of hydraulic tension within the plant xylem vessels does not reach a catastrophic level that will kill the plant (runaway of xylem embolism, Tyree and Sperry 1988). It is well-known that plants have evolved in such a way that they can maintain the balance of water demand and supply to avoid such a catastrophic result (Tyree and Sperry 1988).

In the savanna woodlands typical of Kakadu NP (and the targets of the revegetation efforts at Ranger), by far the bulk of roots are present in the upper one metre of the substrate during the wet season, when growth rates are at a maximum (Janos *et al*. 2008; Hutley 2008). This is in part due to the ferricrete layer (duricrust) that occurs at about 1 to 1.5 m below the soil surface throughout the region (refer Figure 5-24) which limits root development further down but can allow penetration by deeper-tapping roots through macropores (Werner and Murphy 2001; Hutley 2008; Hutley *et al*. 2000). It has been observed that many important tropical savanna species in the NT Top End's soils are able to root to depth of up to five or six metres (Hutley *et al*. 2000; Kelley *et al*. 2002; Kelley *et al*. 2007)

Hutley (2008) summarised the key features of savanna vegetation water use and carbon allocation strategies for vegetation adapted to the Top-End seasonality (refer to Figure 5-25). One of the features is that during the wet season trees maximise growth and water uptake from shallow soil which is nutrient rich. During the dry season the shallow soil water is quickly depleted, and trees stops growing and access water from depth. Water is accessed from depth for trees to maintain photosynthesis and, under more severe conditions, maintain the viability of vital organs to survival the long dry season. Although the water uptake (use) is very low from depth and nutrients are very limited, sub-soil water storage is critical for the survival of the vegetation.

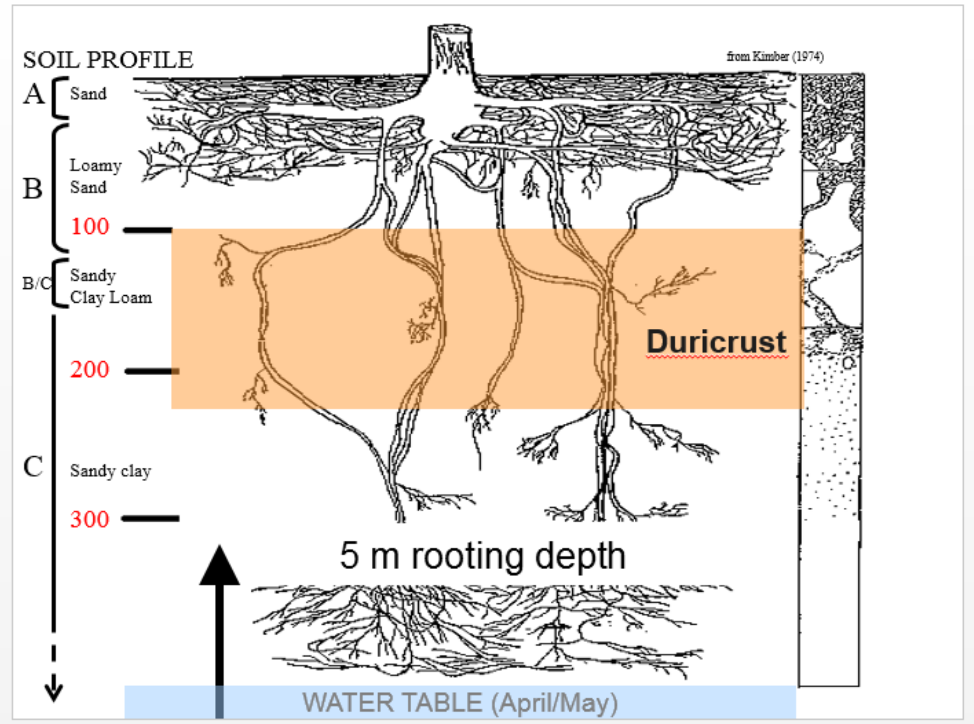


Figure 5-24 Rooting pattern of the savanna woodland trees in the Top-End (Source: Hutley 2008)

ERA

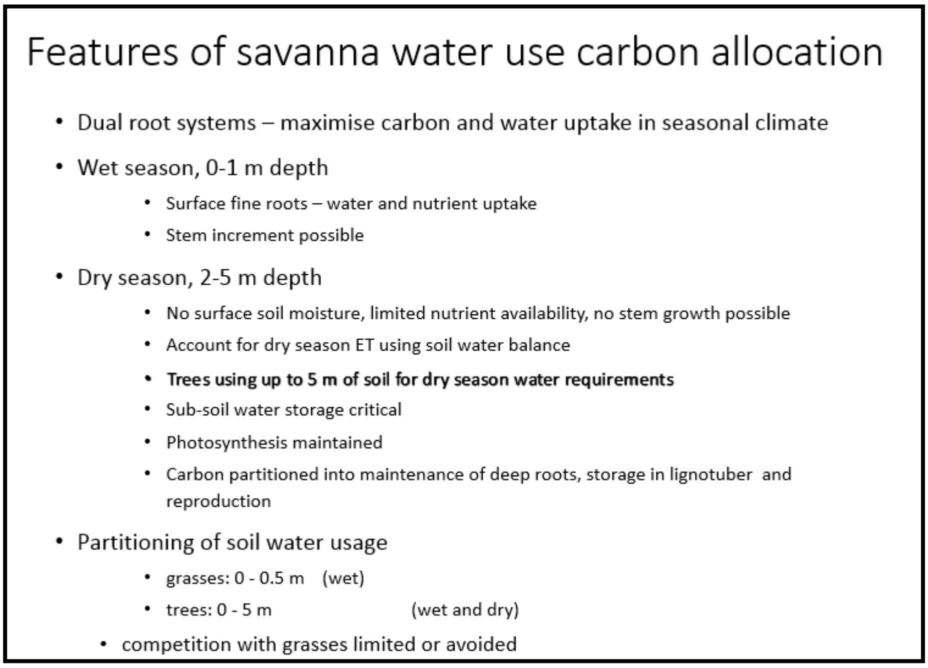


Figure 5-25 Key features of savanna vegetation water-use and carbon allocation strategies adapted to the Top-End seasonality ((Source: Hutley 2008)

In general, rates of plant growth and water demand decline as the wet season ends and the dry season progresses, and the fine root mass can be seen to diminish with the receding soil-water reserve (the cost to the plant of maintaining these fine roots during the dry season for little or no return is too great) (Janos *et al*. 2008). Any residual water demand must be met by the ability of plants to use deeper roots to access the remaining soil-water reserve.

Soil moisture extraction patterns at the Ranger's Georgetown Creek Reference Area (Site 21) demonstrate that soil water was extracted from 5.5 to 5.8 m below the surface in the late dry season (Refer to Section 4.3.3). More information with regards to waste rock studies on the TLF can be found in Appendix 5.1.

Canopy cover dynamics

Long-term canopy cover (as measured by Leaf Area Index, LAI) of the woodlands was monitored at the four ecohydrological study sites at the Georgetown Creek Reference Area and show significant seasonal variability (refer to Figure 5-26). The LAI is highest during the wet season and lowest during the dry season. The seasonal reduction is mostly about 50%, but is higher in some dry years (Note: LAI methodology details can be found Lu *et al* 2019).

Site 21 has the densest canopy (highest LAI) among the sites, and also the highest LAI seasonal variation. At Site 21 the LAI reduced by about 70% over the extended dry period leading into the late 2015-16 wet season. Whole-tree sap flow measurement demonstrated that Site 21 also has the highest annual transpiration (data not shown). Site 21 has a species composition (dominant overstorey species are Eucalyptus tetrodonta and Eucalyptus miniata) and basal area (8 m^2 ha^{-1}) similar to other tropical savanna in northern Australia (Hutley *et al*. 2000).

Trees will shed more leaves and earlier during the driest period in the dry season if water is beyond reach of the roots, as observed at the reference sites 21 and 30. Site 30 sits on a drier site, it sheds more leaves, earlier, and more rapidly than trees at Site 21, as reflected in the seasonal dynamics of the LAI (shown in Figure 5-27). That means, in the worst-case scenario, if there was less PAW than the target, trees will still be able to survive through the dry season and regrow during the wet season.

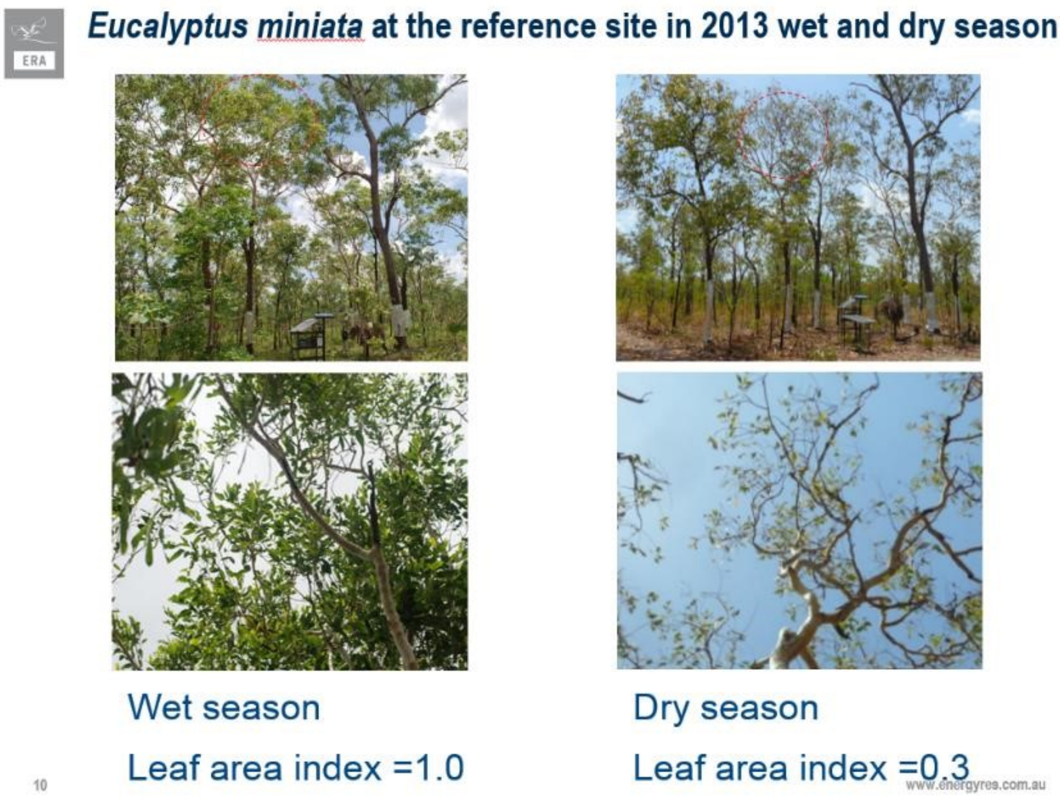


Figure 5-26 Seasonal change in leaf area index at the Georgetown Creek Reference Area (Source: Lu *et al*. 2018)

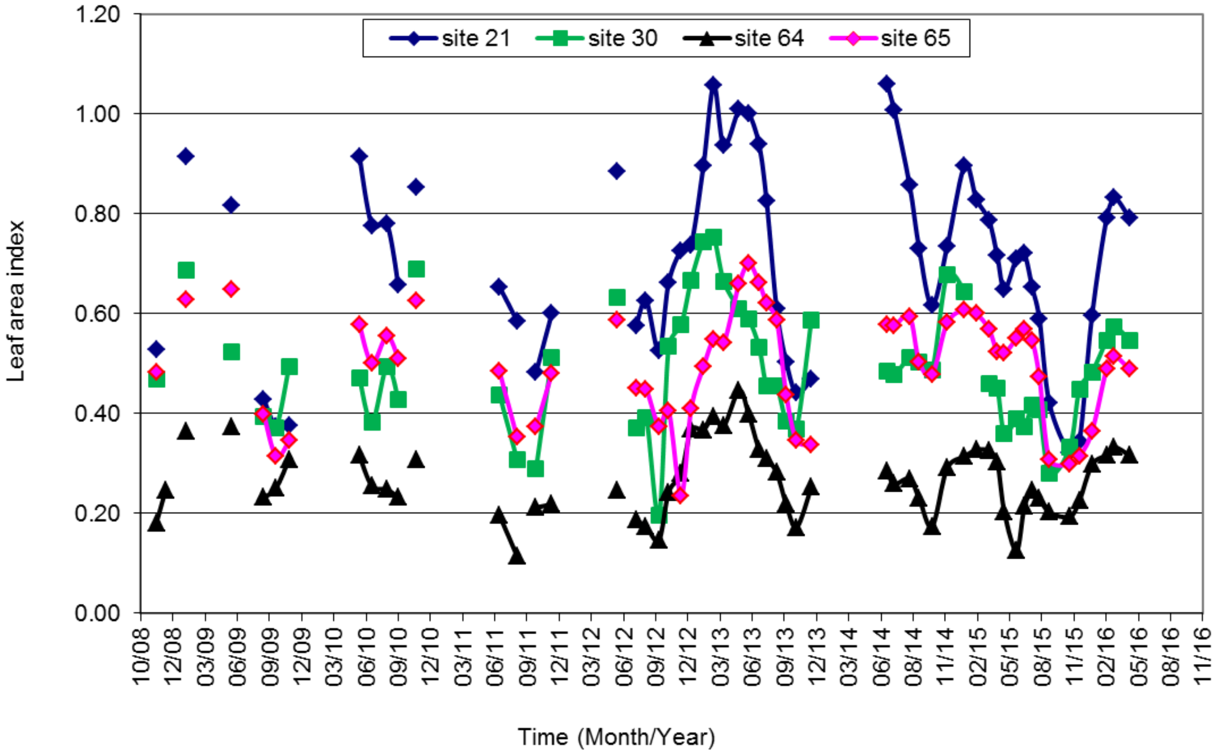


Figure 5-27 LAI dynamics at the four ecohydrological study sites

Total water requirements of the vegetation during dry season

Total water requirement for vegetation is usually measured by the evapotranspiration (ET), which in simple terms is the sum of over storey transpiration, under storey transpiration, and soil evaporation (Figure 5-28). Other closely related processes shown on Figure 5-28 are runoff and groundwater recharge.

In the Top End of the Northern Australia, during the dry season, the woodland vegetation water use is dominated by the overstorey and midstorey vegetation while the understorey dries off rapidly at the beginning of the dry season and its contribution to the ET is minimum and negligible compared to the tree/shrub water use (Hutley 2008, Hutley *et al*. 2000).

Stand transpiration, of the woodland near Ranger site was estimated based on tree stem xylem sap flow measurement at Site 21 of the Georgetown Creek Reference Area (Figure 5-29, refer to Lu *et al* 2019 for details on measurements of sap flow and stand transpiration). Tree water use is at its highest around the end of wet season and/or beginning of the dry season (April, May, June) when the soil water availability is high, the days are sunny, the air is dry (evaporative demand is high) and the LAI is high (refer to Figure 5-27). The transpiration decreases during the dry season as the soil dries up and LAI decreases (Figure 5-27), reaching a minimum at the end of the dry season right before a significant rainfall. During the early wet-season the transpiration increases as the soil water availability and canopy LAI increase, but the transpiration is not at its highest due to wet and raining days.

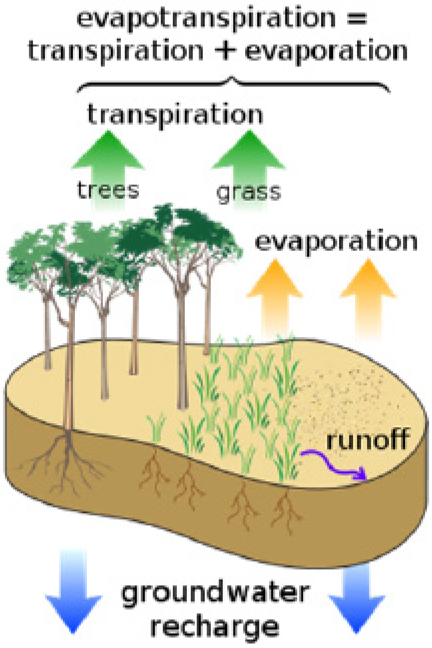


Figure 5-28 Evapotranspiration and its components

Figure 5-29 General view of an instrumented study site

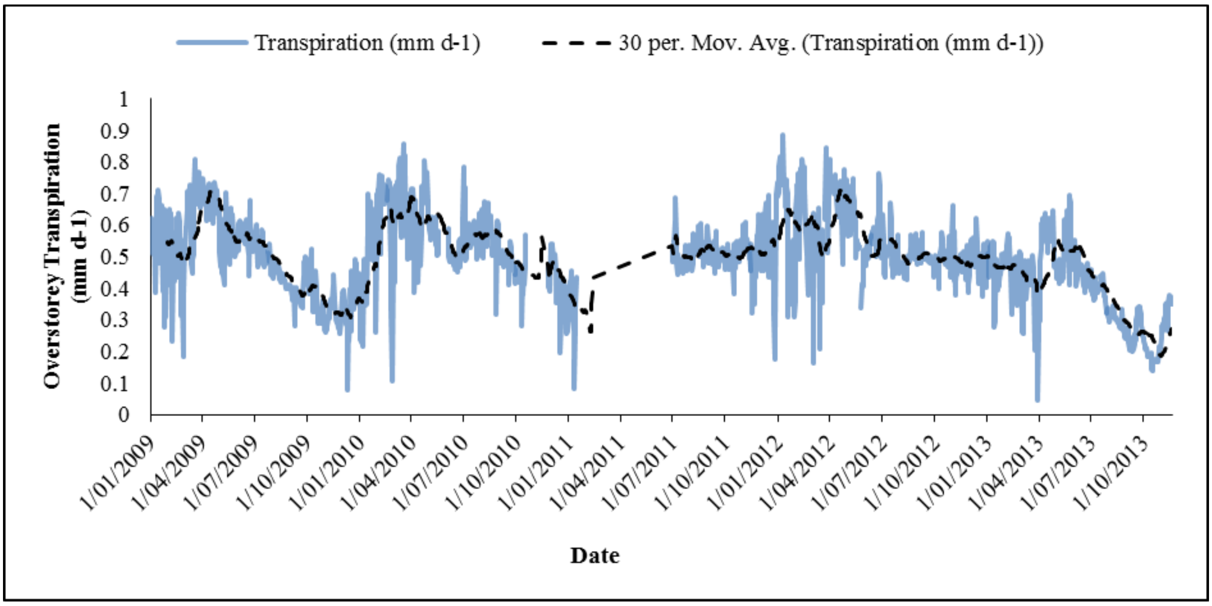


Figure 5-30 Annual dynamics of over storey tree transpiration at Site 21

Canopy cover (LAI) is directly and highly correlated with vegetation water use (Baumgartl *et al*. 2018). Site 21, with the highest LAI and therefore the highest vegetation water use, was selected as a reference site for modelling to compare dry season natural vegetation water requirement with the plant available water (PAW) supply in the final waste rock landform, because the site presents a conservative target for the vegetation water requirement (Baumgartl *et al*. 2018). To be on the more conservative side, an upper envelop of the average dry season transpiration of 0.5 mm day-1 was adopted for the WAVES modelling (refer to Appendix 5.1).

Groundwater table and soil water dynamics

At Site 21, the groundwater table level is very dynamic (Figure 5-31). During the wet season the water level reaches within 0.5 metres of the soil surface and during the dry season it drops below 10 metres below the soil surface. Note that the bore hole depth is slightly deeper than 10 m and the cable length of the hydrostatic pressure transducer was set to 10 m, so when the water level drops below 10 m, the transducer (logged) gives a maximal 10 m depth, but the manual dipper can still give the reading until the bottom of the borehole is dry. Groundwater and soil moisture measurement details can be found in Lu *et al* 2019).

This shallow groundwater system is also very transient during the wet season, with peaks subsiding rapidly after heavy rainfall stops. All these characteristics are typical of a groundwater system of a low hill with porous material in the shallow ground.

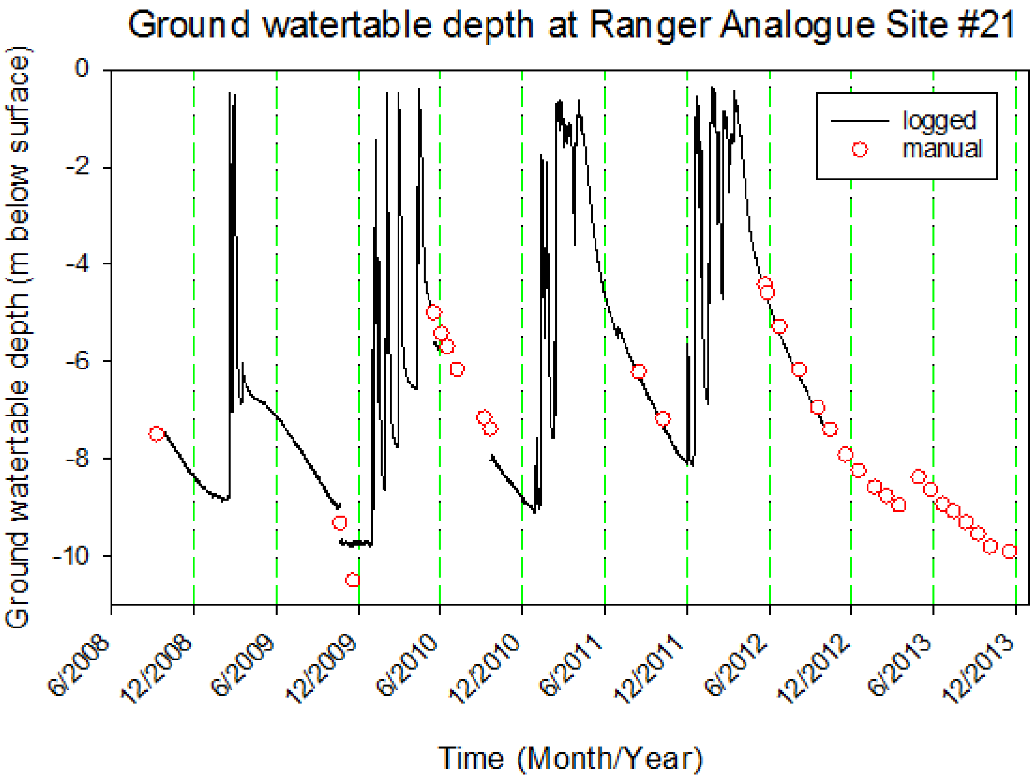


Figure 5-31 Temporal dynamics of the groundwater depth at Site 21

A comparison between the soil water dynamics (as shown by relative extractable water contents, REW) at different depths (0 to 5.5 metres below ground surface) and ground water table level (GWT) at Site 21 is shown on Figure 5-32. The data in Figure 5-32 clearly shows that maximum REW for the whole soil profile occurs during the late wet season. As the dry season progressed, soils dried quickly (within one month) near surface and in the shallow depths (at 0, 0.5 and 1 metres below ground surface). The 0-metre depth corresponds to a probe placed 0.05 metres below the ground surface (measuring soil water content from 0.05 to 0.35 metres below the ground surface). After the shallow soil dried, water was extracted from deeper levels, from 2 to 5.5 metres below ground surface progressively. By November 2012, extractable water from the whole 5.8-metre thick profile was nearly fully depleted (the deepest probe measures soil water from 5.5 to 5.8 metres below ground surface). However, the measurement of the sap flow clearly shows that the trees still maintained a substantial level of transpiration (Figure 5-30) during the same period which demonstrates that tree root systems exploited soil water from deeper soil.

The depth to the ground water table decreased progressively with, but ahead of, the rapid decrease of REW. The depth difference between the REW and the ground water table depth broadly corresponds to the capillary fringe height.

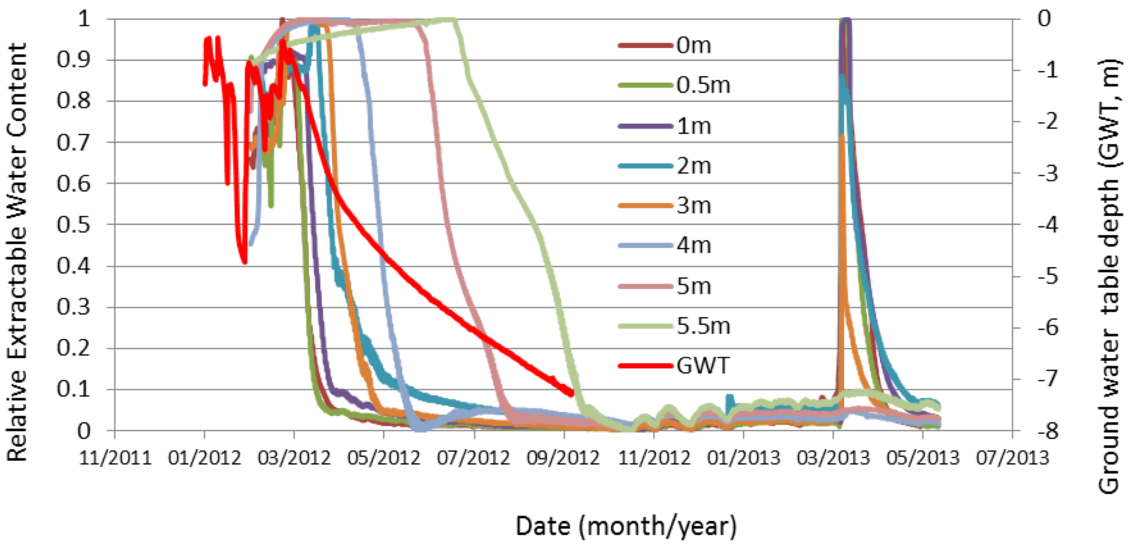


Figure 5-32 Relative extractable water contents measured at different depths and ground water table depth (GWT, in Red) at Site 21

Plant water uptake patterns can often be inferred from soil water depletion pattern (Knight 1999). From Figure 5-32 it is evident that as the dry season progressed, the extractable water was depleted progressively from the surface to deeper depths, reaching the depth of 5.5 to 5.8 m. This suggests that the natural savanna trees at the Ranger Georgetown Creek reference site are able to extract water at depth close to 6 metres below ground level. This is consistent with the finding of a study in Australia by Sharma *et al*. (1987) that a significant amount of soil water extraction under Eucalypt forests in Western Australia occurs to a depth of at least 6 m.

Soil evaporation and under storey transpiration are highly dependent on the shallow soil water content. Based on the soil moisture results shown in Figure 5-32 it is reasonable to expect that the evapotranspiration from the soil and understorey would decrease to near zero within a couple of months after the dry season starts. Therefore, the major component of the evapotranspiration during the dry season is over- and midstorey transpiration. This is consistent with other evapotranspiration studies in the Top End of the NT (Hutley 2008).

Despite that the dry season understorey ET and soil evaporation are negligible and were not directly measured at the Ranger reference site, they were simulated using a locally calibrated WAVES model to obtain the total dry season vegetation ET (Dawes *et al*. 1998, Zhang & Dawes 1998, Segura 2016).

Results of the total evapotranspiration (estimated stand transpiration of 0.5 mm.day^{-1} + simulated understorey ET and soil evaporation) of the reference site over the past 117 years are presented in Appendix 5.1 along with the PAW results for the waste rock landform.

5.3.3.6 Fauna species

Native fauna species

Kakadu NP contains over one third of Australia's bird species (271), one quarter of Australia's land mammals (77), 132 reptile species, 27 frog species and over 246 fish species recorded in tidal and freshwater areas (Director of National Parks 2016).

A number of conservation significant species (including a large number of mostly bird species listed under various migratory agreements) have been recorded on the RPA during previous surveys (Table 5-16). The identified species include the conservation listed Northern Quoll *Dasyurus hallucatus* (Endangered1; Critically Endangered2) and the Partridge Pigeon *Geophaps smithii smithii* (Vulnerable1; Vulnerable2) listed under the 1 EPBC Act and 2TPWC Act (Firth 2012).

A desktop review of flora and fauna data held by ERA included 26 reports presenting the results of fauna surveys; three reports documenting aquatic flora and fauna survey work; seven documents that reviewed previous terrestrial and aquatic flora and fauna work; and relevant data bases of ERA Birdwatch events that occurred on the RPA from 2001 – 2011, inclusive (Firth 2012).

Since the 1990s, a significant decline in the abundance of ten species of small mammals in Kakadu, including the Northern Brown Bandicoot (*Isoodon macrourus*), Fawn Antechinus (*Antechinus bellus*), Common Brushtail Possum (*Trichosurus vulpecula*), the TPWC Act listed Pale Field-Rat (*Rattus tunneyi*) (conservation status vulnerable) and the Northern Quoll (*Dasyurus hallucatus*) (conservation status Critically Endangered), has been recorded. The decline has been attributed to a high fire frequency, feral cats and cane toads (Woinarski *et al*. 2010).

The Northern Quoll population has undergone dramatic declines in the Top End of the NT as a result of ingesting the toxic cane toad (*Rhinella marina*), and in many areas of the mainland, such as Kakadu NP, has become almost extinct. It has not been detected in several recent surveys on the RPA, indicating it is likely extinct on the RPA. The only EPBC Act listed fauna species still known to occur on the RPA with any certainty are the Partridge Pigeon (*Geophaps smithii smithii*), Fawn Antechinus (*Antechinus bellus*) and Black-footed tree-rat (*Mesembriomys gouldii*), the latter two only being recently conservation listed.

During the last fauna survey undertaken on the RPA in September 2013, at least[6] 127 species were recorded, comprising eight native amphibian species, 79 bird species, at least 17 native mammal species, 20 reptile species and three introduced species. Seven EPBC Act or TPWC Act listed species were recorded within the 220 ha survey area, situated towards the east of Pit 3 in the Magela Creek and former Magela land application areas (LAA), and in the vicinity of RP1 (Eco Logical Australia 2014).

[6] There were several bat species whose calls could not be positively identified.

Table 5-16: Conservation listed species known to occur on the RPA (adapted from Firth 2012)

Common name	Scientific name	EPBC Act (CTH) status	TPWC Act (NT) status	Preferred habitat
MAMMALS				
Black-footed Tree-rat	*Mesembriomys gouldii*	Endangered	Vulnerable	Tropical woodlands and open forests in coastal areas
Brush-tailed Rabbit-rat	*Conilurus penicillatus*	Vulnerable	Endangered	Tropical woodlands; declined to near extinction since the 1980s
Fawn Antechinus	*Antechinus bellus*	Vulnerable	Endangered	Savanna woodland; tall open forest
Northern Brown Bandicoot	*Isoodon macrourus*	Not listed	Near threatened	Tall grassland, shrubland, savanna and open forest
Northern Quoll	*Dasyurus hallucatus*	Endangered	Critically Endangered	Eucalypt open forests; rocky areas
Pale Field-rat	*Rattus tunneyi*	Not listed	Vulnerable	Found in in the higher rainfall areas of the Top End of the Northern Territory
BIRDS				
Black-tailed Godwit[1-4]	*Limosa limosa*	Marine, migratory	Not listed	Coastal regions
Black-winged Stilt	*Himantopus himantopus*	Marine	Not listed	Freshwater and saltwater marshes, mudflats and the shallow edges of lakes and rivers
Broad-billed Sandpiper[1-4]	*Limicola falcinellus*	Migratory	Not listed	Sheltered coastal, intertidal mudflats
Caspian Tern[3]	*Hydroprogne caspia*	Migratory	Not listed	Coastal sheltered estuaries, inlets and bays
Cattle Egret	*Ardea ibis*	Marine	Not listed	Wet grasslands, wetlands, mudflats
Common Greenshank[1-4]	*Tringa nebularia*	Marine, migratory	Not listed	Coastal and inland wetlands
Common Sandpiper[1-4]	*Actitis hypoleucos*	Marine, migratory	Not listed	Coastal and inland wetlands, billabongs
Curlew Sandpiper[1-4]	*Calidris ferruginea*	Critically Endangered, marine, migratory	Vulnerable	Coastal areas, non-tidal swamps, lakes and lagoons, inland ephemeral and permanent lakes, dams

Common name	Scientific name	EPBC Act (CTH) status	TPWC Act (NT) status	Preferred habitat
Eastern Great Egret	*Ardea alba modesta*	Marine	Not listed	Range of wetlands, from lakes, rivers and swamps to estuaries, saltmarsh and intertidal mudflats
Glossy Ibis[1]	*Plegadis falcinellus*	Marine, migratory	Not listed	Swamps, flood waters
Great Egret	*Ardea alba*	Marine	Not listed	Wetlands, mudflats, mangroves
Greater Sand Plover[1-4]	*Charadrius leschenaultii*	Vulnerable, marine, migratory	Vulnerable	Sheltered beaches, intertidal mudflats or sandbanks, sandy estuarine lagoons
Green Pigmy Goose	*Nettapus pulchellus*	Marine	Not listed	Coast, tropical freshwater lagoons
Grey Plover[1-4]	*Pluvialis squatarola*	Marine, migratory	Not listed	Coast, inland wetlands
Grey-tailed Tattler[1-4]	*Tringa brevipes*	Marine, migratory	Not listed	Coastal intertidal pools, mudflats and rock ledges
Lesser Sand Plover[1-4]	*Charadrius mongolus*	Endangered, marine, migratory	Vulnerable	Intertidal sandflats and mudflats, beaches, estuary mudflats
Little Ringed Plover[2-4]	*Charadrius dubius*	Marine, migratory	Not listed	Lowland habitats with shallow standing freshwater
Long-toed Stint[1-4]	*Calidris subminuta*	Marine, migratory	Not listed	Shallow freshwater or brackish wetlands
Magpie goose	*Anseranas semipalmata*	Marine	Not listed	Coastal and inland wetlands, billabongs
Marsh Sandpiper/ Little Greenshank[1-4]	*Tringa stagnatilis*	Marine, migratory	Not listed	Coastal and inland wetlands, estuarine and mangrove mudflats
Pacific Golden Plover	*Pluvialis fulva*	Marine	Not listed	Wetlands, shores, paddocks, saltmarsh, coastal golf courses, estuaries and lagoons
Partridge Pigeon	*Geophaps smithii smithii*	Vulnerable	Vulnerable	Lowland woodland
Radjah Shelduck	*Tadorna radjah*	Marine	Not listed	Mangrove flats, swamps, freshwater swamps, lagoons, billabongs
Rainbow Bee-eater	*Merops ornatus*	Marine	Not listed	Open woodlands and forest, grasslands, widespread distribution and habitats

Common name	**Scientific name**	**EPBC Act (CTH) status**	**TPWC Act (NT) status**	**Preferred habitat**
Red-capped Plover	*Charadrius ruficapillus*	Marine	Not listed	Sandflats or mudflats at the margins of saline, brackish or freshwater wetlands
Red-necked Stint[1-4]	*Calidris ruficollis*	Marine, migratory	Not listed	Sheltered inlets, bays, lagoons, estuaries, intertidal mudflats and protected sandy or coralline shores
Ruddy Turnstone[1-4]	*Arenaria interpres*	Marine, migratory	Not listed	Coasts including mudflats
Sharp-tailed Sandpiper[1-4]	*Calidris acuminata*	Marine, migratory	Not listed	Fresh or saltwater wetlands
Swinhoe's Snipe[1-4]	*Gallinago megala*	Marine, migratory	Not listed	Coasts, floodplains, rivers
Terek Sandpiper[1-4]	*Xenus cinereus*	Marine, migratory	Not listed	Sheltered coastal mudflats, mangrove swamps
Wandering Whistling Duck	*Dendrocygna arcuata*	Marine	Not listed	Rivers, billabongs, pools and lakes
White-bellied Sea-eagle	*Haliaeetus leucogaster*	Marine	Not listed	Coasts, floodplains, rivers
Whimbrel[1-4]	*Numenius phaeopus*	Marine, migratory	Not listed	Primarily coastal distribution
Wood Sandpiper[1-4]	*Tringa glareola*	Marine, migratory	Not listed	Coasts, floodplains, rivers
REPTILES				
Estuarine Crocodile[1]	*Crocodylus porosus*	Marine, migratory	Not listed	Marine, freshwater
Merten's Water Monitor	*Varanus mertensi*	Not listed	Vulnerable	Creeks and billabongs
[1]Bonn; [2]China Australia Migratory Bird Agreement; [3]Japan Australia Migratory Bird Agreement; [4]Republic of Korea-Australia Migratory Bird Agreement				

Introduced fauna species

Eleven feral fauna species have been recorded in the RPA and an additional eight species have been recorded in Kakadu NP (Table 5-17). Three species recorded in both the RPA and Kakadu NP (pig, cat and cane toad) are listed under the EPBC Act as key threatening processes to environmental, natural heritage and cultural heritage values.

Table 5-17: Feral fauna species known to occur in Kakadu NP and the RPA

Type	Common name	Scientific name	RPA	Kakadu NP
Mammal	Dog	*Canis lupus familiaris*	Y	Y
Mammal	Buffalo	*Bubalus bubalis*	Y	Y
Mammal	Cattle	*Bos taurus*		Y
Mammal	Cat	*Felis catus*	Y	Y
Mammal	Donkey	*Equus asinus*		Y
Mammal	Horse	*Equus caballus*		Y
Mammal	Black rat	*Rattus rattus*	Y	Y
Mammal	House mouse	*Mus domesticus*	Y	Y
Mammal	Pig	*Sus scrofa*	Y	Y
Insect	Ginger ant	*Solenopsis geminata*		Y
Insect	Pharaoh's ant	*Monomorium pharaonis*		Y
Insect	Singapore ant	*Monomorium destructor*		Y
Insect	Ghost ant	*Tapinoma melanocephalum*		Y
Insect	Big-headed ant	*Pheidole megacephala*		Y
Insect	Cockroach	*Periplaneta spp.*	Y	Y
Insect	European honey bee	*Apis mellifera*	Y	Y
Amphibian	Cane toad	*Rhinella marina*	Y	Y
Reptile	Flower-pot snake	*Ramphotyphlops braminus*	Y	Y
Reptile	House gecko	*Hemidactylus frenatus*	Y	Y

5.3.4 Aquatic ecosystem

BMT WBM (2010) describe the ecological character of the Kakadu NP Ramsar site, which now includes the entire national park. According to BMT WBM (2010) the site contains five major landscape types, including two found on, adjacent to, or immediately downstream of, the RPA, ie Lowlands containing open woodlands and creeks, and Floodplains containing freshwater wetlands, creeks and billabongs.

The terrestrial flora and fauna of Kakadu NP descriptions provided above (section 5.3.3) discuss important water birds and semi-aquatic species (eg amphibians and reptiles).

On the RPA there are no listed or endangered macroinvertebrate or fish species, or aquatic fauna species, or any considered rare or restricted in distribution. Nor are there environments of special significance (such as significant breeding sites, seasonal habitats or wetlands areas). As discussed in section 5.3.3 several migratory bird species listed of international importance and the vulnerable Merten's water monitor have been recorded on the RPA.

5.3.4.1 Vegetation types

The lowland riparian and rainforest vegetation type, which represents denser vegetation of the lowlands, typically associated with streams, creeks and billabongs is described in section 5.3.3. This habitat type is represented throughout the Kakadu NP Ramsar site with about 1% occurring within the RPA.

There has been multiple reports of floodplain vegetation on the Magela Floodplain with varying numbers of classes being identified which suggest a high level of variability over time. Rainfall volumes and patterns affect inundation periods, water level, and soil moisture which along with fire affects community distributions seasonally and inter-annually (Whiteside and Bartolo 2014). Using remote sensing and a review of past reports, Whiteside and Bartolo (2014) identified twelve classes of typical vegetation on the Magela floodplain occurring in May 2010 (Table 5-18). Time-series mapping by the SSB will build on this dataset and classification providing further information on vegetation dynamics on the floodplain.

Table 5-18 Twelve classes of Magela floodplain vegetation described by Whiteside and Bartolo (2014)

Class name	Composition and occurrence	Area of cover on the floodplains in May 2010
Melaleuca woodland	Typically contains *M. cajaputi* and *M. viridiflora* in the northern regions and at the edges of the floodplain, and *M. leucadendra* in the backswamps that are inundated for most of the year. Open forest communities are typically inundated for 5–8 months of the year. This land cover was mostly located in the southern reaches of the floodplain and around the perimeter.	10–50% woody cover; covering 5039 ha
Melaleuca open forest		open forest communities have 50–70% cover; covering 821.8 ha
Oryza grassland	Dominated by the annual grass, *Oryza meridionalis* towards the end of the Wet season. In the Dry season there is mostly bare ground or dead *Oryza*.	4040 ha
Hymenachne grassland	Dominated by *Hymenachne acutigluma* throughout the year. Other species that may occur include *Oryza meridionalis*, *Nymphaea spp.*, and *Pseudoraphis spinescens.*	3639 ha

Class name	Composition and occurrence	Area of cover on the floodplains in May 2010
Para grass	The weed grass, *Urochloa mutica* (Para grass), is an introduced invasive species. It forms dense monocultures and can outcompete native vegetation in communities of *Hymenachne*, *Oryza* and *Eleocharis*. The community cover on the floodplain was mostly in the central plains region.	2181 ha
Eleocharis	Dominated by the sedge, *Eleocharis dulcis* with larger areas mostly occupying the northern areas of the floodplain.	1054 ha
Leersia grassland	Floating mats of *Leersia hexandra.* Larger mats can be found on the western border of Red Lily Swamp.	967 ha
Pseudoraphis	Dominated by the perennial grass, *Pseudoraphis spinescens*. Particularly in the southern half of the floodplain.	943 ha
Pseudoraphis/ *Hymenachne* grassland	Co-dominated by *Pseudoraphis spinescens* and *Hymenachne acutigluma*.	375 ha
Mangrove	Mangrove community is located mostly bordering the Magela Creek as it enters the East Alligator River. (Species not described).	249 ha
Nelumbo herbland	This community is dominated by the water lilies, *Nelumbo nucifera* or to a lesser extent *Nymphoides* spp. These communities occur in permanent and semi-permanent wet areas. Other species that may be present include *Leersia hexandra*, *Hymenachne acutigluma*, *Nymphaea* spp. The largest community is found on the eastern extents of Red Lily Swamp (the open body of water in the western part of the floodplain).	243.3 ha
Salvinia	Dominated by the floating fern, *Salvinia molesta*. This declared Class-B weed can completely cover small areas of open water that are protected from wind. On larger	107.5 ha

Class name	Composition and occurrence	Area of cover on the floodplains in May 2010
	stretches of open water, the fern can be found on the leeward edge.	

BMT (2019) describe the patterns, components, key species and primary productivity of the aquatic ecosystems, of the RPA and surrounds as follows.

5.3.4.2 Aquatic ecosystem patterns

The aquatic ecosystems of the RPA and surrounds are highly dynamic, with seasonal rainfall patterns being a major driver of temporal variability. While fine scale temporal patterns (timing, duration, frequency) and magnitude of rainfall events may vary from year to year, seasonal patterns in the physio-chemical and biological character of waters broadly follow predicable flood-drought cycles.

The wet season is characterised by large increases in aquatic habitat extent, and lateral and longitudinal connectivity, as floodwaters fill lotic and lentic waterbodies and inundate floodplains (Ward *et al.* 2016; Bunn *et al.* 2015). This leads to an explosion of aquatic ecosystem productivity. Most aquatic species have peak reproduction, recruitment and biomass during the wet season (e.g. Bishop *et al.* 2001; Douglas *et al.* 2005, Wharfe *et al.* 2011). Flows are also key drivers of physical (geomorphological) and biological processes that control the structure of aquatic habitats.

Surface water flows cease during the dry season, and aquatic ecosystems are comprised of isolated billabongs on the floodplain and in channels, and sub-surface groundwater-dependent ecosystems (GDE) in channels. Although in wetter years, substantial floodplain areas of the Magela Creek catchment can remain inundated into the dry season (Bunn *et al.* 2015).

Shallow billabongs experience a decline in water levels and water quality, leading to local population crashes, or in the case of semi-aquatic species such as crocodiles, dispersal elsewhere. The dry season retraction in habitat and food resource availability reduces overall aquatic ecosystem biomass, and top-down biological interactions (predation, competition) become increasing important ecosystem controls. Water quality deterioration can lead to significant ecosystem stress, especially in shallow waterbodies (Wharfe *et al.* 2011). Shallow lowland billabongs do not represent important refugia because of their shallow nature and associated dry-season habitat and water-quality deterioration, (Humphrey *et al.* 2016). Furthermore, wet seasons of low rainfall, in conjunction with an extended dry season, can lead to many shallow lowland billabongs completely drying out (Humphrey *et al.* 2016). Similarly, creek channels and seasonally inundated floodplain environments also completely dry out during the dry season, and do not provide refugia functions.

Deep permanent billabongs (such as Mudjinberri Billabong) generally have good water quality year-round. They represent important dry season refugia, providing a source for subsequent population replenishment during the wet season.

5.3.4.3 Aquatic ecosystem components

Biodiversity values, and associated cultural values, are comprised of a variety of ecological components at different hierarchical levels (i.e. species, assemblages, habitats/vegetation types, ecosystems). BMT WBM (2010) list a number of critical and supporting ecosystem components of the Kakadu NP Ramsar site. That work and Garde (2015) describing culturally important species was reviewed to identify key species and groups which are indicators of Ramsar listed and cultural values (BMT 2019).

The key species and groups and their presence in relation to the RPA are listed in Table 5-19.

Table 5-19 List of key species indicators of Ramsar and cultural values in relation to the RPA (BMT 2019)

Category	Species, Conservation Listing and or cultural value	Presence on the RPA or downstream aquatic environment	Species Group
Threatened species	Yellow chat (Alligator Rivers) - *Epthianura crocea tunneyi* (EPBC Endangered)	Possible – occurs in palustrine wetlands and saltmarsh	Water birds
	Pig-nosed turtle - *Carettochelys insculpta* (IUCN Vulnerable)	Not present – not recorded in catchment	Reptiles
Locally endemic species	Kakaducarididae shrimps (*Leptopalaemon* and *Kakaducaris*) (Bruce 1993, Page *et al.* 2008). Endemic genus of isopod (*Eophreatoicus*) (Wilson *et al.* 2009). Seven of the nine *Leptophlebiidae* species (prong-gilled mayflies) in Kakadu are endemic to the Timor Sea Drainage Division (Finlayson *et al.* 2006).	Not present. Restricted to stone country	Macro-invertebrate s
Species with large proportion of geographic range in Kakadu	See locally endemic species	Not present. Restricted to stone country	
	Exquisite rainbowfish *Melanotaenia exquisite*	Not present.	Fish
	Magela hardyhead *Craterocephalus marianae* Sharp-nosed grunter *Syncomistes butleri* Midgley's grunter *Pingalla midgleyi*	Present. Stone country and lowland areas	Fish
	Woodworker Frog *Limnodynastes lignarius*	Not present – restricted to stone country	Frogs
Species identified as having	Significant breeding aggregations of magpie geese *Anseranas semipalmata* and comb-crested Jacana *Irediparra gallinacea*	Present – billabongs and floodplain	Water Birds

Category	Species, Conservation Listing and or cultural value	Presence on the RPA or downstream aquatic environment	Species Group
important populations in Kakadu based on Ramsar	Resident water birds with >1% population criterion in Kakadu: Wandering whistling-duck *Dendrocygna arcuate,* Plumed whistling-duck *Dendrocygna eytoni,* Radjah shelduck *Tadorna radjah,* Pacific black duck *Anas superciliosa,* Grey teal *Anas gracilis,* Brolga *Grus rubicunda,* Black-necked stork *Ephippiorhynchus asiaticus*	Present – billabongs and floodplain	Water Birds
	Migratory shorebird species with >1% of the East Asian – Australasian Flyway population size in Kakadu (Bamford *et al.* 2008):: Marsh sandpiper *Tringa stagnatilis*, Little curlew *Numenius minutus*, Common sandpiper *Actitis hypoleucos*, Australian pratincole *Stiltia Isabella*, Sharp-tailed sandpiper *Calidris acuminata*	Present – billabongs and floodplain (mostly coastal)	Water Birds
Species of notable cultural significance and values	*Acacia holosericea*[7], *Pandanus spp.*, *Melaleuca spp.*, *Barringtonia acutangula* – resource	Present – billabongs and floodplain	Riparian and Floodplain Trees
	Water lily *Nymphaea* spp. fruit and seeds – food Aquatic macrophyte tubers – *Amorphophallus paeoniifolius, Aponogeton elongatus, Dioscorea bulbifera, Dioscorea transversa, Eleocharis dulcis, Eleocharis spp., Nelumbo nucifera, Nymphaea macrosperma, Nymphaea pubescens, Nymphaea violacea, Triglochin procerum* - food	Some species present – billabongs and floodplain	Macrophytes
	Mussels and freshwater prawns – food	Present – billabongs and floodplain	Aquatic Invertebrates
	Barramundi *Lates calcarifer* , Salmon catfish *Sciades leptaspis,* Black bream *Hephaestus fuliginosus*, Saratoga *Scleropages jardinii* – food	Present – billabongs and floodplain	Fish
	File snake *Acrochordus arafurae*, Water python *Liasis fuscus*, Crocodiles *Crocodylus porosus* and *C. johnstoni* eggs, Monitors *Varanus* spp., Turtles - *Chelodina oblonga* and *Elseya dentata* – food.	Present – billabongs and floodplain	Reptiles

[7] Although this species is common on site due to use in early revegetation trials at the site, it is considered a native invasive in Magela Creek Catchment.

Category	Species, Conservation Listing and or cultural value	Presence on the RPA or downstream aquatic environment	Species Group
	See also *Carettochelys insculpta* above		
	Magpie goose *Anseranas semipalmata* – food (meat/eggs)	Present – billabongs and floodplain	Water Birds

The movement patterns and reproductive/recolonisation processes of several of the key species groups listed in Table 5-19 are summarised (below) by BMT (2019).

5.3.4.4 Aquatic invertebrates

Marchant (1982) describes patterns in the richness and abundance of aquatic macroinvertebrates in billabongs of the Magela Creek catchment. In shallow billabongs, the on-set of the wet season saw rapid increase in richness and abundance of invertebrates. The rapid resurgence of fauna early in the wet season suggests very fast growth and/or reproductive/recruitment rates. Both richness and abundance peaked in the late wet/early dry, which was two (richness) to five (abundance) times greater than recorded during the end of the dry season.

There were seasonal differences in composition in shallow billabongs, with high densities of Ephemeroptera, Trichoptera, Mollusca, Hemiptera and Chironomidae during the wet season, and Coleoptera (especially Dytiscidae), Tanypodinae chironomids, Ceratopogonidae, some Hemiptera and Gastropoda, and Macrobrachium prawn numerically dominant in the dry season. Many less common taxa occurred in variable abundance throughout the year. Marchant (1982) speculated that these changes were related to seasonal changes in aquatic macrophyte abundance, an important habitat for many aquatic invertebrates.

By contrast, deep channel billabongs did not show such strong seasonal variability, and maximal richness and abundance values were similar to that in shallow billabongs. Despite differences in habitat structure and wetting-drying cycles, fauna composition was largely similar between shallow and deep billabongs.

Marchant (1982) suggested that short life-cycles (measured in weeks to months rather than 10s of months) and very fast rates of larval growth likely prevail in most invertebrate groups in the Magela catchment billabongs. These are necessary adaptations for organisms living in ephemeral environments subject to seasonal wetting and drying cycles (Williams 1987).

The seasonal patterns described by Marchant (1982) are summarised in Table 5-20

Table 5-20 Seasonal patterns in aquatic macroinvertebrates in Magela catchment billabongs (BMT 2019 after Marchant 1982)

Taxa	Pattern
Gastropoda	Peak abundance of the common species in wet season Hibernate during dry season Planktonic larvae
Ostracoda and Conchostraca	Peak early to mid-wet
Atyidae and Palaemonidae	Atyidae - Dry season peak abundance and breeding (shallow), common year-round in deep billabongs Palaemonidae – dry season peak, absent early wet, breeds in estuary
Ephemeroptera	Peak in late wet/early dry in shallow. Emergence and reproduction continuous for many species
Odonata	Peak abundance in late wet/early dry for most species, but some species only found in early wet and late dry. Breeding peak in wet season for most species only found in early wet and late dry.
Hemiptera	Peak abundance in late wet/early dry for most species, but some uncommon species
Neuroptera	Wet season only, in association with sponges
Diptera	Emergence and breeding of Chironomids appeared to occur continuously while large numbers of larvae were present. Tanypodinae more abundant in dry season Ceratodontidae were more abundant in dry season, disappearing in early wet season
Lepidoptera	Most species only present in wet season, and in low numbers
Trichoptera	Peak abundance typically in early dry, but many species recorded throughout the year
Coleoptera	Adult Dytiscidae peak at the end of dry season, larvae mostly in wet season Except for the Hydrophilidae in the shallow billabongs, breeding of all families appeared to occur during the wet season

5.3.4.5 Fish

Bishop *et al.* (2001) examined the autecology of fish species in the Magela Creek system. Most fish species in the catchment undertake broad-scale movements for reproductive and feeding purposes. Many fish species disperse into lowlands and floodplains during the wet season for feeding and breeding purposes, resulting in high fish productivity during this period.

As water levels decline, fish move from seasonally inundated floodplain and sandy channel environments into dry-season refuges. These refuges include permanent billabongs, and in the case of euryhaline[8] species such as barramundi, estuarine river channel environments.

[8] Species able to tolerate a wide range of salinity.

Sandy creek channels represent important fauna movement corridors during the recessional stage (i.e. late wet/early dry transition). Smaller fish move upstream along the slow-flowing edges of creeks, which was suggested to be due to lower water velocities on the edges of the creek, or as an evolutionary mechanism to avoid larger predators residing in deeper sections of creek channels (Bishop and Walden 1990).

From a reproductive ecology perspective, most species breed around the on-set of the wet, coincident with flooding and associated increase in habitat availability, nutrients and algae production, and food availability (Bishop *et al.* 2001). A small number of spawners can breed at any time of the year, but most of these species typically have a wet season peak.

Within the Magela Creek catchment the most important spawning habitat for most species were the lowland backflow billabongs, and several species breed exclusively in this habitat type (Bishop *et al.* 2001). The escarpment area and sandy creek bed habitats were also commonly used spawning sites for numerous species, but only a small number breed exclusively in these habitat types (including *Neoarius erebi*, *Leiopotherapon unicolor*, *Neosilurus hyrtlii* and *Porochilus rendahli*). A small number of species are catadromous (migrate to sea to breed). Notwithstanding this, most catadromous species are large-bodied species that can be a dominant component of the fauna biomass, as many are important from a fisheries and cultural heritage perspectives – e.g. barramundi, tarpon, eels.

5.3.4.6 Bird/Reptiles/Amphibians

Most bird species in the catchment undertake broad-scale movements for feeding and breeding purposes. During the dry season, water birds are very abundant and diverse (Morton *et al.* 1991). Water birds prefer habitat with varying water depths, however towards the end of the dry season with receding water levels, water birds congregate in high abundances wherever water remains. These areas include the upper floodplain, the western part of the plain and channels through the Melaleuca swamps in the central plain). As flooding of the floodplain increases during the wet season, water birds fly away to other areas and become less abundant (Morton *et al.* 1991).

Migratory birds migrate to the catchment prior to and just after the wettest months (January–March). The most common migratory water bird species include the little curlew (*Numenius minutus*), oriental plover (*Charadrius veredus*), large sand plover (*C. leschenaultii*) and the Mongolian plover (*C. mongolus*) (Morton *et al.* 1991).

There are few water bird species that breed in significant numbers within the Magela Creek system, however, the Comb-crested Jacana (*Irediparra gallinacea*) breeds in abundance (Press *et al.* 1995). The main breeding period of the Comb-crested Jacana is during the late wet season, between the beginning of March to April.

Most reptiles are abundant during the wet season, while in the dry season they are concentrated to remnant waterbodies, such as billabongs (Gardner *et al.* 2002). Some species, such as freshwater turtles, bury themselves in mud as the water dries up during the end of the dry season.

Most frog species breed at the onset of the wet season before the floodplain is completely inundated (Tyler and Crook, 1987). During the dry season, most frog species are totally inactive, with some species burrowing underground, while others are restricted to billabongs.

5.3.4.7 Trophic processes and ecosystem productivity

Based on data in Adame *et al.* (2017), macrophytes represented the dominant primary producers in the freshwater reaches of the Kakadu wetlands (1870 - 2892 mg C/m^2/day) during the wet season, followed by terrestrial inputs (e.g. 970 mg C/m^2/day for Melaleuca litterfall; Finlayson *et al.* 1993), phytoplankton (122-334 mg C/m^2/day) and periphyton attached to macrophytes (13-219 mg C/m^2/day). This agrees with estimates of the relative contribution of primary producer groups in other tropical floodplains (Adame *et al.* 2017). The deeper floodplain backswamp areas had the highest periphyton and macroalgae productivity; these areas also hold water the longest, remaining productive into the dry season (Bunn *et al.* 2015).

Adame *et al.* (2017) found that while primary production in Kakadu wetlands was high compared to many other ecosystems, the wetlands were heterotrophic. This reflects the high inputs of organic matter to the system, such as dead macrophytes, fish carcases and other organic matter during the dry season (Adame *et al.* 2017). The decomposition of organic matter during the following flooding season can results in anoxia in places (Adame *et al.* 2017).

While macrophytes are highly productive, isotope analysis indicates that algae (periphyton and phytoplankton) can be the dominant internal source of carbon to aquatic fauna in the wet-dry tropics (Douglas *et al.* 2005). Douglas *et al*. (2005) suggested that much of the biomass of macrophytes may enter a detrital pool with a microbial 'dead-end' for aquatic ecosystems. Macrophytes do represent important habitats for the periphyton assemblages that sustain aquatic ecosystems (Bunn *et al*. 2015; Adame *et al*. 2017), and are important to the diets of some semi-aquatic and terrestrial fauna (Douglas *et al*. 2005), especially water birds (e.g. magpie goose; Frith and Davies 1966).

Isotope analysis by Bunn *et al*. (2015) in the ARR found that while insects, crustaceans and small fish can be sustained by 'internal' producers from the within the waterhole, external food sources from outside the home waterhole are critical to larger animals such as saratoga, barramundi and crocodiles. External sources can include marine fish and invertebrates (e.g. crabs, prawns, molluscs), small floodplain-associated freshwater fishes, and, in the case of the crocodiles, land mammals such as wallabies and pigs. Bunn *et al*. (2015) concluded that "the greater importance of external sources with increasing body size is a common feature of Kakadu food webs".

Figure 5-33 depicts a foodweb for aquatic ecosystems in the Magela Creek catchment. Diet data of fishes from Magela Creek, and tropical rivers in northern Australia more broadly, show little evidence of dietary specialization. For example, Bishop and Forbes (1991) found that fish assemblages in Magela Creek were largely omnivorous (20-50%, depending on habitat). Because many fish and many other aquatic vertebrates feed on a broad range of items, food webs are short, diffuse, and highly inter-connected (Douglas *et al*. 2005).

Douglas *et al*. (2005) notes that a key characteristic of aquatic foodwebs in the Australian wet-dry tropics is that a "few large bodied consumers control the flows of energy and matter into and through the animal community. Strong top-down control by such macroconsumers is emerging as a characteristic feature of tropical streams and rivers with fish and shrimp capable of exerting a disproportionately large influence on benthic sediments, detritus, nutrient demand and algae and invertebrate communities". Predation by birds and fish is a key top-down control on aquatic productivity at low water levels. High mortality rates can occur in refuge areas due to reduced resources and high rates of predation. During the wet season, bottom-up processes are thought to be more important.

ERA

Combined seasons

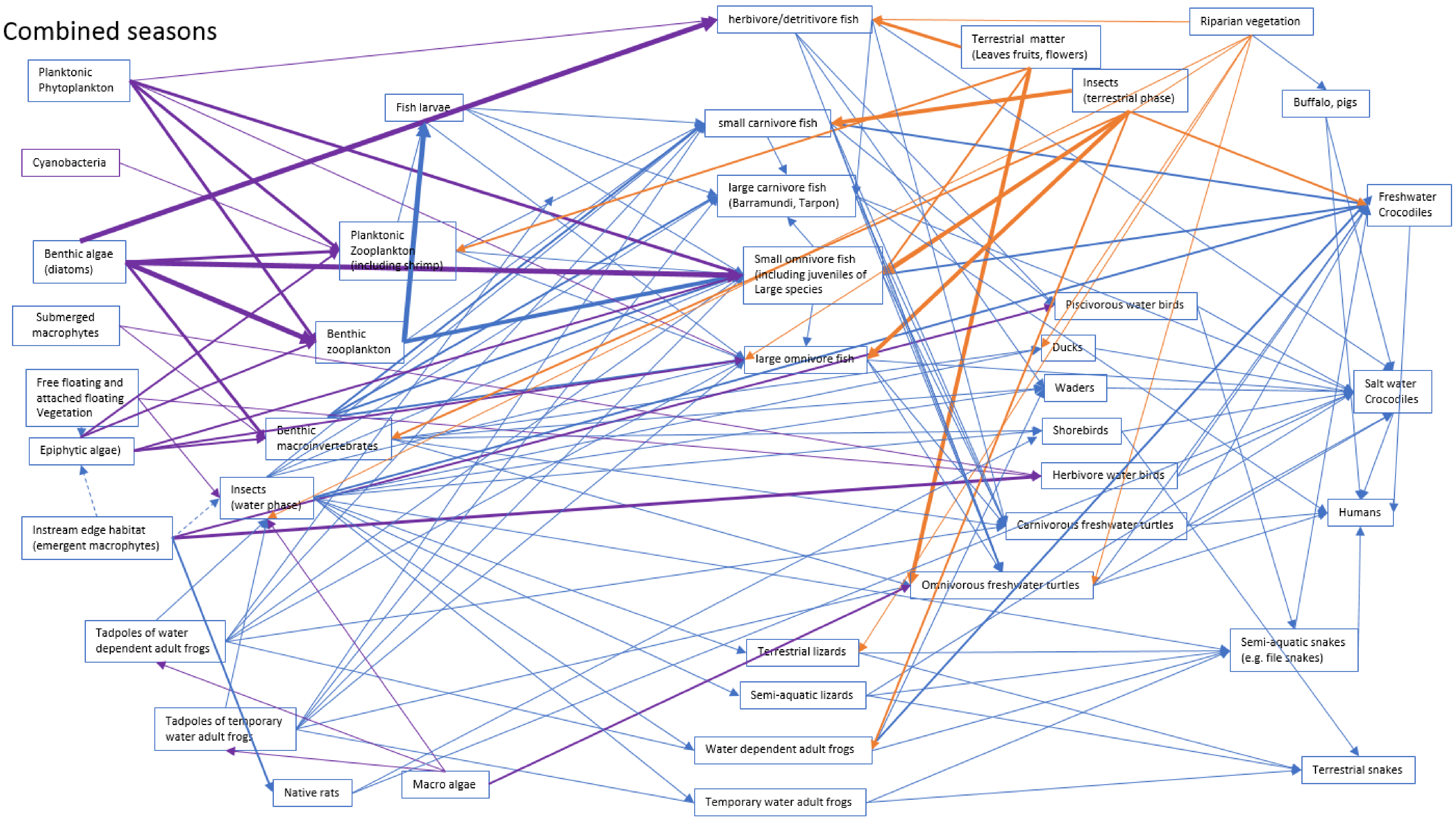


Figure 5-33 Food web for aquatic ecosystems in the Magela Creek catchment (from BMT 2019)

Notes: there are differences between seasons. In dry seasons the system is more closed. Wet seasons system is open and connected. Most organisms are omnivorous feeding on a range of different items. This is important and makes them less susceptible to small changes to food species

5.3.5 Trial Landform

5.3.5.1 Radon exhalation

The TLF has provided a unique setting to investigate seasonal and long-term changes in radon exhalation, soil activity concentration and terrestrial gamma dose rate for the four surface and revegetation treatments, and dependency on cover type, weathering and compaction effects and developing vegetation. Radon exhalation from the four erosion plots (i.e. EP1, EP2, EP3 and EP4) has been measured over several years to investigate whether there were any temporal changes of radon exhalation, taking into account rainfall, weathering of the rock, erosion and compaction effects, and the effect of developing vegetation on the landform (Bollhöfer & Doering 2013).

Although average soil radioactivity was not markedly different across the four erosion plots (Figure 5-34), there was a difference in average radon flux densities for the two different surface treatments (waste rock and waste rock blended with lateritic material). In the dry season, typical average radon flux densities from the surface of the waste rock – laterite treatment were higher than radon flux densities from waste rock only, and decreased markedly in the wet (Bollhöfer & Doering, 2016). In contrast, there was no obvious seasonal trend observed for radon exhalation fluxes from waste rock only until years four and five after construction (Bollhöfer & Doering, 2016).

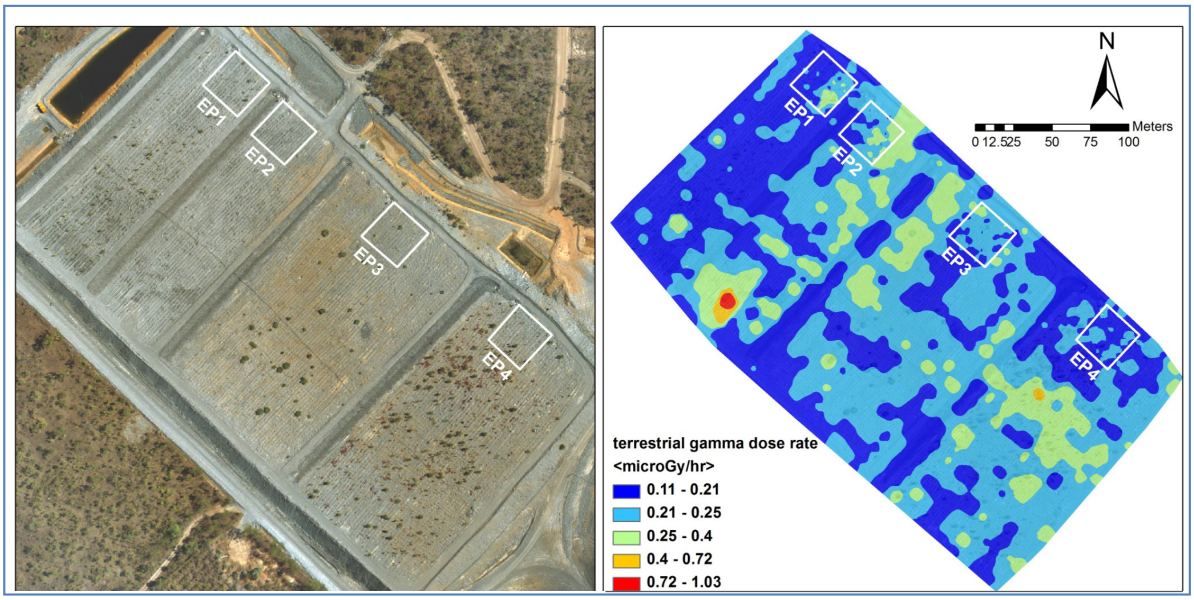


Figure 5-34: Trial landform and contour plot of the terrestrial gamma dose rates measured across the trial landform in June 2012 (Bollhöfer & Doering 2013; p 136)

Radon exhalation measurements recommenced in the second quarter of 2019 to confirm whether the dry season Radon exhalation flux densities have increased since 2014 (McMaster, 2020). Preliminary results indicate a stabilised Radon-222 exhalation flux density with no further increases in radon-222 exhalation (McMaster 2020).

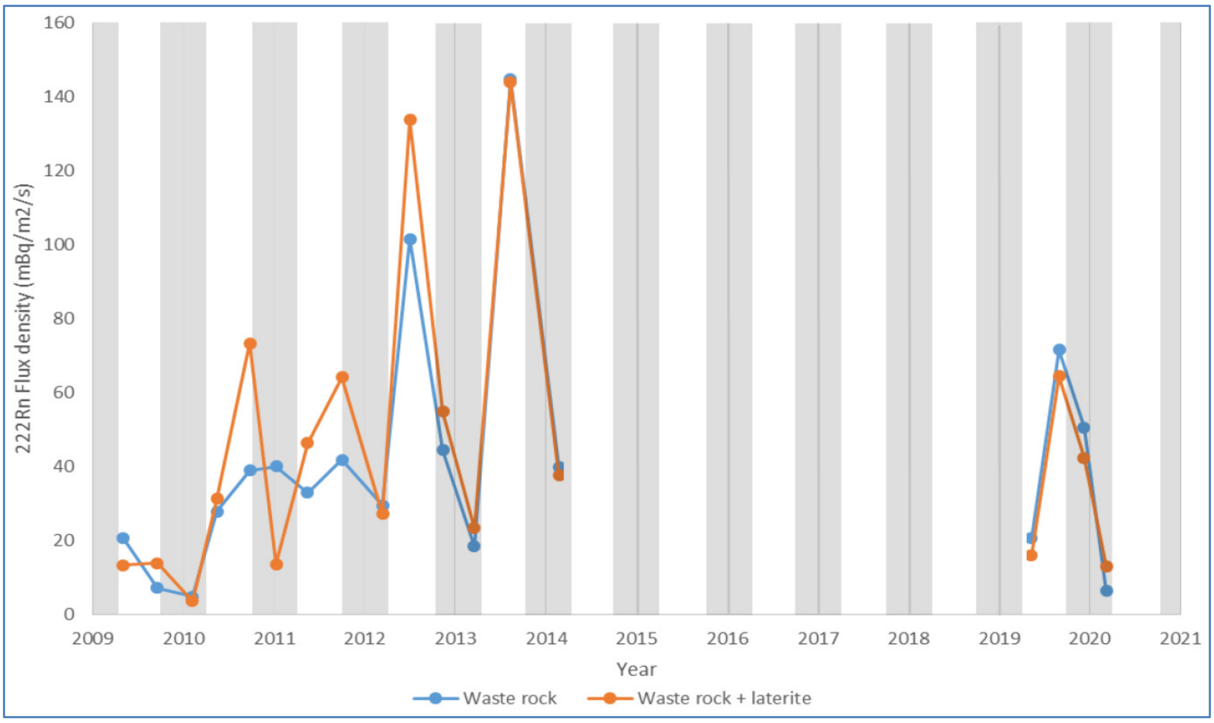


Figure 5-35 Geometric mean radon-222 exhalation flux at the TLF measured since 2009, grey regions indicate the wet season. (McMaster 2020)

Refer to Appendix 5.1 for other studies completed on the Trial Landform.

5.4 Technical knowledge base

The Ranger Mine has been the subject of extensive studies and monitoring programs over the past 38 years. The outcomes of these studies have been presented through various community and stakeholder consultation processes (e.g. ERA 2014b, Iles 2011, Johnston & Milnes 2007, McGovern 2006, Supervising Scientist 2016a) and in statutory reports such as the annual environment reports, mining management plans, Ranger Mine annual wet season reports and groundwater reports. The studies serve to:

- inform the overarching closure strategy and approach
- inform the development of closure criteria (Section 8)
- establish best practicable technology (BPT) and as low as reasonably achievable (ALARA) approaches and strategies for closure implementation that ensure the best environmental and achievable closure outcome for the Ranger minesite that attains compliance with ER requirements (Section 6)
- identify and rank closure risks to ensure the ongoing management of potentially high risks and an iterative approach to mine closure risk assessment (Section 7)
- inform the construction of a final landform (Section 9)
- provide baseline data against which to measure closure performance (Section 10)

- identify knowledge gaps and/or alternative options to past elements of the closure strategy thus ensuring that the most current and practical approaches to closure activities are implemented.

It is recognised that some projects have been finalised whilst others are ongoing. Further updates of the ongoing studies are provided in Section 5.5, Appendix 5.1 and in subsequent MCPs.

5.4.1 Tailings consolidation model

As part of Pit 1 closure planning, ERA commissioned a series of Pit 1 tailings consolidation models (Australian Tailings Consultants, 2003, 2007, 2009, 2012, 2014, Fitton 2015, 2017). These models allow the prediction of final tailings elevation within Pit 1 and the forecast volume of process water to be expressed during consolidation. The model was then later adapted for use in Pit 3. This section describes the model. Subsequent sections detail the specific models of both the Pit 1 and Pit 3 specific models.

The consolidation models have been supported by a number of other studies, including tailings characterisation and geotechnical investigations to predict the subsurface conditions for the final backfill design. These studies are summarised later in this section.

The consolidation modelling software was established in the late 1980s and is based on a formulation developed by Somogyi (1980). The initial purpose of the program was to provide inputs into a sophisticated water balance developed by the author for the Golden Cross Gold Mine in New Zealand (Murphy & Williams 1990).

The program solves the various partial differential equations describing self-weight consolidation using an implicit finite difference method. The author extended the original Somogyi model to include:

- a technique to allow for variable basin geometry and/or changing solids deposition rate with time
- underdrainage to atmospheric pressure
- the application of surcharges

The program models tailings deposition at user defined time steps. The current Pit 3 model is based on time steps of 0.1 days resulting in about 30,000 nodes for the deepest part of the pit.

The program also models quiescent consolidation with or without a surcharge.

The program was presented as a minor thesis (Murphy 1994) as part of a Master of Engineering Science at Monash University in 1994. The examiner was David Williams (now Professor) of the University of Queensland.

5.4.1.1 Method of addressing variable basin geometry

Variable geometry is addressed by considering the tailings impoundment as a series of five annular areas, as described in Appendix 5.2. As the tailings level rises, the effective discharge rate reduces as the area increases at each stage. At each stage, the mass of solids discharged into each annulus is modified to compensate for the greater consolidation settlement in deeper columns. The relative mass of solids deposited is greatest in the deepest column and reduces towards the edge of the TSF. This technique ensures that the model compensates for the greater settlement in deeper parts of the deposit. For example, in a deep pit, such as Pit 1 at the Ranger Mine, a dished surface does not exist until after deposition ceases. At this time, tailings no longer progressively fill the area above the deeper parts of the pit where consolidation is greatest and a "dish" subsequently develops.

The technique, developed in 1987, is effectively a pseudo 3-dimensional consolidation model and is believed to pre-date other such models. Figure 5-36 compares the actual Pit 3 at the Ranger Mine with the "as-modelled" pit. The "annular" boundaries are shown on the figure.

Typical density profiles for an earlier Pit 3 consolidation analysis are shown in Figure 5-37. The figure shows density profiles at the end of deposition. The impact of the effective discharge rate is seen as the degree of consolidation being greater for tailings of lesser depth at the end of deposition.

5.4.1.2 Underdrainage

Underdrainage is introduced into the model by allowing for seepage forces and negative excess pore pressure. The various pore pressures for an under-drained deposit are presented in Appendix 5.3.

It should be noted that at equilibrium, provided a water pond is maintained at the surface and the underdrain remains operational, there will be constant flow from the surface to the base. At this time consolidation is complete and the flow is constant seepage. This concept is illustrated in Lambe & Whitman (1997) page 258, Figure 17.11.

5.4.1.3 Outputs

Program outputs include:

- density, permeability, void ratio and effective stress profiles for each "column" at user defined times
- cumulative consolidation flows to the surface and base for each "column".

With respect to flows, the integrated flow out of the base of each "column", effectively determines the flow out of the base and sides of the pit.

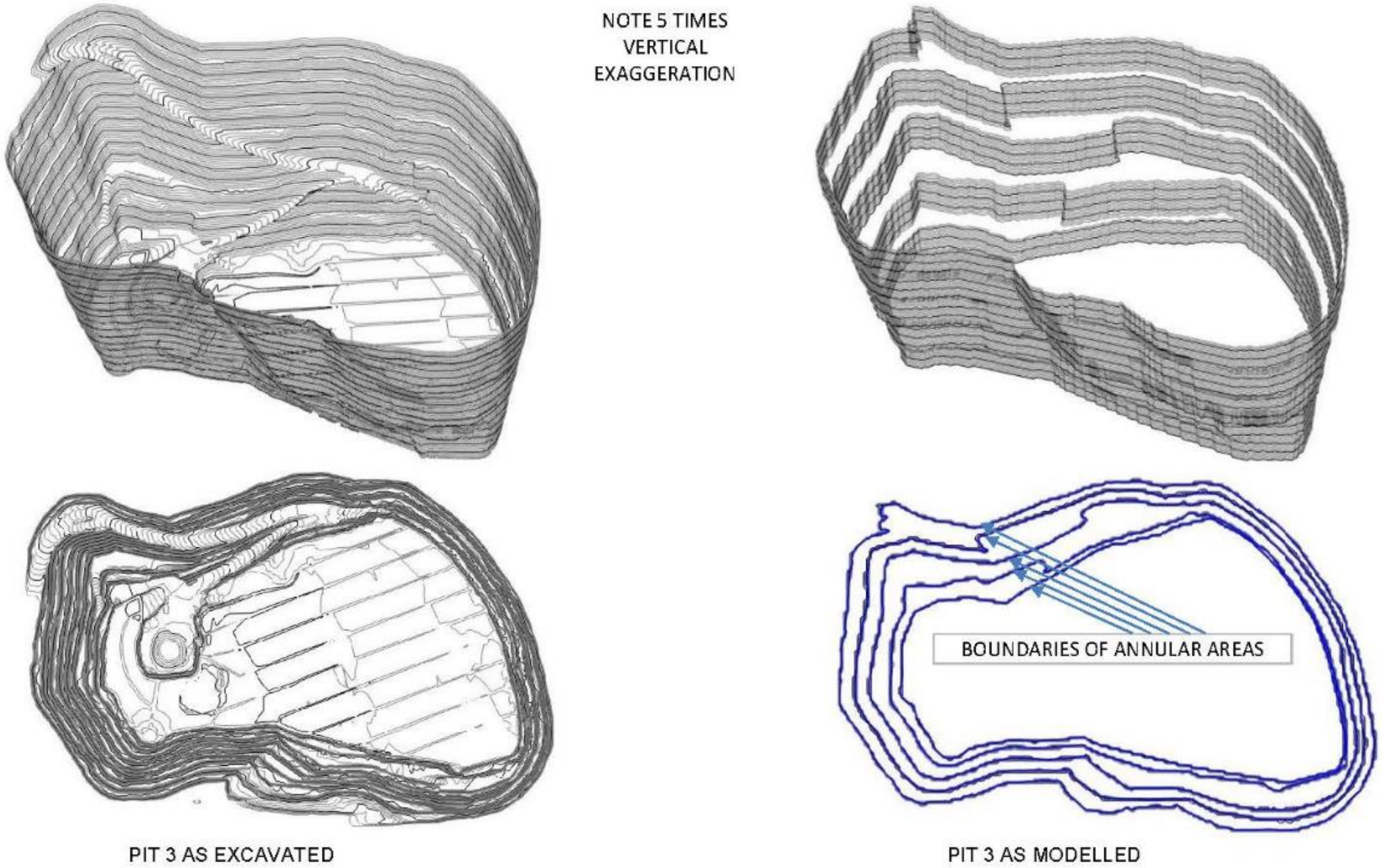


Figure 5-36: Pit 3 as excavated and as modelled

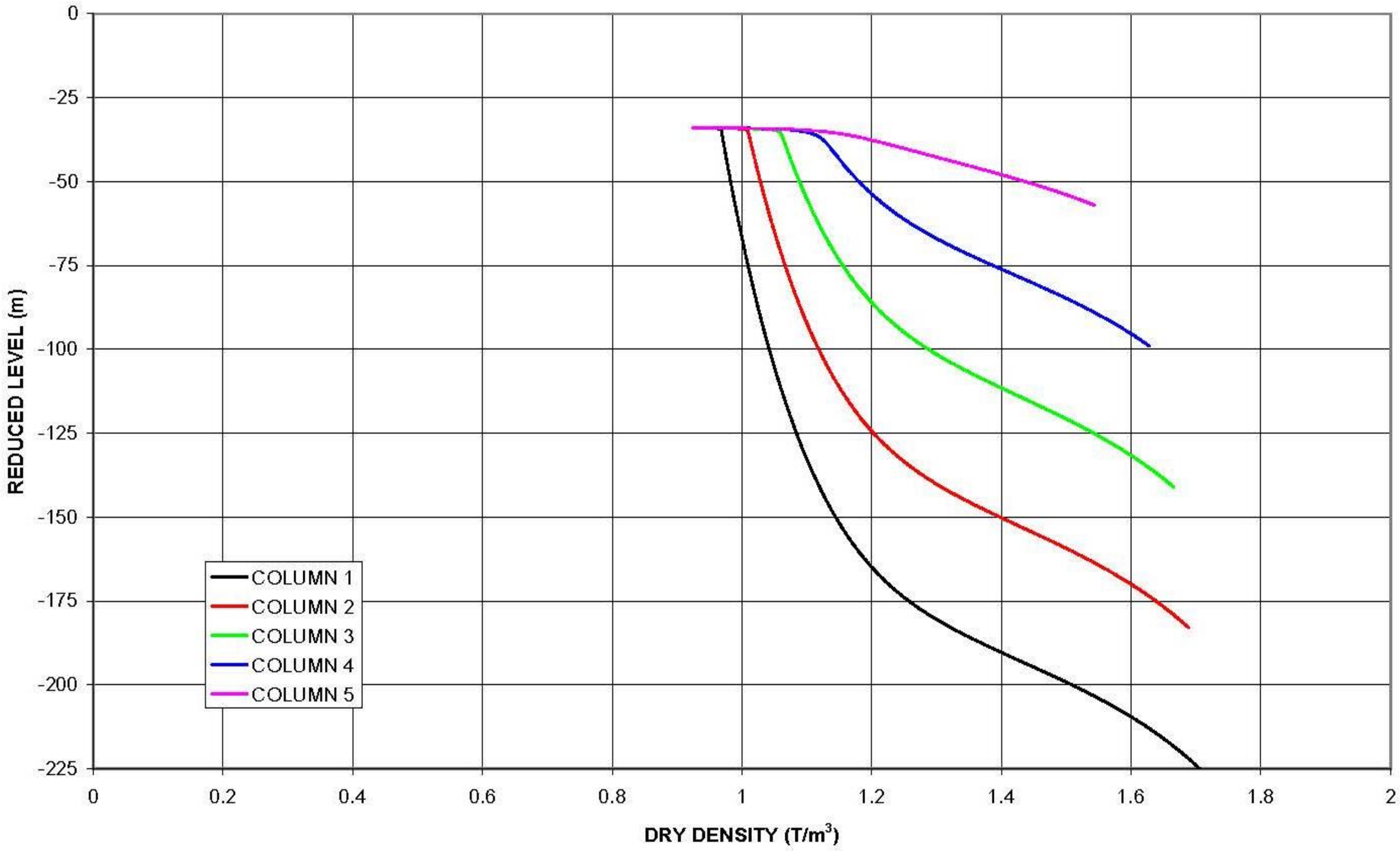


Figure 5-37: Pit 3 density profile - end of filling

5.4.1.4 Validation

The computer program was initially validated against a number of published examples (Townsend 1990). The Townsend paper presented the results of a number of scenarios whereby practitioners were invited to present solutions to the scenarios. All of the modelled scenarios resulted in excellent agreement.

The underdrain case was validated against a large-scale experiment carried out by Glenister & Cooling (1986). Again, the model showed excellent agreement and the author has been able to validate the model against many real applications including:

- Golden Cross Gold Mine New Zealand (Murphy 1997)
- Century Zinc Mine, Queensland (Murphy 2006)
- The Granites Gold Mine, Northern Territory (Murphy 2007)
- A coal mine in the Hunter Valley (Seddon & Pemberton 2015)

In these examples the model was able to predict:

- tailings elevation with time
- density profiles
- pore pressure profiles.

It should be noted that closure of Bullakitchie Pit (Murphy, 2007) at The Granites Gold Mine is featured as a case study in *Tailings Management: Leading Practice Sustainable Development Program for the Mining Industry* published by the Australian Government (2016). The original paper for this example was presented by the author at a conference in 2007.

5.4.1.5 Pit 1 tailings consolidation

Tailings consolidation modelling in Pit 1 has been ongoing since 2003. The ATC Williams 2012 model predicted that the average final tailings level in Pit 1 would be 7 mRL with a minimum level of 0.5 mRL in the centre and approximately 12 mRL near the edges. The predicted final tailings level across the pit is shown in Figure 5-38.

The model was updated in 2015 by Fitton Tailings Consultants (Fitton). Prior to the placement of the pre-load in the fourth quarters of 2013 and 2014, 28 settlement monitoring plates and standpipes were installed across the pit and were raised concurrent with the initial bulk fill layers. The monitoring plates enable regular verification and updating of the consolidation model; the most recent validation of the model was conducted by Fitton (2017). Ongoing measurements of tailings settlement are undertaken on a monthly basis (Figure 5-40) and confirm the model is still valid.

The validation is based on the settlement data from the monitoring plates and earlier consolidation models and confirms the consolidation rate. This validation also estimated the volume of expressed process water over time (Figure 5-39). These results indicate that most process water (greater than 99 %) will be removed via the decant structures by January 2026.

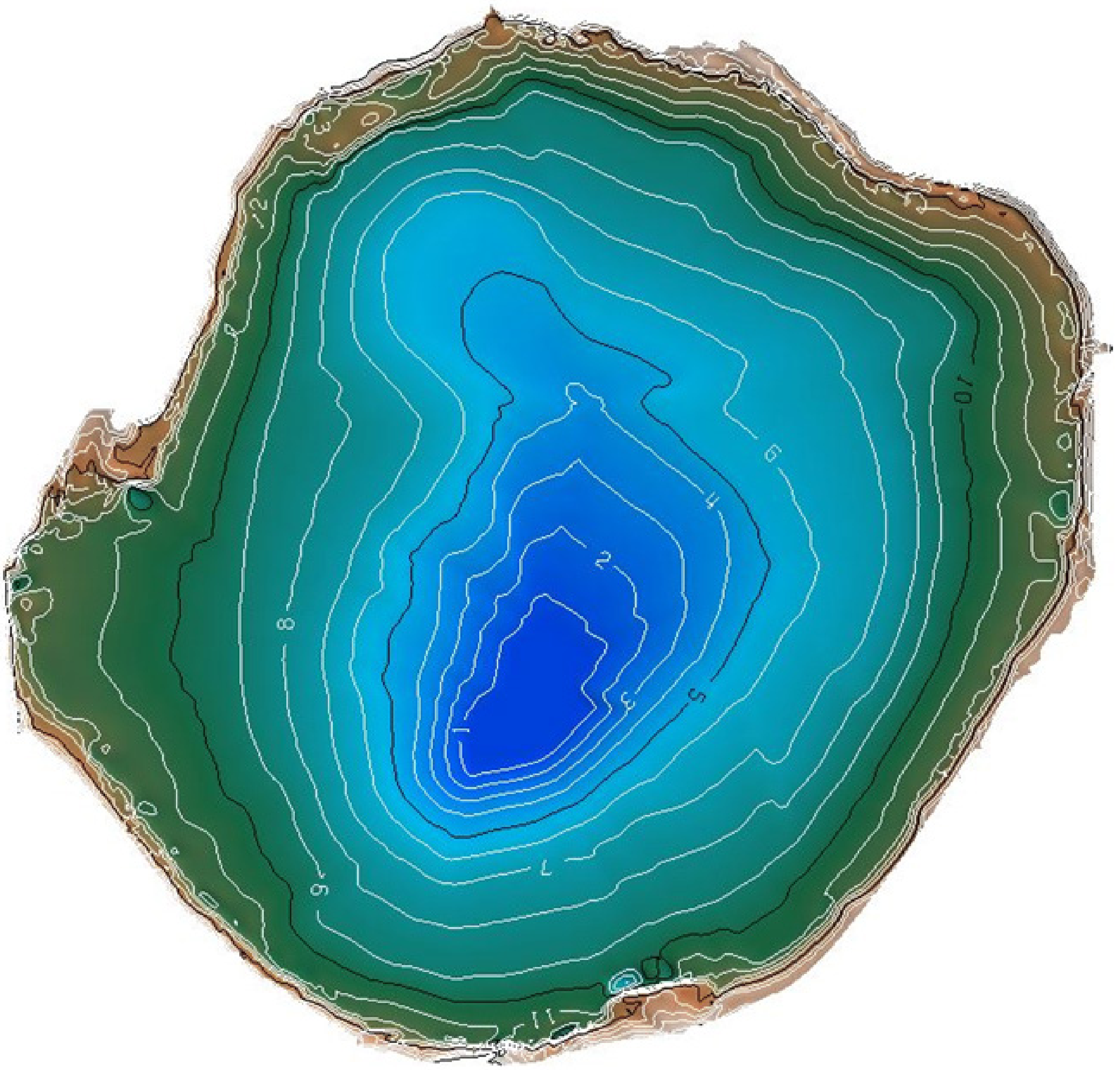


Figure 5-38: Predicted final tailings level (m) across Pit 1

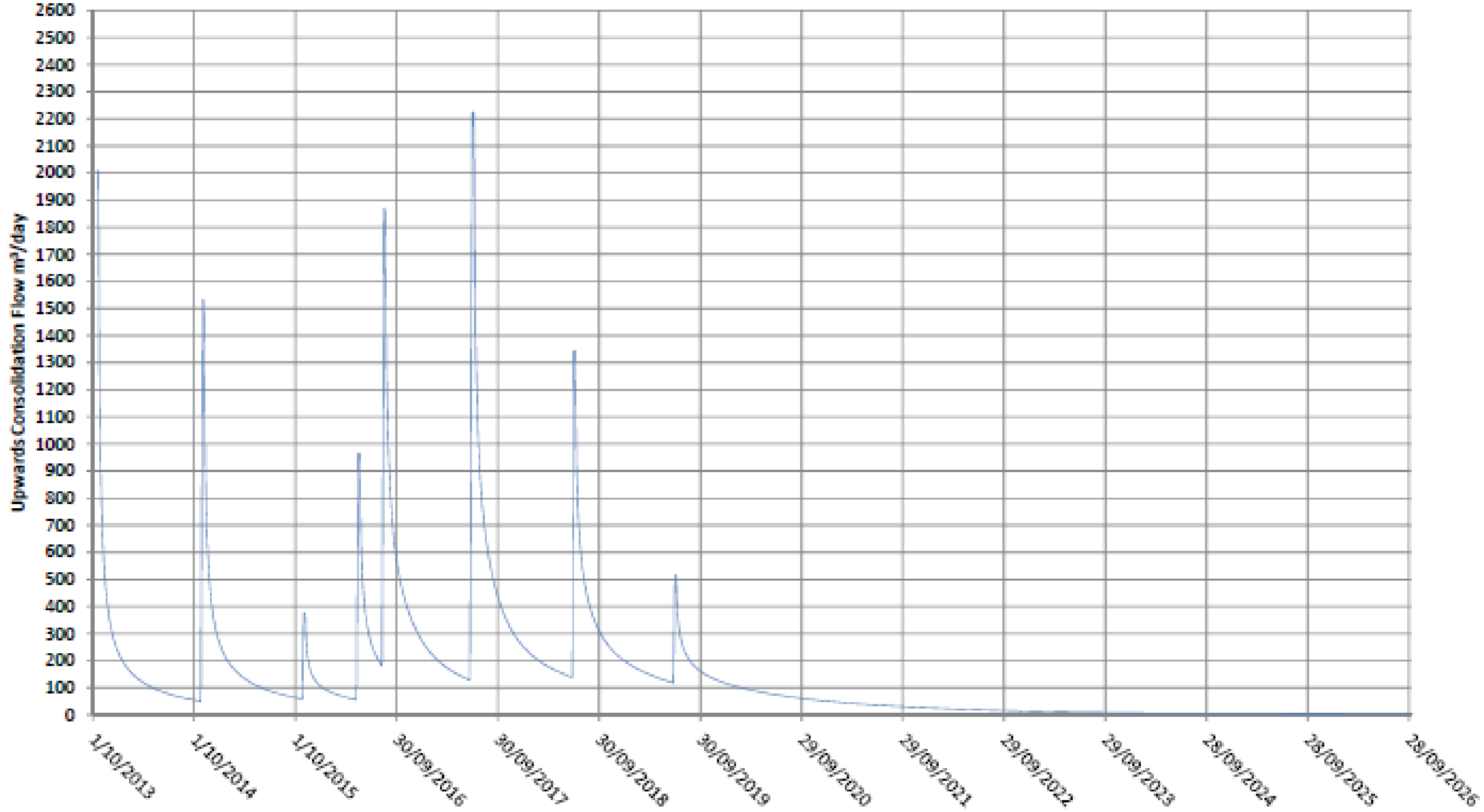


Figure 5-39: Predicted flow of process water from Pit 1 during consolidation (Fitton 2015, 2017; Figure 5)

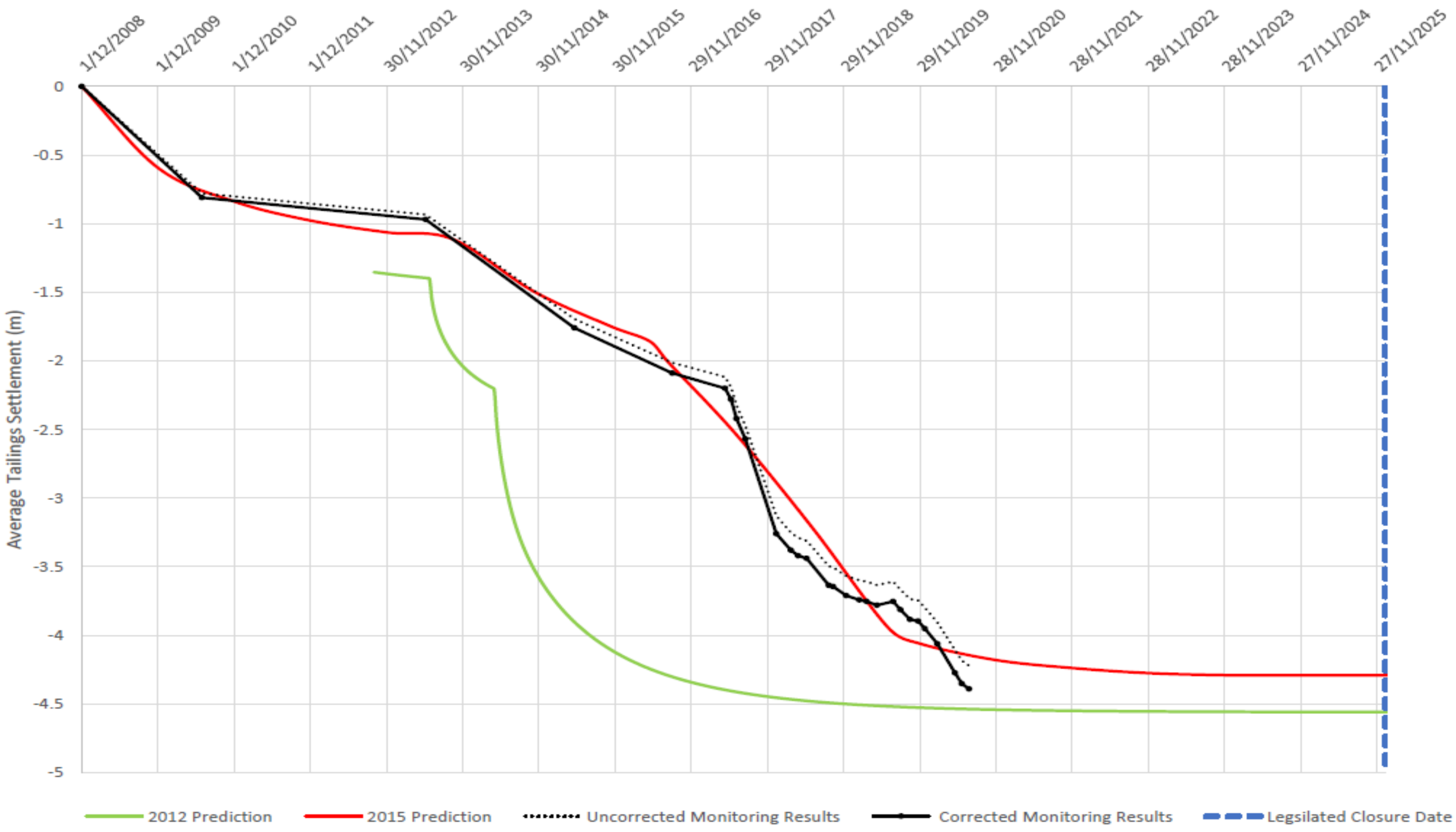


Figure 5-40: Predicted versus measured average tailings settlements in Pit 1

Available measurements relevant to flows in and out of the waste rock cap on top of Pit 1 have been used to construct a solute mass balance, using magnesium as the representative solute, and a water (volume) balance. Both balances have been conducted on a daily basis over a two year period, from 1 January 2017 to 31 December 2018. The solute balance indicates that the measured mass of solute recovered through the decant towers matches the mass of solute estimated to have been expressed from the tailings (Figure 5-41). Other sources of solute in the system are considered to be insignificant. The volume balance indicates that the decant structures are recovering additional volume from the waste rock cap, beyond that expected from catchment yield (rainfall less evaporation) and tailings consolidation flux. Both balances support the conclusion that all tailings consolidation flux is being recovered by the decant structures (Harvey 2019), an indication that the process water expressed by consolidation will be recovered for treatment before the end of rehabilitation activities in January 2026.

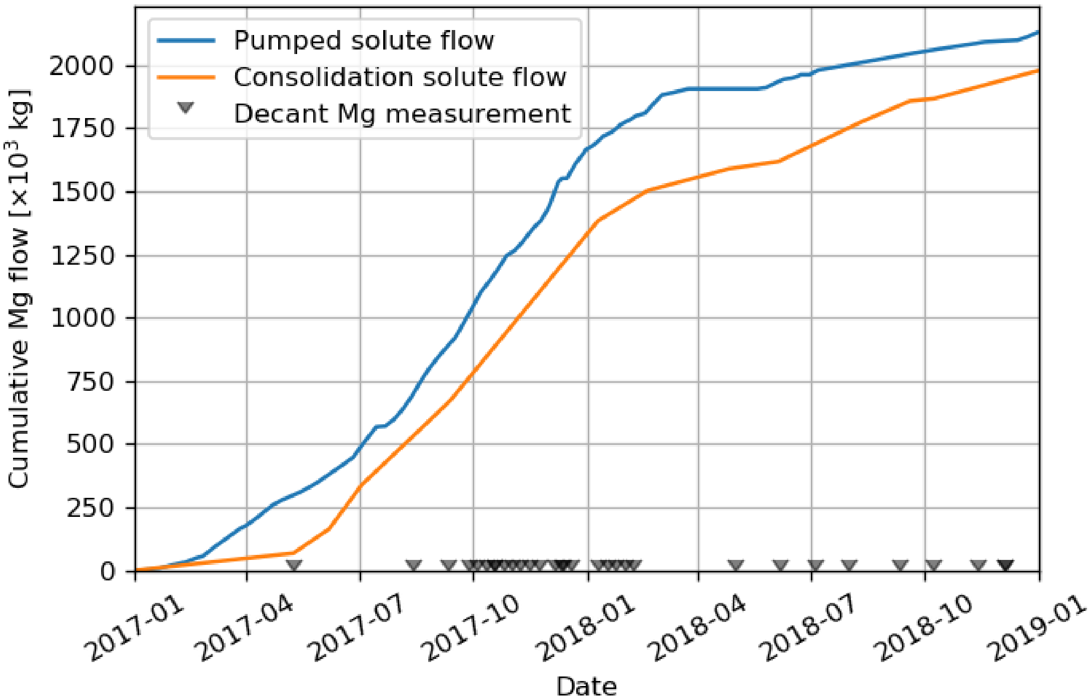


Figure 5-41: Cumulative magnesium flows

5.4.1.6 Pit 3 tailings consolidation

ERA made a submission to the Minesite Technical Committee (MTC) in August 2014, describing the assessment of potential environments impacts from the interim final tailings level in Pit 3 (ERA 2014a). Included in this submission were the results of the predicted tailings consolidation; excerpts of which are provided below, along with the most recent updates of the tailings consolidation model.

Australian Tailings Consultant (2014) outlines the various field and laboratory studies they have conducted to confirm the tailings geotechnical properties and provide up-to-date parameters for the in-pit tailings consolidation modelling.

Testing indicated that the geotechnical properties of the Ranger Mine tailings have and will continue to vary with time, likely due to the inherent variability of the ore type and historical changes to the process. To account for this and provide a sensitivity analysis, three sets of consolidation parameters were considered in the modelling as follows:

- conservative (i.e. relatively slow consolidation) model - based on a Rowe Cell test of the reconstituted sample of pre-1996 TSF tailings and recent mill tailings
- best estimate model - based upon 'best fit' curves from Rowe Cell test results
- non-conservative (i.e. relatively fast consolidation) model - based on the consolidation process in Pit 1.

Consolidation modelling was conducted for all three parameters. Results demonstrated that consolidation could be achieved by 2026 for all cases. The consolidation model was updated to reflect the "as constructed" situation in early 2016 and was completed for the best estimate case only. The model was again updated in 2018 to understand the impact of tailings segregation, and estimate the tailings surface over the deposition and post deposition phases. Results of the consolidation models are provided in Table 5-21. These show that the majority of parameters are essentially the same. They achieve effective consolidation by December 2026, indicating that wick drains will be required to promote consolidation and achieve the January 2026 target. However, less wick area is now required across the surface of Pit 3, in order to achieve a similar consolidation result reported in 2014. Water expression, during deposition, for the May 2016 analysis is 30% greater than for the February 2014 analysis because the thickener was deleted from the former case, and the impact of the thickener is readily apparent. For the thickened case, there is 1.9 m^3 of water per tonne of solids less arriving in Pit 3. The difference between the dry density at deposition and the end of deposition is significantly less for the thickened case and thus the water expressed during the deposition phase is less. The consolidation model is currently being updated. The results will be included in the next MCP.

The consolidation model for Pit 3 was verified with the results from the cone penetration test (CPT) conducted in the Pit in the latter part of 2018 (Fitton 2019). It was noted that the measured excess pore pressure profiles closely agree with those predicted by the consolidation model. Figure 5-42 shows a typical comparison between the measured and predicted excess pore pressure profile.

Wick drains will be installed to promote the consolidation (Figure 5-43), similar to those which have been installed in Pit 1. A rock drainage layer will be installed on top of the tailings to act as an interception layer so that water expressed up through the tailings can be pumped out (Figure 5-43). Expression of tailings pore water with respect to local scale and regional scale ground water impacts is to be assessed within the groundwater solute transport modelling being undertaken by INTERA. A detailed assessment of the post-closure Mg loading to Magela Creek from Pit 3 tailings was undertaken to support the Pit 3 tailings deposition application, this study specifically considered the heterogeneous nature of the deposited tailings following consolidation. Figure 5-44 shows the flow of process water in Pit 3 estimated from the most recent model.

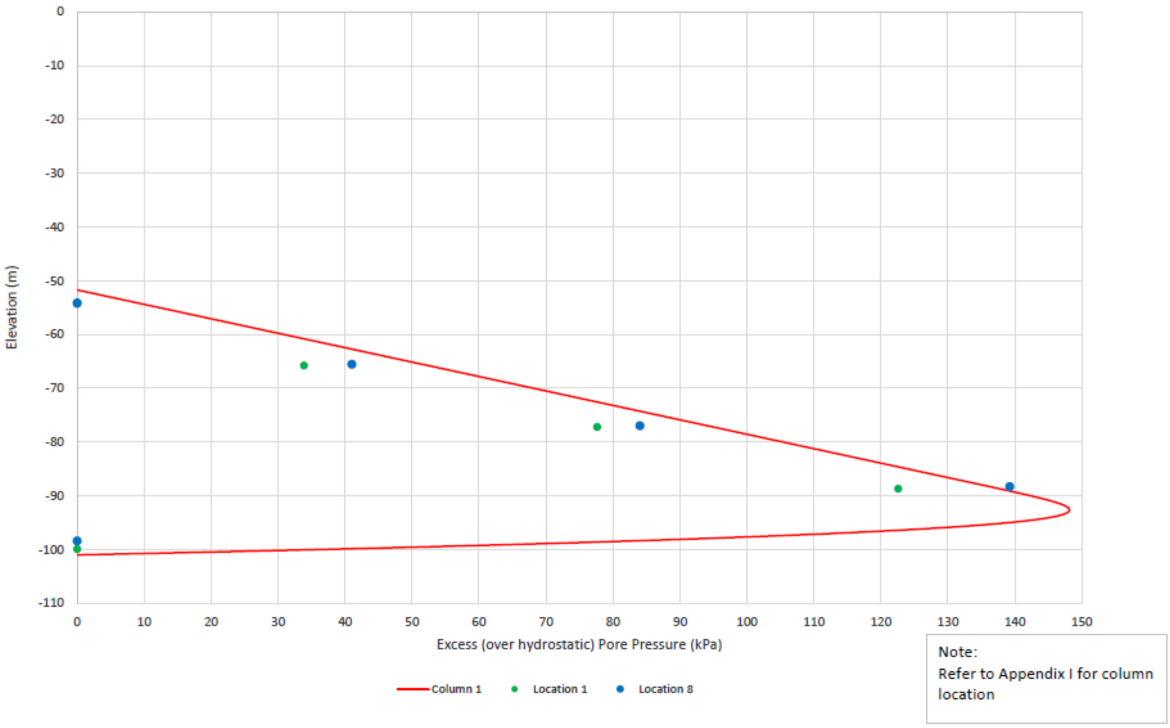


Figure 5-42: Measured versus predicted excess pore pressure profile

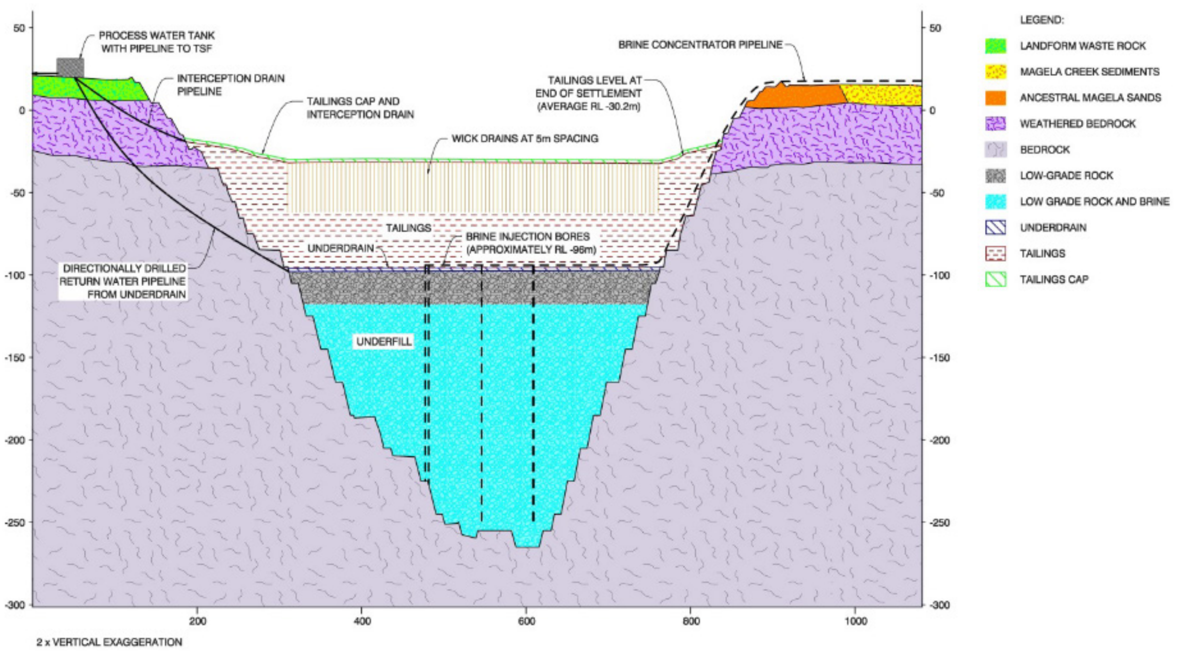


Figure 5-43: Indicative conceptual cross-section of Pit 3 at the end of consolidation, as at 2014 (INTERA 2014a)

Table 5-21: Consolidation model results, comparison of 2014 and 2018

	February 2014	May 2016	2018
Average base level (mRL)	-100	-99.7	-99.7
Underfill/drain volume (m^3)	15,298,380	15,658,180	15,658,180
Tonnes	41,781,246	40,345,324	40,345,324
Deposition duration (yr)	5.75	5.92	6.00
Thickening?	After year 1	No	No
Dry density - end of deposition (t/m^3)	1.42*	1.39	1.35
Dry density - end of consolidation (t/m^3)	1.68	1.66	1.63
Average level -end of deposition (m)	-21.30	-21.53	-20.00
Average level - end of consolidation (m)	-31.0	-31.3	-30.3
Average cover depth (m)	48.64	48.94	50.93
Cover volume (m^3)**	25,292,800	25,448,800	26,534,530
Water expressed - during deposition (m^3)	14,707,410	21,938,520	16,860,080
Water expressed - post deposition (m^3) ***	4,370,360	4,721,000	5,163,690
Wick area (m^2)	238,235	416,216	145,000
Water expressed by wicks (m^3)	2,334,780	2,125,840	430,439
Consolidation complete	May 2027	May 2027	May 2028
Consolidation practically complete****	February 2025	December 2024	June 2025

* The number of decimal places presented in this table does not imply a level of accuracy. The numbers are presented to identify, sometimes, small differences in results.

** In previous reports, volumes were based on an adopted pit edge. The volumes in this table are less than previously presented as they have been based on final tailings area in accordance with this report.

*** Includes wick volume.

**** Based on removal of 95% of mobile pore water

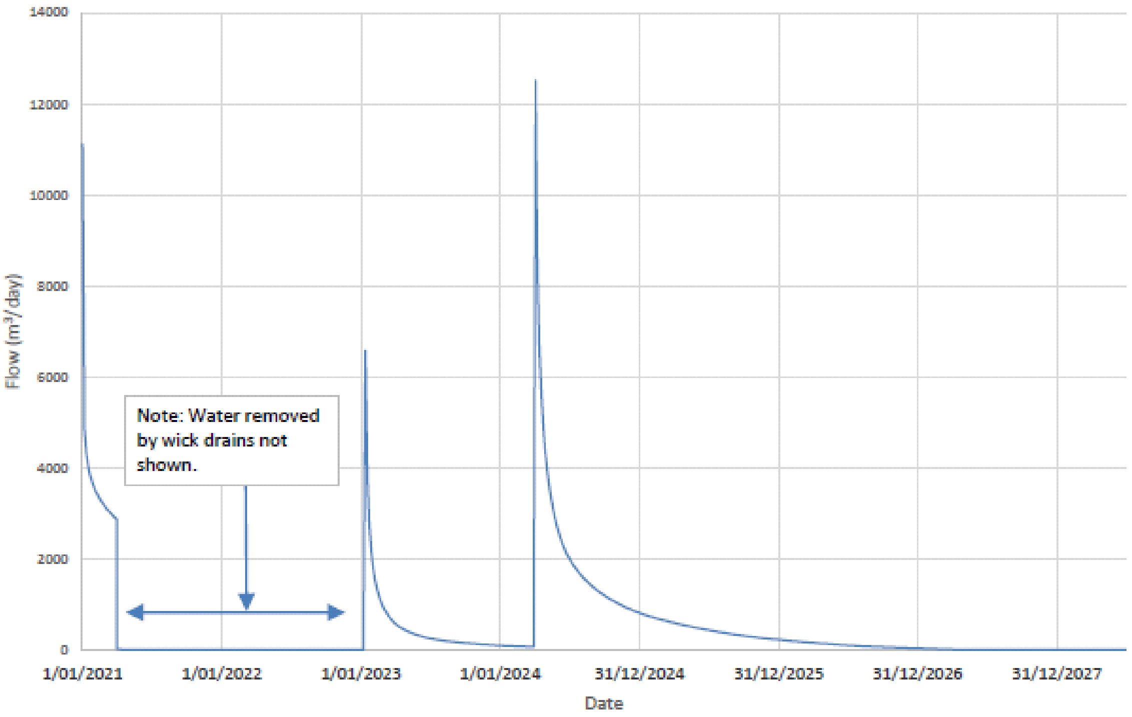


Figure 5-44: Predicted flow of process water from Pit 3 during consolidation

A new tailings deposition strategy has been developed for Pit 3 (Fitton 2019). This involves subaerial discharge from five spigots (DP1- DP5) from the eastern end as shown in Figure 5-45. and subaqueous discharge from two diffusers, from locations 1-15, on the western end as presented in Figure 5-46.. The adopted deposition method is based on the outcome from BPT workshop (GHD 2019). The tailings deposition into Pit 3, per the new strategy, will be monitored by conducting monthly bathymetric and six-monthly geophysical surveys, along with yearly CPTs. The results from these investigations (bathymetric, geophysical and CPTs) will be utilised to review and amend the deposition plan if required and review the consolidation model.

Refer to Section 9.3.2 for more information on current tailings deposition in Pit 3.

Figure 5-45: Mill tailings deposition locations

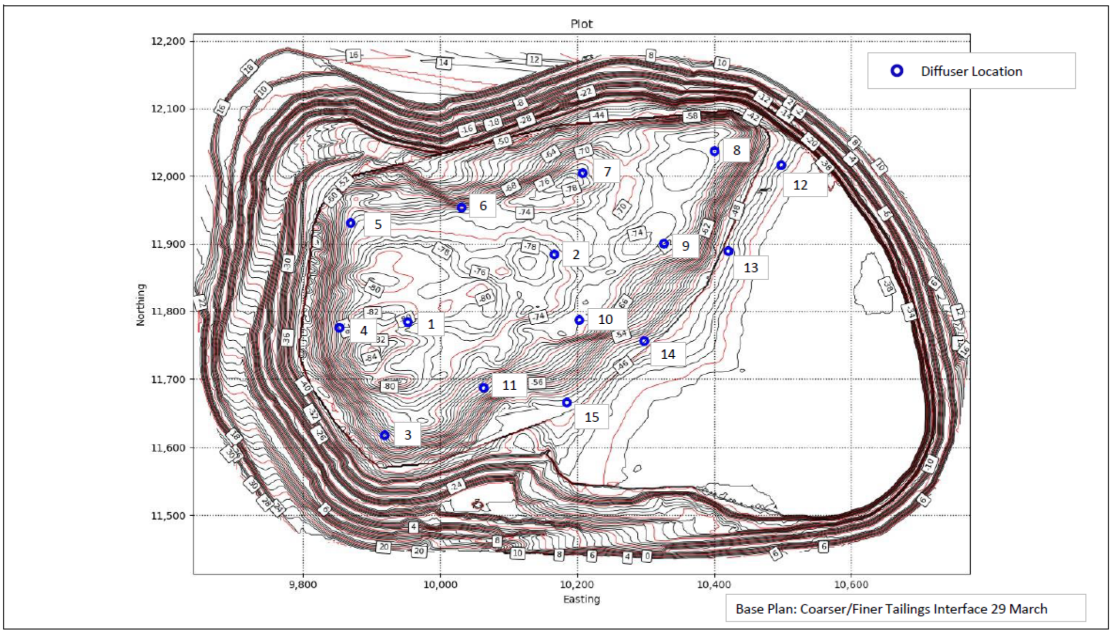


Figure 5-46: Diffuser locations

5.4.2 Tailings properties

Around 40 Mt of dry tailings from the mill and the TSF will be transferred to Pit 3 by January 2021. It was calculated that tailings would be deposited to a thickness of approximately 80 m and a volume of about 30.3 Mm^3. Section 9.3.2 provides details of tailings transfer activities.

Tailings transfer from the TSF is supported by a number of studies undertaken in order to validate the expected tailing volumes and also to provide key information to feed into the overall dredge program currently underway. Studies included:

- TSF geophysical surveys (Fugro 2012 and 2018) (Figure 5-47)
- TSF magnetometer survey (Fugro 2012)
- Magnetic survey (Surrich 2019)
- TSF characterisation and CPT program (Shackleton 2013; in2Dredging 2020).

5.4.2.1 TSF Bathymetric surveys and geotechnical investigation

Prior to commencement of dredging and every quarter during the dredging operation a bathymetric survey was completed. The initial bathometric survey determined that there were 23.1 Mm^3 of tailings contained within the TSF. As of June 2019, 11.8 Mm^3 of tailings had been dredged to Pit 3. Typical survey results are presented in Figure 5-47.

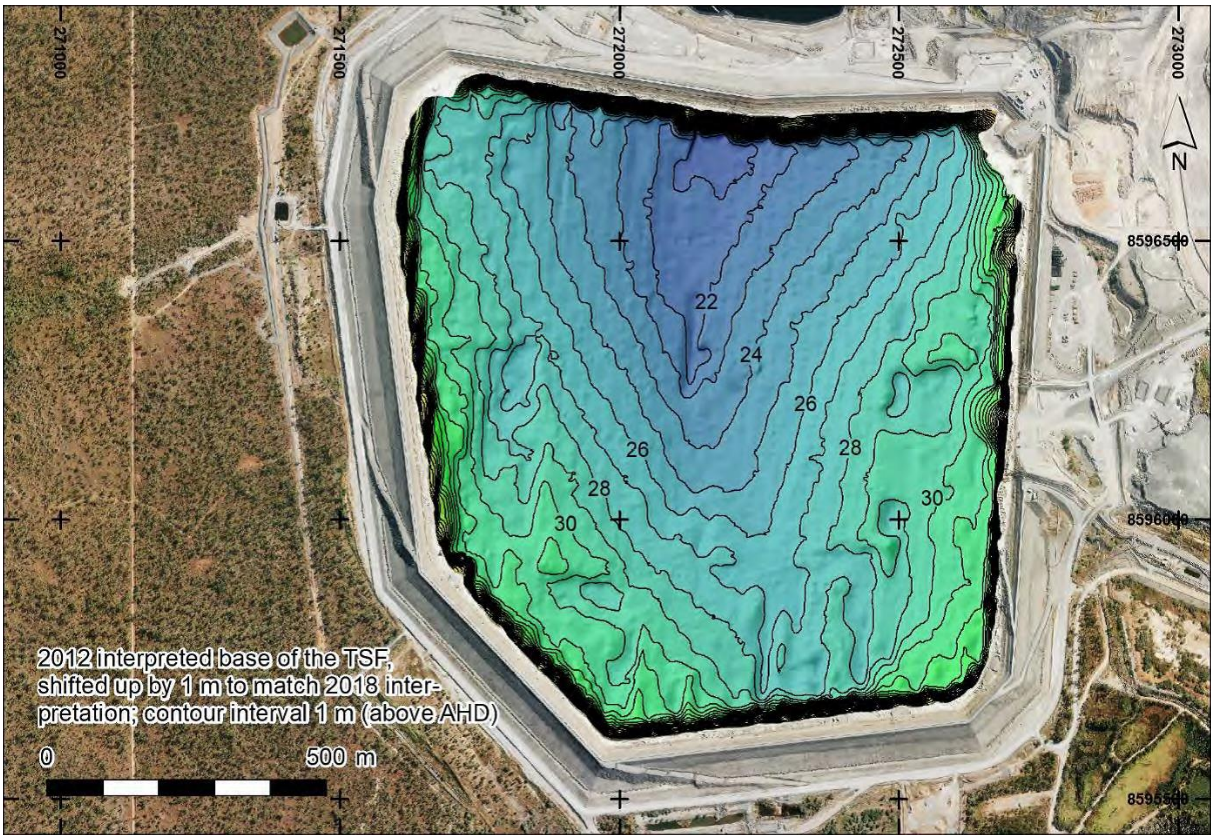


Figure 5-47: TSF topography (blue: low elevation; green: high elevation) (Fugro 2018)

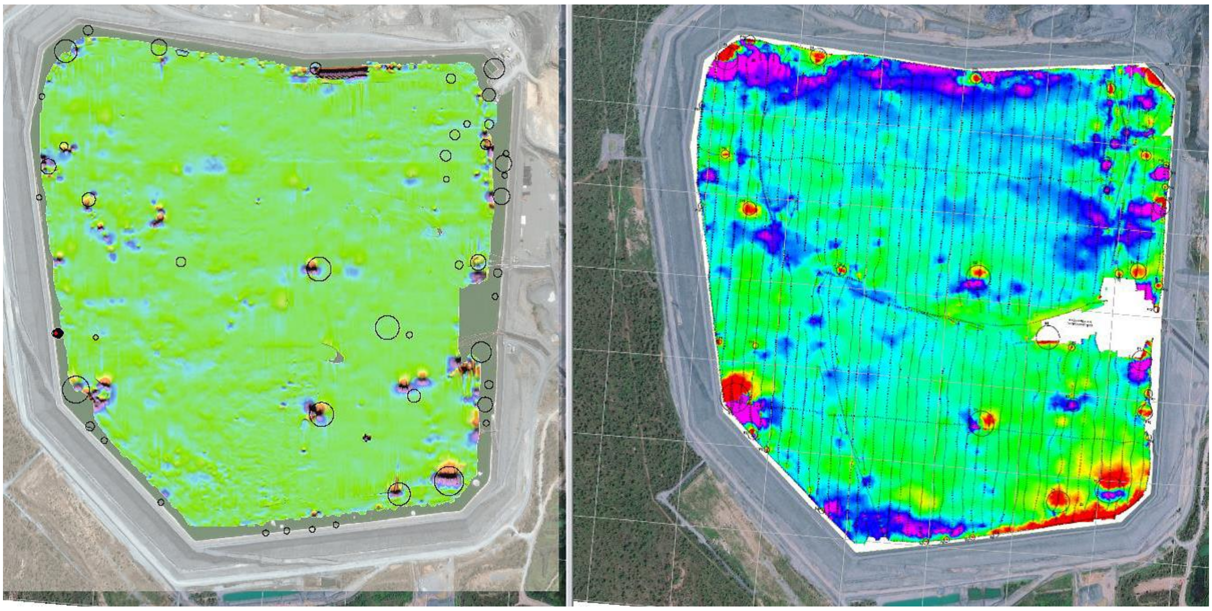


Figure 5-48: April 2019 Magnetic Anomaly Map (left frame) comparison with the 2012 Magnetic Anomaly Map (right frame)

Magnetometer surveys provide magnetic intensity data from a towed magnetometer. The data from the 2019 magnetometer survey compared to that from 2012 is shown in Figure 5-48. The primary objective of the survey was to locate any potential buried iron objects which could impact proposed dredging operations.

As expected, 'magnetic' objects were identified close to the TSF embankments, whilst the central area was relatively free of anomalies. The magnetometer detected a very strong anomaly on the south-eastern side of the dam, believed to be the sunken remains of the old survey barge/pontoon. No other features of similar magnitude were found. Many anomalies, either localised or diffuse, are likely to be caused by magnetic material in the tailings, accentuated by variations in the water depth that changes the range between source and detector. Small, localised anomalies, particularly around the TSF perimeter, probably represent iron debris.

Between 27 August and 25 November 2012, ATC Williams was assigned to undertake an investigation into the *in situ* condition of the tailings in the TSF (Shackleton 2013). This study was undertaken during the integrated tailings, water and closure (ITWC) prefeasibility study (PFS); designed to gain a better understanding of the conditions within the TSF and facilitate the selection of an appropriate dredge and pumping equipment, along with the design of a feasible work method. This work entailed cone penetrometer tests and tailings sampling.

The data analysis from the CPTs, laboratory results and onsite observations indicated two separate zones within the TSF:

1. an outer zone comprising of sands and silty sands, overlying a sandy layer, followed by the foundation on the perimeter of the TSF in shallower water (Figure 5-49 blue)

2. an inner zone of under consolidated fines of very low strength, overlying a sandy layer, followed by the foundation, located within the deeper sections of the TSF (Figure 5-49 brown) (Shackleton 2013; p 11).

The outcomes of the TSF geophysics and magnetometer surveys validated the expected tailings volumes and provided valuable knowledge on the segregation and characterisation of tailings in the TSF. These studies together with the CPTs assisted the overall design of the TSF dredge and subsequent dredging method. Additional geotechnical investigation was carried out in the TSF by in2Dredging (May 2020) to augment the previous investigation conducted by ATC Williams (2012). It involved CPTu, vane share test, and tailings sampling. The study determined the undrained shear strength of the tailings and the approximate floor of the TSF to optimise the use of the two dredges, Brolga and Jabiru (In2Dredging 2020).

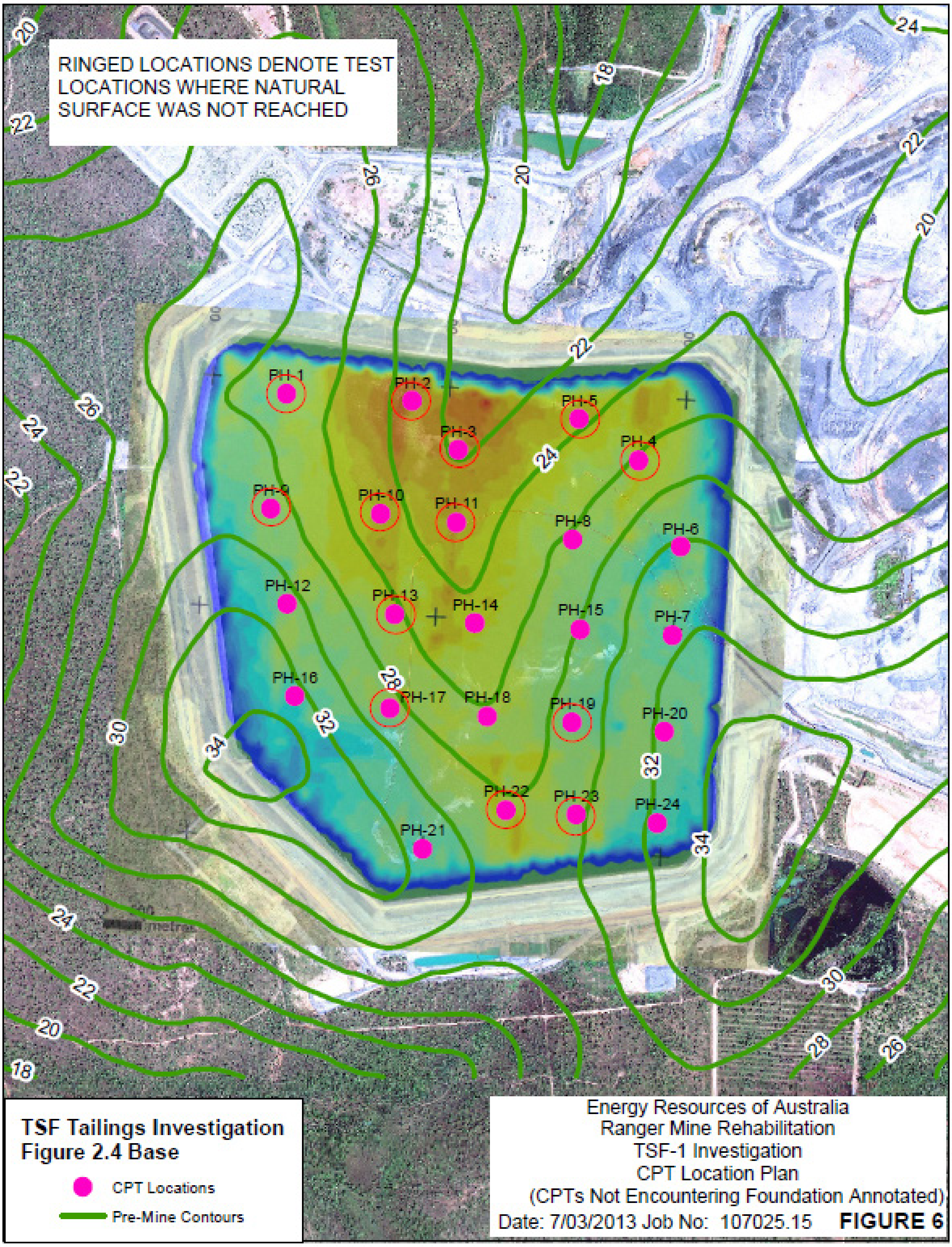


Figure 5-49: Cone penetration locations (Shackleton 2013)

5.4.2.2 Pit 3 geotechnical investigation

A geotechnical investigation was conducted in Pit 3 from October to November 2019 to verify the consolidation model (Fitton 2020b). It involved cone penetration test with pore pressure measurements (CPTu) at locations shown in Figure 5-50. A few tests locations from 2018 investigation were re-tested to understand how the fine tailings consolidation was occurring. Details of the CPTu is summarised in Table 5-22 Details of 2019 CPTu.

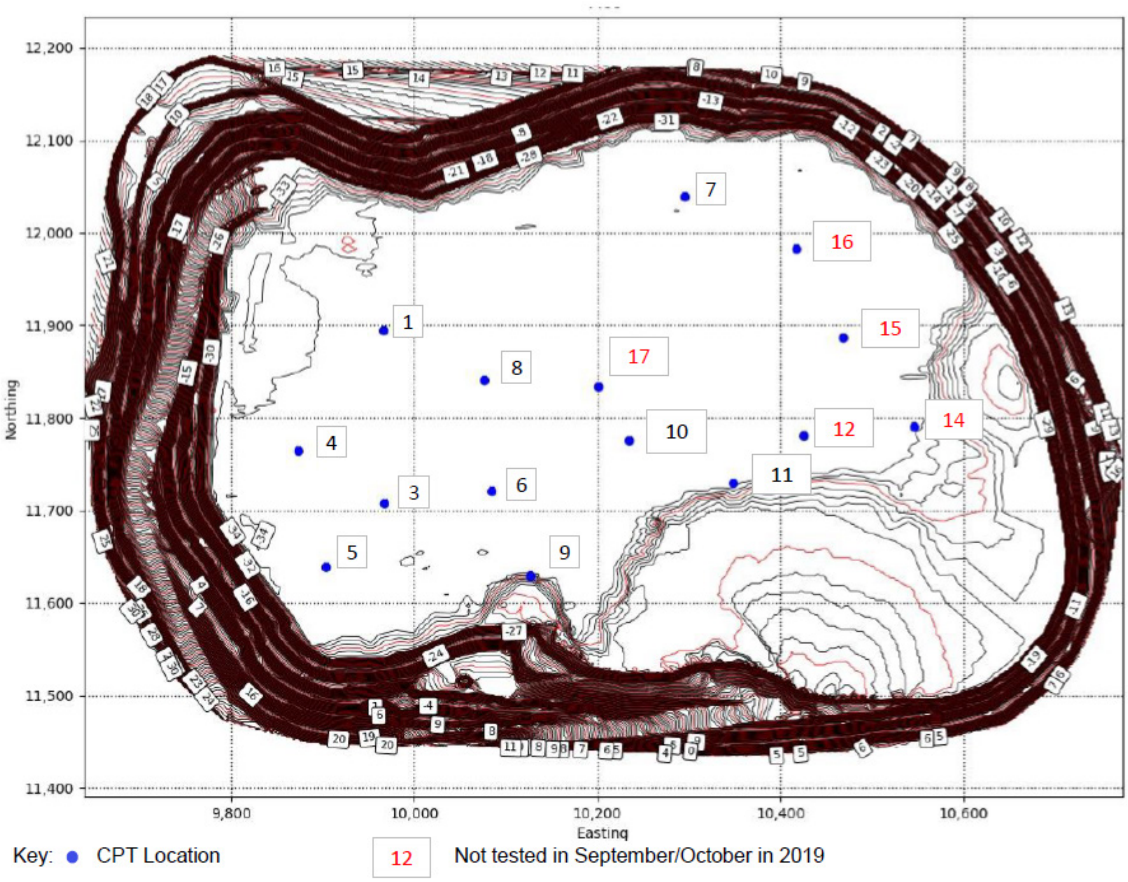
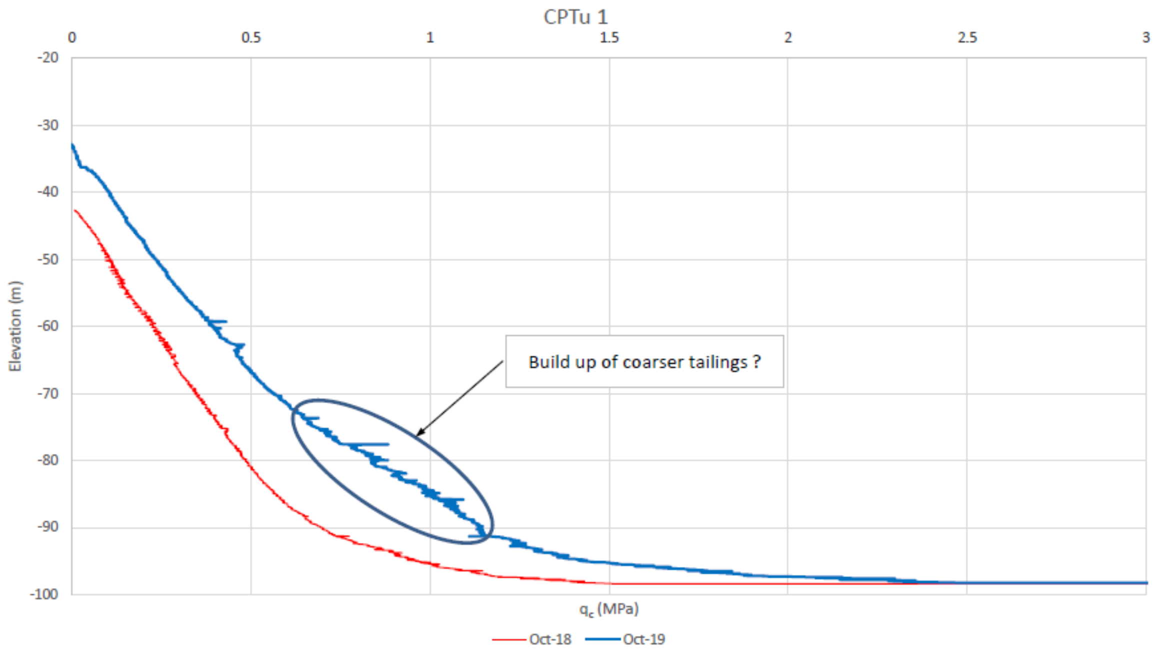


Figure 5-50 CPT Locations

Table 5-22 Details of 2019 CPTu

CPTu Location	Date of Test	Previously Tested (2018)	Recorded Water Depth (m)	Water Level RL (m)	Depth of Penetration (m)
1	4/11/2019	*	1.5	-32.82	65.5
3	30/10/2019	*	2.7	-32.60	68.8
4	2/11/2019		1.8	-32.68	61.1
5	31/10/2019		2.2	-32.51	50.0
6	17/10/2019	*	3.5	-32.81	51.1
7	11/10/2019		3.0	-32.63	40.2
8	7/11/2019	*	1.4	-32.80	67.4
9	16/10/2019	*	3.5	-32.83	51.8
10	8/11/2019	*	1.8	-32.77	38.1
11	6/11/2019	*	1.5	-32.83	34.7

The CPTu results indicated a clay like soil behaviour type at locations at 1, 3, 4, 5, 6, 7, 8 and 10, and a sand like soil behaviour type at locations 9 and 11. The cone resistance recorded at 1, 3, 6, 8, 9, 10, and 11 from the 2019 investigation is greater than that of 2018, indicating that the in-situ density and undrained shear strength of the tailings have increased and thus pore pressure dissipation and hence consolidation of the tailings has occurred. A typical cone resistance comparison profile is shown in Figure 5-51



q_c Uncorrected cone resistance

Figure 5-51 Typical 2018/2019 cone resistance comparison

One of the outputs from the consolidation model is the fine/coarse tailings boundary, which was determined with the cone resistance and compared with the predicted interface (Figure 5-52). The predicted and the measured boundaries are in close agreement.

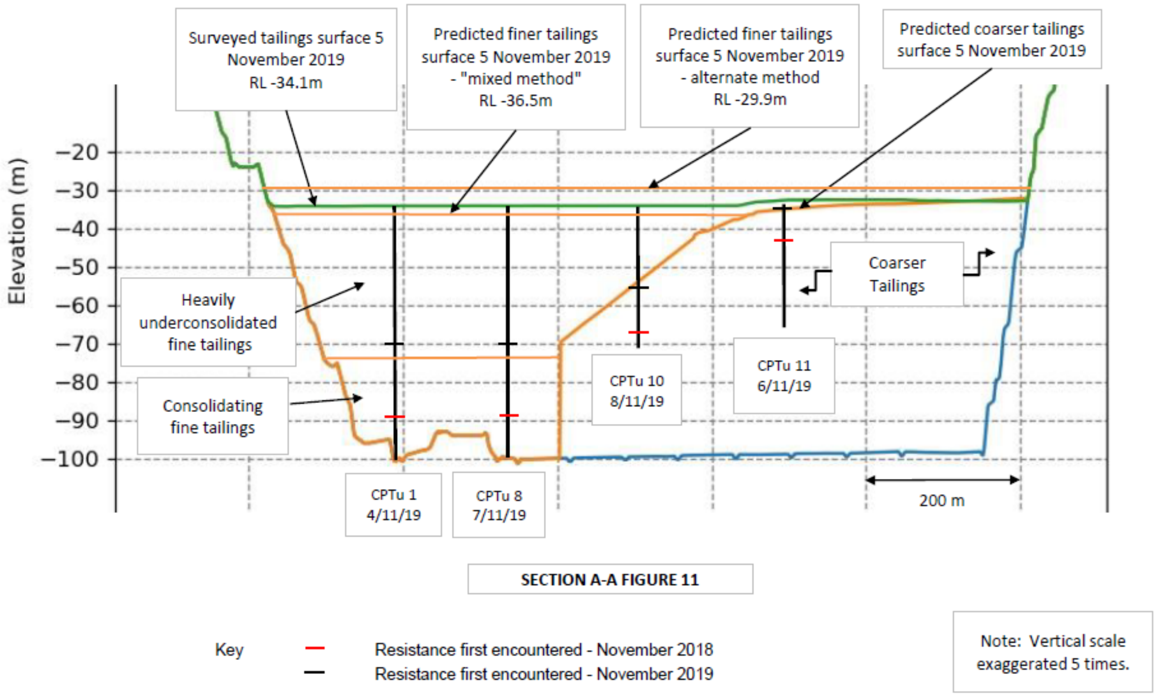


Figure 5-52 Predicted versus measured fine/coarse tailings interface

It is planned to undertake another geotechnical investigation in Pit 3, from September to November 2020, to verify the consolidation model and provide tailings parameters for the capping design. The investigation will comprise cone penetration test with pore pressure measurements, pore pressure dissipation test, vane shear test, tailings sampling and laboratory testing. After completion of tailings deposition into Pit 3, the tailings consolidation model will be updated then utilised for the settlement monitoring during Pit 3 capping and bulk backfill period.

5.4.2.3 Pit 3 geophysical surveys

A geophysical survey was conducted in December 2019 by Fugro Australia Marine Pty Ltd (Fugro), in Pit 3 to determine the distribution of tailings and their quantity within the pit. The survey used echo sounding to locate the tailings surface and Boomer and Chirp sub-bottom seismic profiling to investigate the tailings. The volumes of tailings and water in pit, established from the survey, are summarised in Table 5-23 and their surfaces presented in Figure 5-53.

Table 5-23 Summary of Geophysical survey

No.	Volume	Quantity [mm^3]	Comment
1	Water	0.55	The top of the water (i.e. the water level) is taken from the limit of the bathymetric survey and interpolated up to -31.067 m AHD (average water level) on the DTM of the pit shell. The volume of water represents the difference between the Water Surface (dark blue) and Top of Tailings surface (light blue) (or Total Tailings)
2	Total Tailings	24.19	The Total Tailings volume is provided by the difference between the Base of Tailings surface* (pink) and the Top of Tailings surface (light blue)
3	Total Pit Fill	24.74	The total pit fill volume is the sum of the water and total tailings volumes
4	Delta Total Pit Fill Difference between April 2019 and December 2019	3.13	The difference between the total pit fill between the April and December 2019. The total pit fill volume in April 2019 survey was 21.61 mm^3 It is noted that the difference in water volumes between April and December 2019 surveys is 1.72 mm^3 and the difference in Total Tailings volumes is 4.85 mm^3

*Note it is assumed the tailings include the filter layer or underdrain surface

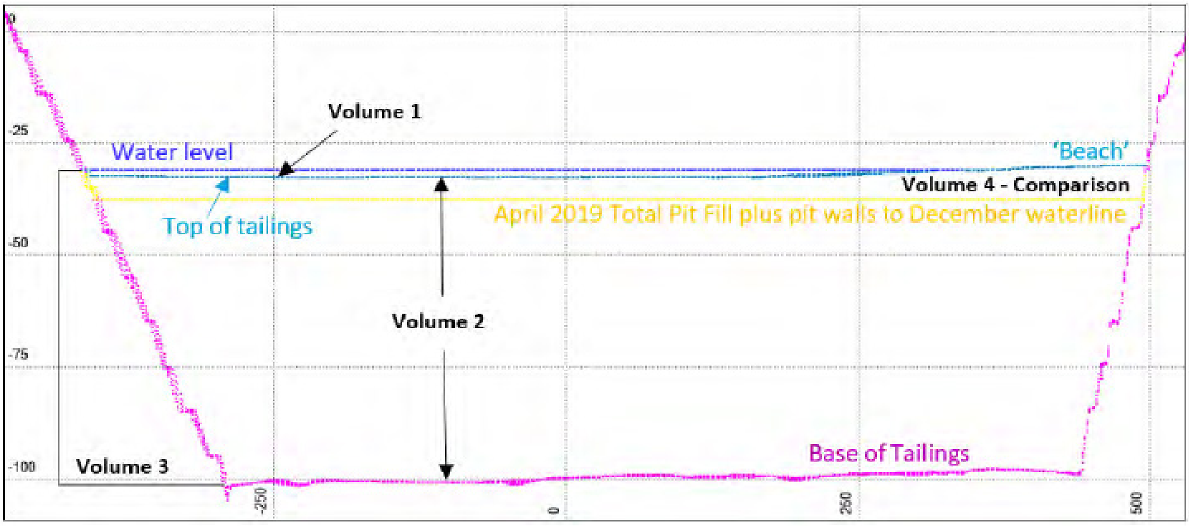


Figure 5-53 Cross section of tailings and water within the Pit

The volume of water, total tailings and total pit fill, estimated during the investigation, is 0.55 Mm3, 24.19 Mm3 and 24.74 Mm3, respectively. The total pit fill increased by 3.13 Mm3 since the previous survey in April 2019. It should be noted that the results from the geophysical surveys are usually used to augment the CPTu data, especially the fine/coarse tailings interface and mass ratio, to verify the consolidation model. The 2019 survey could not determine the fine/coarse tailings boundary due to the low depth of water (< 2m), in the pit, during the survey. It is understood that at least 7 m depth of water is required to establish the fine/coarse tailings interface. As this water depth is not likely to be achieved to the end of operations (January 2021), ERA has explored alternative methods to the geophysical survey, including the use of the "SmartDiver" and "Eorca" equipment, to establish the fine/coarse tailings boundary. Recent site water balance modelling suggests that there is a potential to achieve a minimum of 5 m water depth in the Pit in April 2021, and hence the potential to conduct the final geophysical survey. Results from this survey will be utilised in the final tailings consolidation model update and proposed wick installation in Pit 3.

5.4.3 Groundwater modelling

5.4.3.1 Ranger Conceptual Model

The Ranger Conceptual Model (RCM) was initially developed by INTERA in 2016. In 2018 ERA requested that INTERA undertake a review and update conceptual and numerical models for groundwater flow for use in assessment of potential impacts from post-closure conditions at the mine in accordance with requirements in the Ranger Authorisation. INTERA completed the update to the Ranger Conceptual Model in March 2019.

The update to the Ranger Conceptual model included:

- incorporation of recent information gained since completion of the previous RCM in 2016
- increase of the domain of the site wide model to encompass all source material and post-receptors
- calibration of all hydraulic properties using all appropriate observed data from the pre-mining period through to present
- inclusion of the full range of mining related stresses on the groundwater system

The calibrated flow model is intended to provide the foundation for simulating groundwater flow and transport from all mine sources to potential receptors under post-closure conditions. The RCM report describes the data, methods, and results for the site wide hydrogeological conceptual model update; construction, calibration, and sensitivity analysis of the site wide groundwater flow model; and completion of a preliminary groundwater flow model for post-closure conditions. The executive summary from the 2019 Ranger Conceptual Model report is provided below.

The conceptual model for the new site wide domain was iteratively updated through compilation and examination of all available climate, surface water, groundwater, geologic, and bore data to provide the highest level of detail and confidence in accordance with the modelling objectives and available resources. The updated conceptual model describes the most important hydrogeologic elements governing groundwater flow and transport at the Ranger Mine. The work produced data sets from nearly 2,000 exploratory bores, many hundreds of monitoring and other bores, many dozens of pump and slug tests, all major geologic contacts, more than 80,000 individual groundwater head measurements collected at more than 450 monitoring bores across the sitewide domain, and information about rainfall, evapotranspiration (ET), and creek stages spanning 37 years from 1980 to 2017.

The Ranger Conceptual Model domain was expanded to encompass all available information both upstream and downstream of the Ranger minesite. The conceptual model domain is larger than that for the calibrated groundwater flow model in order to use data outside of the model domain to constrain the HLU extents at the model boundaries and to define HLUs for an area large enough to fall within an appropriate extent for post-closure groundwater flow and transport modelling. The model domains are presented in Figure 5-54.

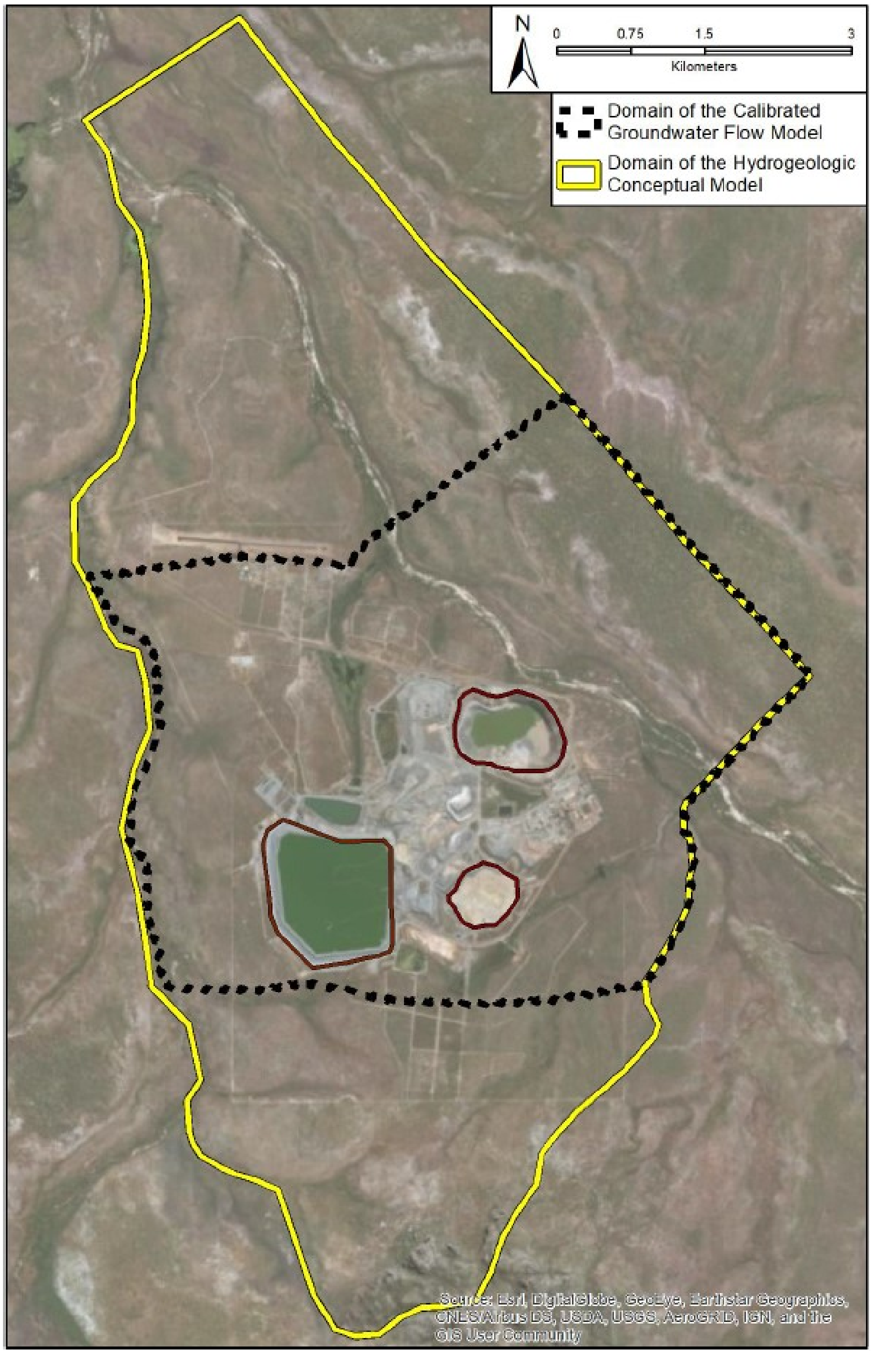


Figure 5-54 Spatial domain of the hydrogeological Ranger Mine conceptual model relative to the domain of the calibrated groundwater flow model.

Updates to the conceptual model focused on extending and improving the HLUs and hydrogeologic framework as well as determining site-specific estimates of recharge and ET. HLUs are hydrogeologic units or volumes defined on the basis of similar geologic and groundwater flow and transport characteristics. All material in which groundwater flows is assigned to an HLU and the HLUs are the building blocks for the material components of the groundwater flow model. The extensive data sets from bores, geologic mapping, and hydraulic testing were used to modify existing HLUs and add new HLUs (Table 5-24). New estimates of recharge and ET were calculated using observed seasonal changes in groundwater heads at shallow bores distributed across the Ranger minesite.

Table 5-24 Summary of differences in name/geometry between the updated HLUs and previous HLUs in INTERA (2014a, b, c; 2016)

Updated HLU	**Corresponding Previous HLU**	**Difference in Name/Geometry**
Shallow HLUs		
Magela Creek sediments	Magela Creek sediments near ancestral sands/other Magela Creek sediments	combined into a single HLU; larger extent to HCM boundaries; slight modifications to width in some areas; no change to thickness
other creek sediments	other creek sediments	addition of sediments for Djalkmarra, Coonjimba and Gulungul creeks; larger extent to HCM boundaries; slight modifications to width of Corridor Creek and its tributary; no change to thickness
Djalkmarra sands	Ancestral Magela Sands	new name; larger extent; no change to thickness
shallow weathered Cahill	shallow weathered rock	larger extent to HCM boundaries; separation of shallow weathered Cahill and shallow weathered Nanambu into two different HLUs; no change in thickness
deep weathered Cahill	deep weathered rock	weathered rock/fresh bedrock contact totally revised; larger extent to HCM boundaries; separation of deep weathered Cahill and deep weathered Nanambu into two different HLUs; thickness increased in some areas and decreased in some areas
Zone C weathered carbonate	LMS carbonate between Pit 1 and Pit 3	wider near Pit 3 margin; shorter extent between pits; thicker
Pit 1 permeable zone	Pit 1 permeable zone	similar extent; slightly thinner
depressurised UMS confining unit	NA	new HLU

Updated HLU	Corresponding Previous HLU	Difference in Name/Geometry
shallow weathered Nanambu	shallow weathered rock	larger extent to HCM boundaries; separation of shallow weathered Cahill and shallow weathered Nanambu into two different HLUs; no change in thickness
deep weathered Nanambu	deep weathered rock	weathered rock/fresh bedrock contact totally revised; larger extent to HCM boundaries; separation of deep weathered Cahill and deep weathered Nanambu into two different HLUs; generally thicker
Deep HLUs		
shallow bedrock Cahill	undifferentiated bedrock	larger extent to HCM boundaries; separation of shallow bedrock Cahill and shallow bedrock Nanambu into two different HLUs; thicker
shallow bedrock Nanambu	undifferentiated bedrock	larger extent to HCM boundaries; separation of shallow bedrock Cahill and shallow bedrock Nanambu into two different HLUs; thicker
HWS	HWS	modified HWS/UMS contact; larger extent to HCM boundaries
UMS	UMS	modified HWS/UMS and UMS/LMS contacts; larger extent to HCM boundaries
MBL zone	MBL Zone near Pit 1	new name; larger extent; dips with UMS rather than being flat; thicker
depressurised UMS	UMS carbonate north of Pit 3	new name; larger extent; deeper; thicker
Zone C shallow bedrock	NA	new HLU
LMS	LMS	modified UMS/LMS and LMS/Nanambu contacts; larger extent to HCM boundaries
lower-K DWPZ	DWPZ	subdivision of previous DWPZ; overall DWPZ extent slightly larger
higher-K DWPZ	DWPZ	subdivision of previous DWPZ; overall DWPZ extent slightly larger
Nanambu Complex	Nanambu Complex	modified LMS/Nanambu contact; larger extent to HCM boundaries
Mine Backfill HLUs		
waste rock underfill	Pit 3 underfill	no change
tailings	Pit 1 and Pit 3 tailings	no change

The calibrated groundwater flow model incorporates the major stresses applied to the Ranger Mine groundwater flow system at Pit 1, Pit 3, and the TSF. Mining of Pit 1 and associated pumping of a dewatering bore and mining of Pit 3 caused very large head decreases in the adjacent HLUs over many years. Partial backfilling locally raised the heads in the pits in relatively short times. For more than 37 years, process water storage in the TSF applied a head increase on the footprint of the TSF. These mining activities stressed large volumes of the shallow and deep Ranger Mine groundwater flow systems to a far greater degree and spatial extent than any long-term pump tests. To accommodate all the changes in pit materials and stresses over time, the calibrated flow model is sub-divided into five sequential models: a pre-mining, steady-state model, and four transient models covering the time periods 1980 to 1996, 1997 to 2005, 2006 to 2012, and 2013 to 2017. To enable reasonable calibration model run times, annual stress periods representing water years were used for 33 of the 37 water years simulated. For four water years, monthly stress periods were used to calibrate the model to observed seasonal fluctuations in groundwater heads. Recharge, ET and surface water stages are also included as stresses.

The numerical groundwater flow model was constructed using the MODFLOW-NWT code to encompass the Ranger Mine, all surface water receptors downgradient of the mine, all important areas driving groundwater flow to the receptors from the mine area, and all important HLUs from shallow to deep. The calibrated model covers about 29 km^2 and vertically spans nearly 800 m, making it the largest Ranger Mine groundwater flow model to date. Discretised into 30 m by 30 m grid cells in the horizontal plane and 19 layers, the model grid contains roughly 612,940 active cells. The model simulation period encompasses a pre-mining, steady-state period and the 37-year mining period, which is far longer than in any previous Ranger Mine calibrated flow model.

The groundwater flow model was calibrated by compiling calibration head targets and iteratively using manual and automated methods to adjust model parameters, compare simulated and observed head targets, and calculate calibration statistics. From examination of the available groundwater head data from more than 450 bores, about 100 head targets were estimated for the pre-mining, steady-state calibrated flow model and more than 8,500 head targets were developed for the transient calibrated flow model. A manual or trial-and-error process was used to define, modify, and refine the spatial extents of model zones representing key HLUs. Calibration of zone hydraulic properties for all appropriate HLUs was conducted by coupling PEST software with MODFLOW-NWT. Calibration statistics, hydrographs, and other standard metrics were used to quantify whether the change in zone properties improved the match between observed and simulated heads.

Results from the flow model calibration reveal that the model simulates groundwater flow with small average error relative to measurement errors and captures temporal groundwater head variations. The calibration statistics are provided in Table 5-25 for all HLUs with the exception of HLUs with less than 25 calibration targets due to insufficient data to provide meaningful statistics.

Table 5-25 Calibration statistics for the transient groundwater flow model

HLU(s)	Count	Mean Error (m)	Mean Absolute Error (m)	Root Mean Square Error (m)	Absolute Minimum Residual (m)	Absolute Maximum Residual (m)	Measured Range (m)	RMSE/ Range (%)	MAE/ Range (%)
Model Domain	***8,536***	***-0.02***	***1.42***	***2.11***	***0***	***26.49***	***81.8***	***3***	***2***
Shallow HLUs									
All	5,560	-0.24	1.21	1.73	0	16.27	44.99	4	3
Magela Creek sediments	0								
other creek sediments	0								
Djalkmarra sands	84	0.31	1.28	1.78	0.01	5.97	9.56	19	13
shallow weathered Cahill	184	0.04	0.93	1.35	0.01	5.85	10.35	13	9
deep weathered Cahill	920	-0.15	1.34	2.02	0	16.27	33.82	6	4
Zone C weathered carbonate	144	-0.53	1.68	2.35	0.01	8.39	21.83	11	8
Pit 1 permeable zone	293	-1.38	1.61	1.99	0.02	4.77	7.71	26	21
depressurised UMS confining unit	0								
shallow weathered Nanambu	1,661	0.08	0.81	1.1	0	4.15	27.72	4	3
deep weathered Nanambu	2,274	-0.38	1.4	1.91	0	8.58	25.85	7	5
Deep HLUs									
All	**2,976**	**0.4**	**1.82**	**2.68**	**0**	**26.49**	**81.8**	**3**	**2**
shallow bedrock Cahill	410	-2.06	2.4	2.98	0.01	10.82	23	13	10
shallow bedrock Nanambu	1,473	0.71	1.54	2.19	0	10.25	22.29	10	7
HWS	0								
UMS	0								

HLU(s)	Count	Mean Error (m)	Mean Absolute Error (m)	Root Mean Square Error (m)	Absolute Minimum Residual (m)	Absolute Maximum Residual (m)	Measured Range (m)	RMSE/ Range (%)	MAE/ Range (%)
MBL Zone	844	0.14	1.2	1.55	0	6.31	23.25	7	5
depressurised UMS	196	4.36	5.33	6.55	0.01	26.49	61.65	11	9
Zone C shallow bedrock	43	0.21	1.57	2.46	0.07	7.68	30.31	8	5
LMS	10				0.55	4.03	5.25		
lower-K DWPZ	0								
higher-K DWPZ	0								
Nanambu Complex	0								

Simulated monthly heads at many bores adequately represent observed seasonal head changes in both timing and magnitude and simulated annual average heads at most bores adequately represent year-to-year changes. Scatter plots of simulated versus observed heads depict random scatter about the 1:1 line for both the entire model and most individual HLUs, indicating negligible bias. Overall, the calibration metrics indicate that both the pre-mining, steady-state and transient models are well calibrated to the observed data. Water balance errors are negligible for the pre-mining, steady-state and transient calibrated flow models and the water balances show good agreement with conceptualisation.

Model validation, through comparison of simulated and observed inflows to the Ranger 3 Deeps (R3D) decline over roughly 5 years, reinforces the high level of confidence in the conceptual and calibrated flow models. The calibrated groundwater flow model was updated to include the stress on the groundwater system from the excavation of the R3D decline and was used to simulate inflows into the R3D decline for comparison to observed data from start of excavation in 2013 through August 2017 (end of transient model calibration period). This implementation of the model provided a check on the calibrated hydraulic properties for both shallow and deep HLUs intersected by the decline. Inflow to the decline modelled using the calibrated hydraulic properties yielded a good match to the observed inflows. This simulation of inflows to the R3D decline serves as validation for the calibrated flow model and shows that the model calibration process incorporated both groundwater head and flux data.

A thorough sensitivity analysis was performed on the calibrated model to determine how model predictions varied with changes to model parameter values and boundary conditions. A sensitivity analysis is a widely accepted means of formally describing the change in model outputs (predictions) caused by changes in specific model inputs or groups of inputs (parameters). The sensitivity analysis on the Ranger Mine calibrated flow model first systematically increased and decreased individual model input parameters for hydraulic properties and boundary conditions from their calibrated values whilst all other input parameters remained constant, ran the model and recorded changes in model predictions for the pre-mining, steady-state model and the transient model. The sensitivity analysis also looked at how model predictions were affected by changing the properties of the Ranger Fault used to define the model southern boundary and by changes to the amount of recharge applied to the waste rock stockpiles.

The analysis revealed that the calibrated flow model is sensitive to a sizeable number of model parameters, demonstrating that the site-specific data used to build and calibrate the flow model do constrain the values of the model parameters. The real-world constraints on the parameters effectively decrease the uncertainty in the parameter values, which in turn means there is increased confidence gained through the calibration process. In particular, the sensitivity analysis shows that the calibrated groundwater flow model for the Ranger Mine is sensitive to many of the parameters previously identified to be important for evaluation of post-closure solute loading to receptors. Removing the Ranger Fault as a low-permeability barrier to groundwater flow did not affect the calibration statistics. A large increase in the amount of recharge applied to the waste rock stockpiles also did not affect the calibration statistics.

Development of the post-closure groundwater flow model consisted of modifying the calibrated groundwater flow model to represent backfill, landform conditions, and the time scale of post-closure hydrogeologic conditions. The hydraulic stresses driving groundwater flow during the post-closure period are essentially the same as those in the pre-mining period. For the purpose of this task, and consistent with previous modelling, the stresses driving groundwater flow during the 10,000-year assessment period were represented as steady driving forces based on long-term averages. The steady flow stresses were calculated using the same 37-year historical record that was used to develop the pre-mining, steady-state stresses for the calibrated flow model. The HLU assignments for the post-closure flow model mostly follow those from the calibrated model except where additional backfill materials were included in the pits and where waste rock will be placed to create the final landform.

Simulated shallow and deep groundwater heads demonstrate that the post-closure groundwater flow model is a topographically-driven flow system. Heads are highest where the topography of the final landform waste rock is highest, and groundwater flows from the higher elevation recharge areas to the lower elevation discharge points in the creeks. Vertical groundwater head gradients are also consistent with topographically-drive flow, with downward gradients in topographically higher areas and upward gradients in topographically lower areas.

The Ranger Mine site wide modelling process and conceptual and numerical flow models were examined to determine compliance with the relevant guiding principles from the Australia groundwater modelling guidelines. The examination demonstrated that the Ranger Mine site wide modelling process complies with the guiding principles from the Australian Groundwater Modelling Guidelines. Agreement of the calibrated Ranger Mine groundwater flow model with the applicable guiding principles demonstrates that the planning, conceptualisation, design and construction, calibration and sensitivity analysis, and reporting of the Ranger Mine conceptual and numerical calibrated flow models were completed appropriately and provide the model with a very high level of confidence. The Ranger Mine groundwater calibrated model will me*et a*ll indicators for the Level 3 confidence level (highest confidence level) after completion of the planned peer review by an independent hydrogeologist with modelling experience.

The updated Ranger Conceptual Model report was provided to the SSB. The SSB sought expert advice from Dr Glenn Harrington of Innovative Groundwater Solutions to determine whether the models are fit for purpose and appropriate for informing future interconnected models. The model was found to be a significant improvement over past models and majority of questions or comments identified by the SSB were resolved during consultation process with ERA (SSB 2019). The outstanding concerns relate to development of a formal uncertainty analysis. INTERA has commenced this analysis and it will be detailed in future versions of the MCP and the MTC Pit 3 closure application.

Further to the review undertaken by Dr Harrington and the SSB, ERA commissioned Brian Barnett, one of the key authors of the Australian Groundwater Modelling Guidelines (Barnett *et al*. 2012), to undertake an independent technical review of the Ranger Conceptual Model to ensure compliance and consistency with the Australian Groundwater Modelling Guidelines. The Ranger Conceptual Model was found to be undertaken in a thorough, considered and professional manner and that the model meets appropriate industry standards (Barnett 2019). A number of relatively minor issues were identified, that in the author’s opinion, both individually and cumulatively do not amount to significant or fatal flaws in the work. These issues have all been addressed by INTERA in the final report. Additionally the author concluded that the modelling to date is in line with a fit-for-purpose conclusion provided the additional modelling tasks required to complete the investigation are undertaken in an appropriate manner.

Figure 5-55 is a graphical high-level representation of the various models developed and used to demonstrate the transport and fate of contaminants within the context of the whole of site conceptual model. The figure also shows the links between the whole of site conceptual model and the various numerical models developed to date.

Ranger Mine conceptual and model solute transport areas of interest/concern

Individual mine workings or features are areas of interest/concern for COPC sources and migration within and from the Ranger Minesite. These include Pit 3, Pit 1, the TSF, the processing plant area, LAAs, the existing R3D workings, and the final landform waste rock. Smaller-scale conceptual models were developed for each of these.

Conceptual models for the areas of interest/concern examined the operational and decommissioning period and the post-closure period. Steps for developing the area of interest/concern conceptual models included describing the setting, identifying the source(s) and COPCs, and identifying the transport pathways and receptors, including soil, groundwater, and surface water.

COPC sources in the areas of interest/concern can be divided into mine wastes and releases from mining activities. Mine wastes comprise waste rock, tailings, pit tailings flux (PTF), and brine. Waste rock is a potential COPC source for Pit 1, Pit 3, R3D, TSF and the final landform constructed with waste rock. Tailings are a potential COPC source for Pit 1, Pit 3 and the TSF. PTF is a potential source in Pit 1, and brine may be a source for Pit 3. COPC releases from mining activities comprise LAA irrigation and dust release and fluid spills or leaks in the processing plant area.

Conservative and reactive COPCs were evaluated for each of the different conceptual models. These included, for example, magnesium (Mg), uranium (U), manganese (Mn), radium-226 (^{226}Ra), total ammonia as nitrogen (TAN), nitrate as nitrogen (NO_3-N), total phosphorus (total-P) and polonium (^{210}Po), as well as others specific to a few areas of concern/interest.

Mg is a COPC because of its potential toxicity to the Magela Creek biota. Based on the previous ERA work and new calculations presented herein, estimates of Mg loading to Magela Creek were discussed for four areas of concern/interest: Pit 1, Pit 3, R3D, and landform waste rock. For the period 1999 to 2003 and 2005 to 2012, the natural Mg solute loading in Magela Creek upstream of the Ranger Mine varied between 75 and 181 tonnes per year, with an average of 135 tonnes per year, whereas the mine-derived loading varied between 72 and 375 tonnes per year, with an average of 178 tonnes per year. The estimated Mg loadings from the areas of concern/interest were compared to these historical natural and mine-derived Mg loadings, shown in Figure 5-56. Loading from waste rock is the largest potential source, and is discussed below under landform waste rock.

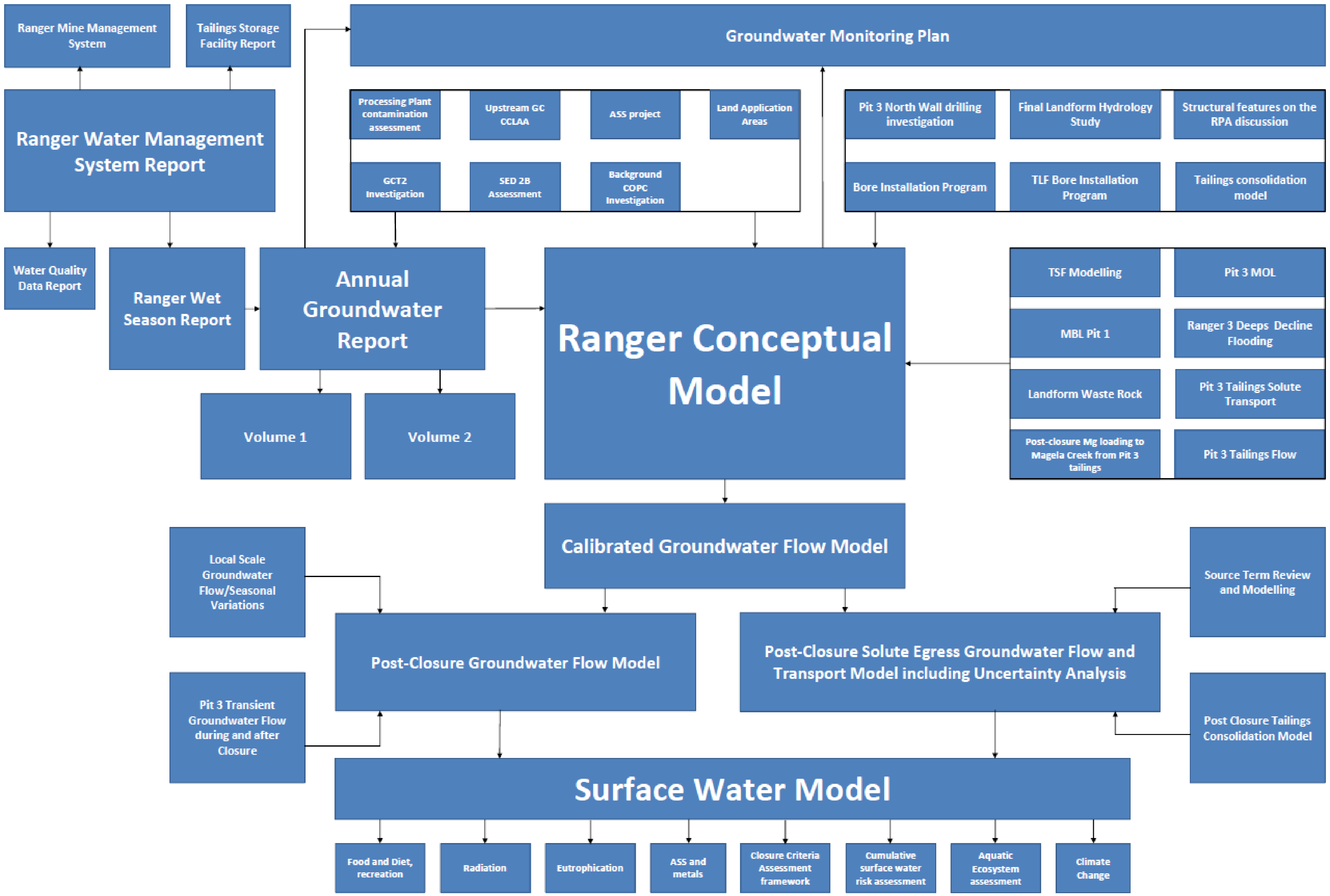


Figure 5-55 Indicative flowchart showing various numerical and solute transport model development for the RPA

Only the TSF, processing plant area and LAAs released COPCs into groundwater, surface water, soil or some combination in the Ranger Mine area during the mining operational and decommissioning period. None of the other areas of interest/concern released COPCs into the Ranger Mine environment during this period. R3D, Pit 1, and Pit 3 act as hydraulic sinks, allowing inward groundwater flow only (Figure 5-57). Evaluations of solute egress during the post-closure period are discussed below for each of these areas of interest/concern.

Discussion in the subsequent sections is based on 2 complementary but discrete packages of work. Discussion on hydrogeological conceptualisations is based on the updated INTERA 2019 Ranger Conceptual Model update as detailed in Section 5.4.3.1 whilst discussion on solute transport and impacts is based on the 2016 Ranger Mine groundwater modelling. Solute transport modelling based on the updated Ranger Conceptual Model is scheduled to commence in early 2020 following completion of a number of supporting models,and will be included in future revisions of the MCP and Pit 3 closure application.

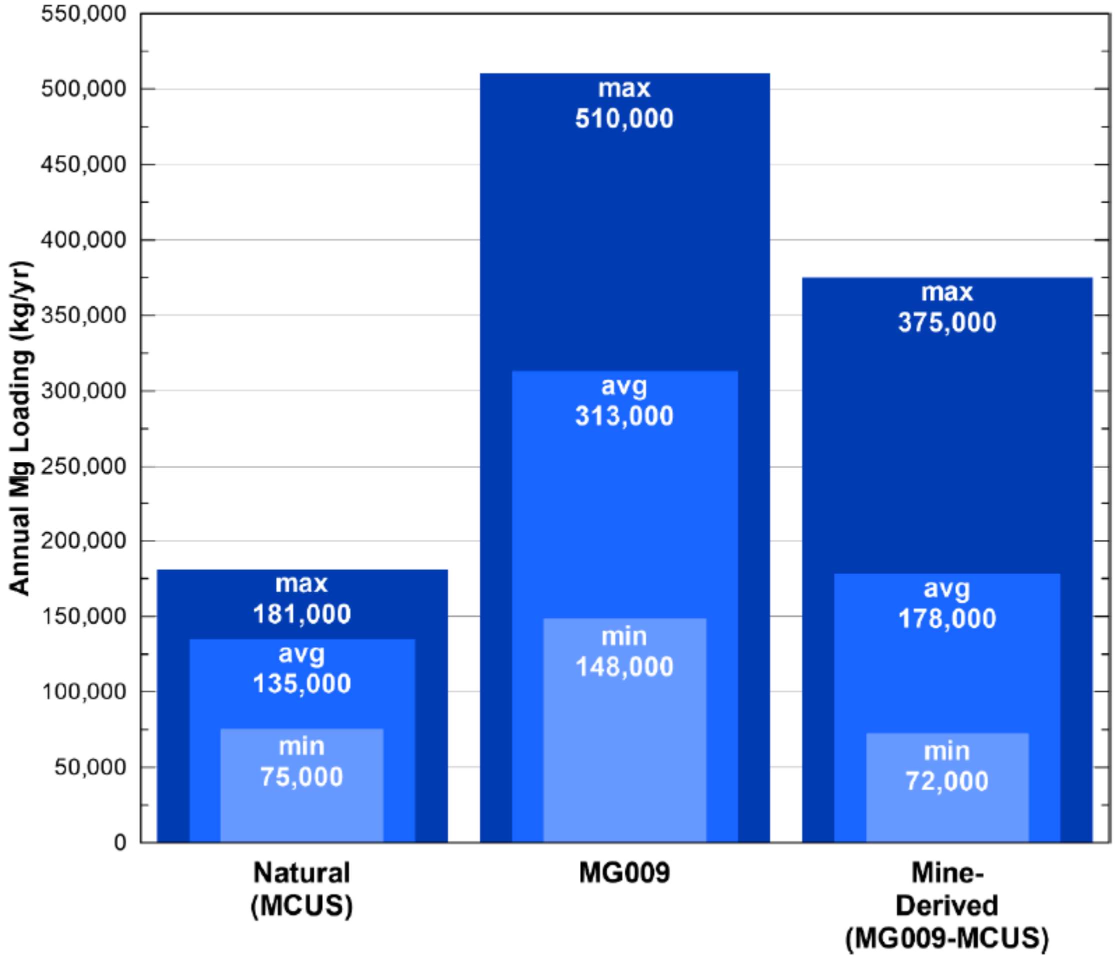


Figure 5-56: Mg solute loads at monitoring stations MCUS and MG009 and derived from the mine (INTERA 2016)

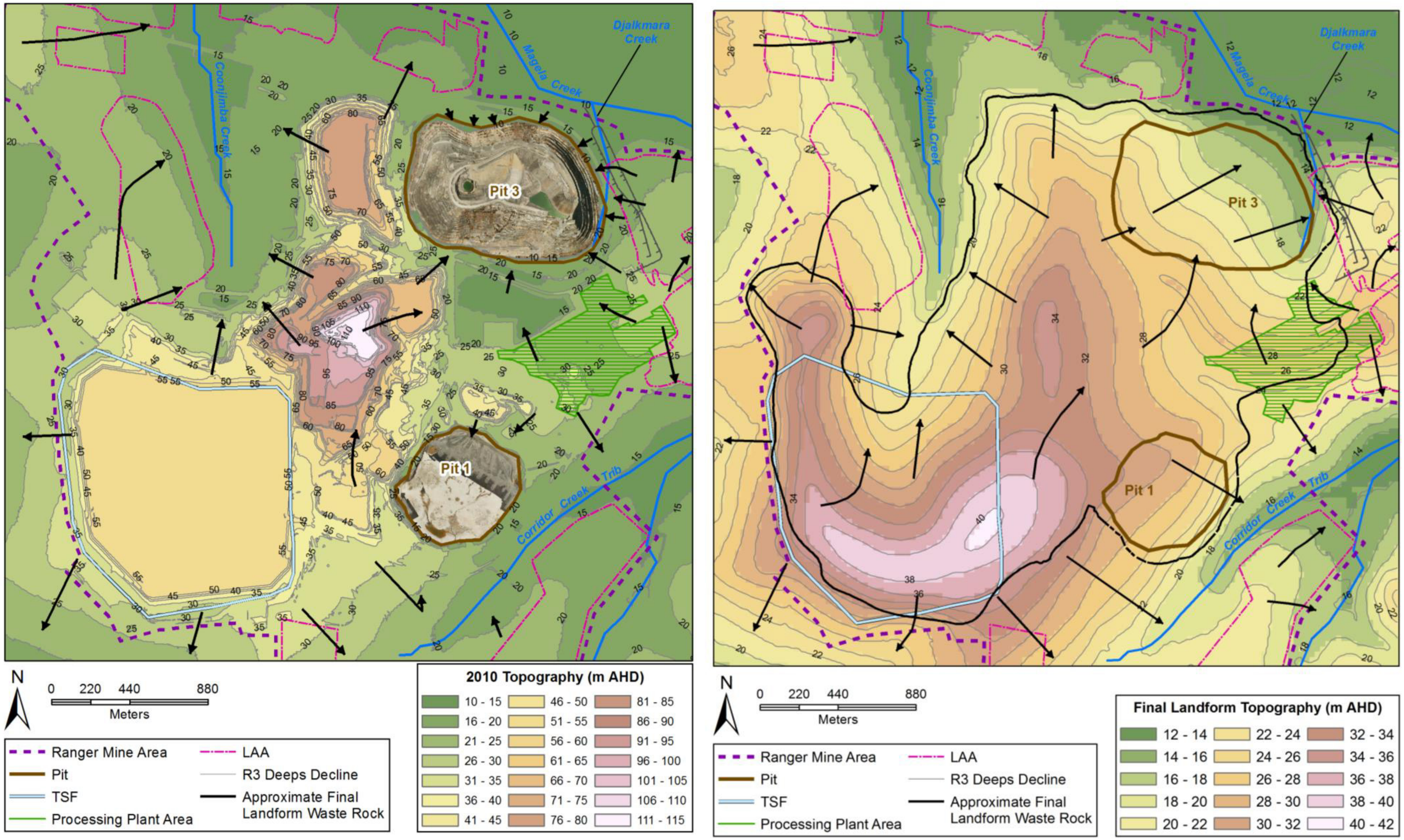


Figure 5-57: Operational groundwater flow (left) compared to post-closure groundwater flow (right) (INTERA 2016)

Ranger Conceptual Model: Pit 3

Located on the Ranger 3 orebody, Pit 3 is the largest mine pit and the nearest to Magela Creek. Conceptual models have been developed for Pit 3 since even before the start of excavation. Except for the sitewide CM by Salama and Foley (1997), each of the other CMs were developed to support modelling of groundwater flow and solute transport.

The key features and processes for pre-mining and during mining for the Pit 3 vicinity include the following:

- Magela Creek is located downgradient of the pit vicinity so groundwater flowed from the pit area to Magela Creek prior to excavation. The minimum distance between the pit and Magela Creek is about 150 m.
- Prior to excavation, the pit outline encompassed both a local topographic high in the west and a local topographic low in the Djalkmarra Creek drainage to the east and south. At the sitewide scale, groundwater flow prior to pit excavation would have been from south to north across the pit vicinity. In the near vicinity of the pit, however, groundwater would flow from the local topographic high north and northeast to Magela Creek, east and southeast to Djalkmarra Creek, and west to Coonjimba Creek. Both the local topographic high and the central portion of the Djalkmarra Creek drainage were replaced by the pit void.
- The pit area straddles the contacts between the LMS, UMS, and HWS hydrolithologic units. Hydraulic conductivity in this area is typically very low (less than or equal to 10-4 m/d), but higher values have been found in shallow weathered rock, the LMS carbonate on the south perimeter of the pit, and the UMS carbonate at the north perimeter or the pit.
- Several faults intersect the pit shell, including the two strands of the Djalkmarra Fault and the Amphibolite Fault. Straddle-packer testing of the strands of the Djalkmarra Fault indicated relatively low hydraulic conductivity of between 10-6 and 10-3 m/d
- Beginning in 2005, more than 400 depressurisation bores were drilled around the perimeter of the pit at depths between the elevations of 8 and -150 m AHD. The purpose of these bores, which had lengths up to 150 m, was to increase pit shell stability by dewatering the surrounding hydrolithologic units.
- Pit dewatering and the depressurisation bores created a hydraulic sink at Pit 3 during the mining period.
- Dewatering of the R3D decline has also led to depressurisation of the deep bedrock hydrolithologic units near Pit 3.
- When open-cut mining was completed in November 2012, the bottom elevation of the deepest part of the pit was about -255 m AHD.

The key features and processes for Pit 3 during and after decommissioning for consideration in groundwater conceptual model include the following:

- Placement of 30 million tonnes of low-grade rock underfill from the bottom of the pit to an elevation of -100 m AHD began in December 2012 and was completed in 2015. An engineered underdrain consisting of a nominal 2-m waste rock layer was constructed at the top of this underfill. The purpose of the underdrain is to remove water expressed downwards by the overlying tailings during consolidation and to remove entrained groundwater displaced upwards from the underfill by the brine injection process

- Deposition of tailings from the milling of ore stockpiles into Pit 3 commenced in 2015 and will cease in January 2021 when ore processing also stops. Transfer of tailings from the TSF by dredge operations began in 2015 and is planned to continue until 2020 at which time the tailings will have reached a maximum elevation of -15 m AHD in Pit 3. By the end of decommissioning in 2026, reduction in the tailings level due to consolidation is expected to reach an average level of -30 m AHD.

- Approximately 2.0E09 litres (L) of brine will be emplaced in the lower 150 m of the Pit 3 underfill up to a final maximum elevation of approximately -118 m AHD. Produced by passing supernatant from the TSF through the brine concentrator, injection of the brine through a bore network into the underfill at elevations between -250 and -210 m AHD began in the 2015 to 2016 time frame. Brine injection is expected to continue through.

- If necessary, tailings consolidation will be enhanced through the installation of wick drains. A rock drainage layer will be installed on top of the tailings to act as an interception layer for removal of expressed tailings water. Following installation of the wick drains and interception layer, and subject to further evaluations, the interception layer may be capped with a low-permeability layer or cap.

- The tailings, drainage layer, and low-permeability cap, if installed, will be covered by waste rock backfill, a second low-permeability cap, and a layer of growth media. The waste rock and growth media will be emplaced to match the final landform design, which moves and truncates the re-created Djalkmarra Creek drainage to the eastern edge of Pit 3 and truncates it

- Until Pit 3 backfilling is completed and the hydraulic heads in the shallow waste rock backfill increase to levels higher than those in the hydrolithologic units located between Pit 3 and Magela Creek, the pit will continue to act as a hydraulic sink preventing groundwater in the waste rock and tailings from flowing away from the pit.

- Once hydraulic heads in Pit 3 increase to levels higher than those in and near Magela Creek, groundwater will begin to flow from the pit, carrying solutes from the backfill into the ancestral Magela sands and weathered and unweathered hydrolithologic units between Pit 3 and Magela Creek.

- Eventually, the Ranger Mine post-closure groundwater flow system in the vicinity of Pit 3 will reach the topographically driven south-to-north flow expected for the final landform. Groundwater from Pit 3 will then discharge into Magela Creek when it is flowing. When flow in Magela Creek ceases, groundwater is expected to continue to flow within the sediments of the creek bed. The rate of solute migration from the pit to the creek will decrease when creek water levels rise more quickly than nearby groundwater hydraulic heads. In the beginning of each wet season, this rapid rise in creek water levels can cause surface water to infiltrate into the subsurface, temporarily minimising solute migration into the creek. This can occur over a relatively large area when the creek flood waters exceed 14 m AHD. Groundwater and solutes will eventually discharge to the creek during the remainder of the wet season, but groundwater discharge cannot significantly affect surface water solute concentrations because the creek flow rate is many orders of magnitude greater than the groundwater discharge rate.

Ranger Conceptual Model: Pit 1

Located on the Ranger 1 orebody east of the TSF, south of Pit 3, and west of the Corridor Creek tributary, Pit 1 was the first of Ranger's two pits. Open cut mining of Pit 1 commenced in May 1980, ceased in December 1994, and produced approximately 19.8 million tonnes of ore. Once the pit was mined out, tailings deposition into the pit commenced in 1996 and ceased in November 2008, yielding an average elevation of 12 m AHD for the tailings surface. Pit 1 served as a process water storage facility until 2012. Backfilling of Pit 1 with non-mineralised waste rock started in 2015 and was completed in 2020. Pit 1 is a likely source of COPCs because it has been used to store process water and tailings during the operations period and will hold tailings and waste rock after closure.

The key features and processes in the pre-mining period and during mining for the Pit 1 vicinity include the following:

- The pit vicinity is located on the western end of the Corridor Creek tributary, which receives managed released water, east of the TSF, and south of Pit 3.

- Prior to excavation, nearly the entire Pit 1 outline fell within the Djalkmarra Creek watershed, with the north-western margin draining toward Coonjimba Creek. The southwest part of the pit outline was a local topographic high. Groundwater would flow from south to north at the sitewide scale, but flow in the pit vicinity was from the topographic highs in the west to lower-elevation discharge areas in the Corridor Creek tributary.

- Like Pit 3, the Pit 1 area straddles the contacts between the LMS, UMS, and HWS hydrolithologic units, but its western margin also includes the Nanambu- LMS contact. Hydraulic conductivity in the pit shell rocks is typically very low (less than 10-4 m/d), with little to no inflow in the bedrock hydrolithologic units because of the large amounts of massive chlorite and chert. In 1984, after 4 yrs of mining, groundwater inflows abruptly increased to an average of about 8 L/s in the southeast margin of the pit between elevations of 0 and 12 m AHD.

- Early interpretations of the inflows in the southeast pit margin devised a new high permeability hydrolithologic unit called the MBL aquifer. Subsequent work by URS (2004) and Anderson *et al*. (2009) indicated that the inflows occurred along a permeable fracture set attributed to a pegmatite intrusion into the HWS rocks along a shallow horizon several tens of metres wide. They also indicated that the surrounding rocks had a lower hydraulic conductivity than that estimated by previous workers for the MBL aquifer. The INTERA (2014b) calibration included a hydrolithologic unit called the MBL zone, which was defined by hydraulic head responses during the calibration period and which had a lower hydraulic conductivity than that previously estimated for the so-called MBL aquifer. The MBL zone extent was further refined in the update to Ranger Conceptual Model in 2019.

- Injection and recovery packer testing of boreholes in the MBL zone near Pit 1 estimated very low hydraulic conductivity values (1E-05 m/d) at depths below 100 m, low values (1E-04 m/d) below 50-m depth, and higher values (1E-02 to 1E-03 m/d) between depths of 43 and 48 m . All the measured hydraulic conductivity values were at least three orders of magnitude lower than those used for the MBL aquifer in earlier models of Pit 1.

- In part, based on the conceptual and numerical modelling from Townley and Associates (2004), ERA constructed a seepage barrier along the south-eastern margin of Pit 1 in 2005 and 2006 to slow solute egress from process water and tailings stored in the pit. The Pit 1 seepage barrier was constructed at an angle that follows the slope of the Pit 1 wall from elevations of 0 to 14 m AHD across a 350-m length and with a design hydraulic conductivity of about 10-3 m/d.

- A single northwest-trending fault has been mapped as intersecting the pit shell at its northern margin, but inflows at that location were small to negligible (Salama and Foley 1997; Kin and Salama 1999; Kalf and Associates 2004; Townley and Associates 2004). Pegmatite intrusions have been mapped at the southeast margin and are associated with the highest observed pit inflows.

- Pit dewatering was aided by intermittent pumping at bore MBL and others from 1987 into late 2005 . Townley and Associates (2004) cite Kalf and Associates (2004) as providing evidence that bore MBL was pumped between 23 and 46 L/s for long periods of time through the end of 2003, but those data were not found in the cited report.

- Pumping at bore MBL was stopped in 2005 because it induced pit supernatant to migrate into the hydrolithologic units on the southeast margin of Pit 1, leading to rapid increases in solute concentrations at nearby bores. From 2006 through 2013, temporarily high pit water levels caused similar increases in solute concentrations at nearby bores on three occasions, but concentrations decreased within a few months.

- Pit dewatering rates after 1984 were estimated to average about 8 L/s.

- Dewatering in the pit created a hydraulic sink at Pit 1 during the mining period.

- When open-cut mining was completed in December 1994, the bottom elevation of the deepest part of the pit was about -150 m AHD.

The key features and processes for Pit 1 during and after decommissioning for consideration in the groundwater conceptual model include the following:

- After an underdrain was constructed, deposition of tailings into Pit 1 commenced in August 1996 and ceased in November 2008. Tailings reached a maximum elevation of 12 m AHD in Pit 1 and are expected to consolidate to an average tailings level of 7 m AHD at the end of decommissioning in 2026.
- Between May and October of 2012, 7,700 prefabricated vertical drains (wicks) were installed within the upper 40 m of the Pit 1 tailings mass to accelerate removal of tailings pore fluids and to promote development of a trafficable surface upon which to commence backfill operations.
- In recent years, waste rock was placed on Pit 1 as a pre-load to assist dewatering by the wicks and tailings consolidation. A layer of laterite was used to cover the waste rock pre-load beginning in 2015 and continuing into 2016.
- The tailings and pre-load will be covered by waste rock backfill to match the final landform design. The uppermost waste rock is intended to serve as growth media for revegetation.
- Until Pit 1 backfilling is completed, and the hydraulic heads in the shallow waste rock backfill increase above the heads along the downgradient pit margin, the pit will continue to act as a hydraulic sink preventing groundwater in the waste rock and tailings from flowing away from the pit.
- The majority of the pit tailings flux will be removed and treated.
- Once heads in Pit 1 increase to levels higher than heads along the downgradient pit margin, groundwater will begin to flow from the pit, carrying solutes from the backfill into weathered and unweathered hydrolithologic units between Pit 1 and the Corridor Creek tributary.
- The seepage barrier constructed along the southeast margin of Pit 1 has a top elevation of about 15 m AHD. The ground surface elevation in this area after decommissioning will be between about 20 to 22 m AHD. Since groundwater heads after closure are predicted to be about 20 m AHD, groundwater will easily flow through the 5-m thick area above the top of the seepage barrier, as well as around the ends of the barrier. Therefore, the seepage barrier and its long-term hydraulic properties will have negligible to no effect on solute release from Pit 1 after closure. The migration rate and loading from the tailings source is primarily controlled by the low hydraulic conductivity of the tailings and the surrounding rock up gradient of the tailings.

- Eventually, the post-closure groundwater flow system in the vicinity of Pit 1 will reach the topographically driven northwest-to-southeast flow expected for the final landform. Groundwater from Pit 1 will then discharge into the Corridor Creek tributary when it is flowing. When flow in the creek tributary ceases, groundwater is expected to continue to flow within the sediments of the creek bed. The rate of solute migration from the pit to the creek will decrease when creek water levels rise more quickly than nearby hydraulic heads. In the beginning of each wet season, this rapid rise in creek water levels can cause surface water to infiltrate into the subsurface, temporarily minimising solute migration into the creek. This can occur over a relatively large area when the creek flood waters exceed 14 m AHD. Groundwater and solutes will discharge to the creek tributary during the remainder of the wet season. Based on the observations that there is negligible base flow to the creek tributary during the dry season under current conditions, there will be negligible groundwater discharge to the creek tributary during the post-decommissioning period.

TSF conceptual model

Multiple studies into the conceptualisation of groundwater movement during the operation of the TSF as well as post closure have been undertaken over the years. Weaver *et al*. (2010) developed a comprehensive CM for the TSF and provided recommendations for additional work that would allow refinement and verification of their model. Golder Associates (2011) sought to implement that CM in a three- dimensional numerical model of solute migration from the TSF. Wakeman and Weaver (2015) provided an assessment of, and CM for, solute migration from the TSF to Gulungul Creek. Weaver (2015) provides assessment of solute migration from the TSF. INTERA (2016) further refined the conceptual model for the post closure TSF and undertook post closure solute transport modelling. The conceptual model has been further updated in 2019 by INTERA and post closure solute transport modelling with uncertainty analysis is currently underway and will be detailed in subsequent MCPs and the MTC Pit 3 closure application.

The key features and processes for the TSF vicinity prior to its construction include the following:

- The TSF footprint straddled a local topographic high that was part of the watersheds for Coonjimba, Gulungul, Djalkmarra, and Corridor creeks. In the original natural drainage, most of the surface water flow from the area covered by the TSF was to the north towards Coonjimba Creek, with the remainder flowing toward Gulungul Creek to the southwest and west, Djalkmarra Creek to the northeast, and Corridor Creek to the southeast.

- The TSF vicinity spans an area where the bedrock consists of granitic gneiss, biotite gneiss, and biotite schist of the Archean-age Nanambu Complex. Fresh (unweathered) Nanambu bedrock is overlain by approximately up to 20 m of highly weathered rock which is in turn overlain by up to 6 m of laterite, soils, and loose material. Minor pegmatites are present in the bedrock. Hydraulic conductivity in this area is typically very low (less than or equal to 10-3 m/d), but higher values are found in the shallow alluvium within the creek tributaries draining the local topographic high.

- Salama and Foley (1997) estimated pre-mining hydraulic heads of about 15 to about 25 m AHD in the vicinity of the TSF. Groundwater flow at the sitewide scale followed sitewide topography from south to north around the TSF vicinity, but within the TSF footprint, groundwater would flow from the local topographic high toward and along the nearest downgradient creek and tributary channels.

- Coffey and Hollingsworth (1979) identified a number of linear features in the TSF footprint that they considered as potential or inferred faults, which are depicted in Salama and Foley (1997). Based on their detailed mapping and logging of these linear features, Coffey and Hollingsworth (1979) determined that most of the potential faults were "healed", which means that minerals had formed to occupy the entire void volume along the feature and left little or no pathways for fluid migration. They also conducted permeability measurements on 2- to 3-m- long intervals in bores drilled into most of the features and found that the hydraulic conductivity for all but a few of these intervals was typically low, on the order of 2.0E-3 m/d. The few exceptions were several shallow intervals with hydraulic conductivity values on the order of 10-1 m/d and two shallow intervals in the Coonjimba drainage with high values on the order of 9 m/d similar to that expected for alluvium. However, all of the deeper intervals in the Coonjimba drainage had hydraulic conductivity values that were orders of magnitude lower than the two shallow intervals, reaching about 10-3 m/d or lower. Hydraulic conductivity values for intact Nanambu bedrock were very low (<10-3 m/d) for nearly all intervals.

- A recent evaluation by Weaver *et al*. (2010) of the linear features identified by Coffey and Hollingsworth (1979) stated that there was little to no evidence that the inferred faults act as more permeable pathways than bedrock for solute transport, with the possible exception of the feature mapped as striking north from the TSF toward the Coonjimba drainage. Weaver *et al*. (2010) called this "the feature referred to as Fault 2A" as they had no evidence that it was a fault.

The key features and processes for the TSF during mine operations include the following:

- Surficial materials were scraped away down to the top of the weathered bedrock to provide a firm foundation for the footings of the TSF walls, which have a compacted clay core keyed into the weathered bedrock by an excavated cut-off (Weaver, *et al*. 2010 citing Volk, *et al*. 1980). Within the TSF, only the vegetation was removed.

- Construction of the TSF's seven lifts from 1980 to 2012 raised local elevations by about 25 to 40 m over the original ground surface of about 18 to 34 m AHD Weaver *et al*. (2010), each time increasing the volume of tailings and process water held.

- Available water-level data for bores completed in the early 1980s and located on the perimeter of the TSF indicate that hydraulic heads continually rose at a relatively rapid rate from the time of construction through about 1984 to 1986. Several of the bores with the longest period of record show a sudden increase in hydraulic head in about

1999, but after this time, heads remained fairly stable with seasonal fluctuations. The addition of four lifts and an increase in height of 15.5 m between 1999 and 2012 had little impact on surrounding hydraulic heads suggesting that the effects of the TSF on hydraulic heads reached their maximum in about 1999.

- Recharge through the waste rock forming the TSF walls and hydraulic connection with the TSF are the likely causes for the local rise in hydraulic heads in and around the TSF up through 1999. The addition of four lifts thereafter apparently did not increase recharge and groundwater heads above their 1999 values.
- COPCs have migrated in groundwater away from the TSF. The COPC plumes have migrated farthest along Coonjimba Creek and Gulungul Creek tributaries 1 and 2 located south and west, respectively, of the TSF.

The key features and processes for the TSF during and after decommissioning include the following:

- Dredging and transfer of tailings out of the TSF will reduce the source mass and gradually lower the hydraulic head that is driving COPC migration away from the TSF area.
- Process water will be stored in the TSF following completion of dredging and tailings cleaning activities until water treatment has reduced the process water inventory sufficiently to transfer to a smaller storage facility.
- Reclamation of the TSF walls and re-distribution of the waste rock from the walls and stockpiles to match the final landform will change the recharge rates and likely cause a significant decrease in local hydraulic heads and gradient around the TSF resulting in much lower rates of groundwater flow.
- Groundwater will continue to flow from the TSF footprint toward the nearest tributary and creek channels. Rates of flow will be lower than those during the operations period because the construction and revegetation of the final landform will lead to an increase in ET and a decrease in recharge.
- Groundwater COPCs from the TSF footprint may potentially reach surface water in the nearest downgradient creeks and tributaries through base flow and transport of salts from groundwater exfiltration by overland flow. When surface water flows cease in the dry seasons, groundwater may continue to flow within the sediments of the creek channels.

The impacts to groundwater after site closure from the reclaimed TSF are expected to be less than those observed during the operational period because the majority of the COPC source mass (i.e., tailings and process water) will be removed and the driving force from the hydraulic gradient in the TSF area will be significantly reduced. Under closure conditions, most groundwater flow under the TSF footprint will be toward the north at a lower hydraulic gradient, resulting in slower transport rates, than exist under operational conditions. On the western side of the TSF footprint, groundwater flow will have lower hydraulic gradients, resulting in longer travel times and lower fluxes toward Gulungul Creek (Figure 5-58). The hydraulic gradient to the south will decrease under closure conditions, so that solutes that have already moved south of the TSF will be transported even more slowly (Figure 5-59).

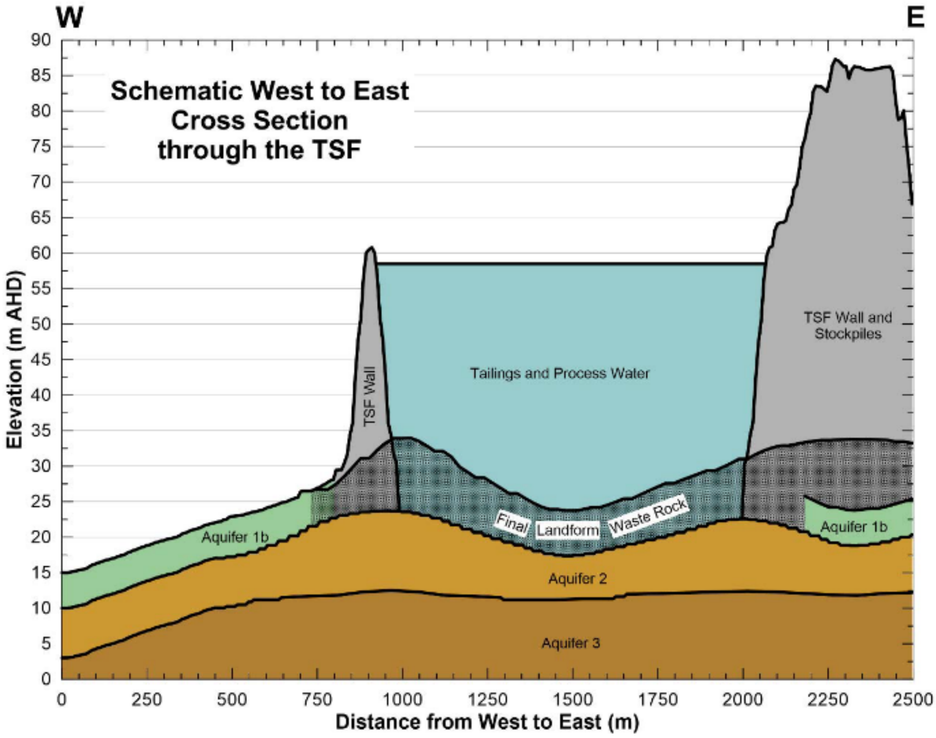


Figure 5-58: Schematic west to east cross-section through the TSF for the current configuration and the final landform waste rock (INTERA 2016)

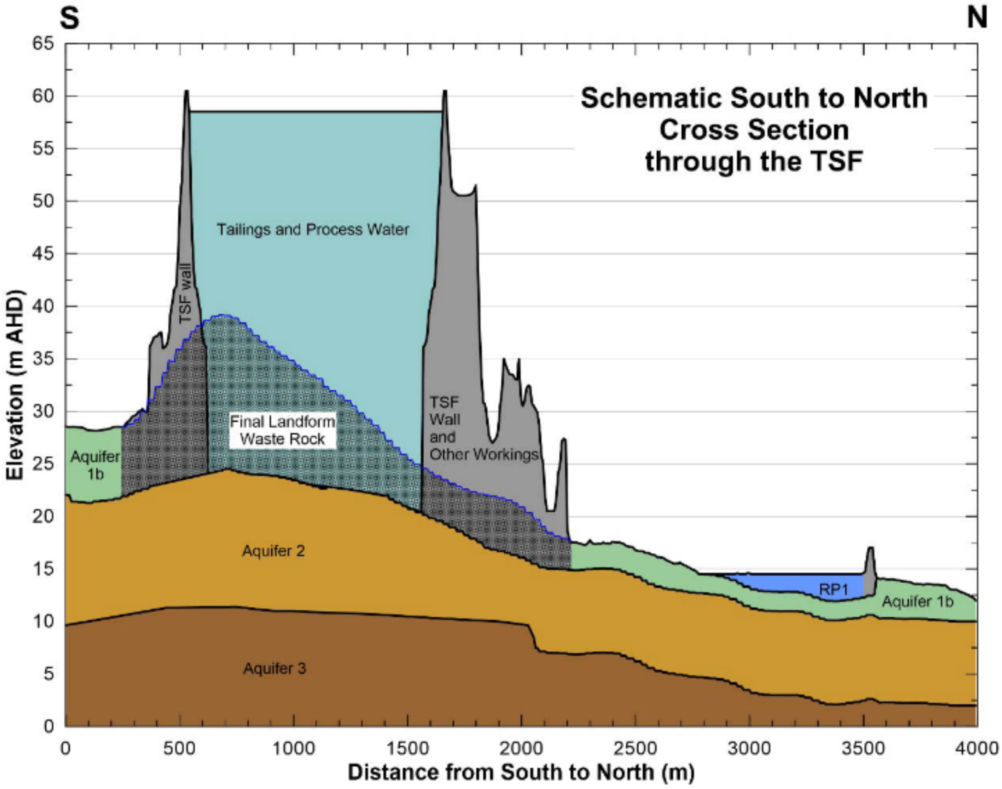


Figure 5-59: Schematic south to north cross-section through the TSF for the current configuration and the final (INTERA 2016)

Processing plant area conceptual model

The source of COPCs in the process plant area and some non-point (areal) sources associated with dust and dispersion from operational activities that have occurred at the site over many years are summarised in Table 5-26. Figure 5-60 shows the groundwater flow pathways from the processing plant area. Contours of long-term average hydraulic head (metres AHD) (white and yellow lines), groundwater divides (red lines), and general groundwater flow directions (large orange, blue, green, and purple arrows) in the vicinity of the processing plant area.

As planned in the closure strategy, shallow contaminated soil in the processing plant area is to be removed during decommissioning. Studies between 2006 and 2009 revealed that groundwater beneath the processing plant area had been affected by magnesium, manganese, sulfate, uranium, and organic contaminants, primarily total petroleum hydrocarbons, released by operational activities (Figure 5-61 to Figure 5-63). Additional investigations into the contamination of groundwater and soils under the process plant area commenced in late 2019 (Section 5.5.2.5).

Table 5-26 Contaminated sites located in or near the processing plant area (INTERA 2016)

Site #	Site name	Area (ha)	Source or nature of contaminant							
			Process water, pond water or tailings	Acid	Hydrocarbons	Chemicals	Metals	Organics and nutrients	Radiation, radioactive dust	Low risk industrial waste
3	Bulk fuel area - diesel storage and pump facility	0.76			Y					
4	Supply waste oil tanks	0.00			Y					
9	Maintenance workshop	1.31		Y	Y		Y			
10	Vehicle refuelling station	0.04			Y					
12	Mine maintenance workshop	0.17			Y	Y				
13	Mine wash down bay	0.15			Y				Y	
15	Acid plant*	1.34		Y						Y
16	Ammonia handling	0.25								Y
18	Emergency dump tank	0.60						Y		
19	Emergency response training facility/ gatehouse	0.09			Y	Y				
20	Fine crushing	2.44			Y				Y	
21	Grinding and pyrolusite	0.82	Y		Y	Y				
22	Hydrogen peroxide tanks	0.02				Y				
23	Laterite plant	2.62	Y			Y			Y	
24	Leaching CCDs clarification	2.14	Y						Y	
25	Lime mill	0.03	Y		Y	Y				
26	Neutralisation	0.27	Y		Y	Y				
27	Pond water holding tanks	0.38	Y		Y					

Site #	Site name	Area (ha)	Source or nature of contaminant							
			Process water, pond water or tailings	Acid	Hydrocarbons	Chemicals	Metals	Organics and nutrients	Radiation, radioactive dust	Low risk industrial waste
28	Precipitation, drying and packing	0.36							Y	
29	Primary crushing	1.12	Y		Y				Y	
30	Product warehouse	0.42							Y	
31	Sand blasting yard	0.35			Y	Y	Y			
32	Sand filters	0.18							Y	
33	Solvent extraction	1.17	Y		Y	Y				
34	Sulfur stockpile	0.77				Y				
35	Power station	1.15			Y					
36	Old sewage trenches	0.14						Y		
40	Demineralisation plant	0.04		Y		Y				
41	Radiometric sorter	1.07							Y	
43	Water treatment plants	0.92	Y			Y				
61	Old core yard	1.61							Y	
63	Plant services	0.13			Y					
66	Brine concentrator	0.93	Y		Y	Y				
67	New sewage trenches	0.28						Y		
69	R3D exploration facilities	0.20			Y	Y				
73	Leach tank failure	1.48		Y		Y			Y	
74	Shellsol underground tanks	0.06			Y					
75	Turbo burning yard	0.05			Y					

* Site 15 (former acid plant) is now the location of the brine concentrator (site 66).

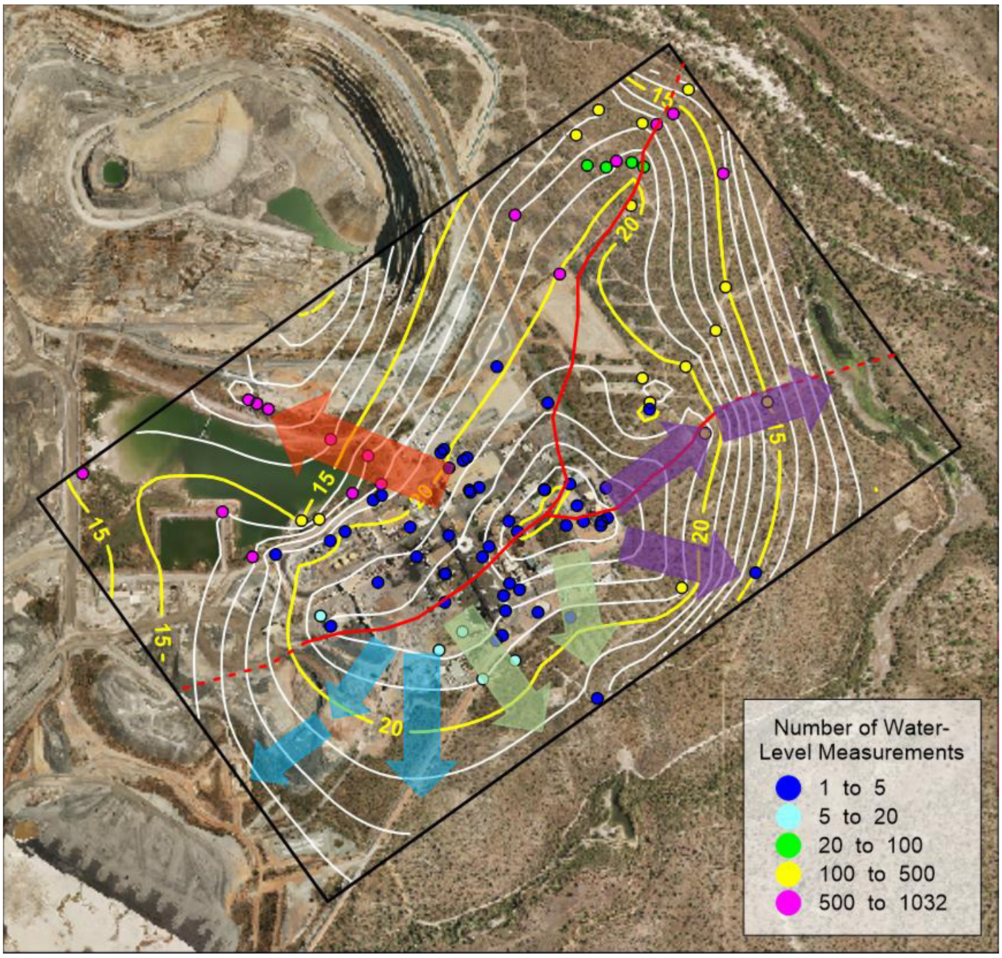


Figure 5-60: Groundwater flow pathways from the processing plant area towards Pit 1, Pit 3, Georgetown Billabong and Corridor Creek tributary (INTERA 2016)

Impacts to groundwater from operational activities appear to be minimal and located in the near vicinity of the processing plant area. During the preparation of this modelling it was noted that there was a lack of recent water quality data throughout much of the processing plant area leaving uncertainty about current groundwater conditions. Reclamation is expected to remove much of the COPC sources in the shallow soil, so groundwater concentrations are expected to decrease over time. Thus, the processing plant area was not expected to be an area of concern for groundwater after mine closure during the preparation of this modelling.

Based on the distance from the affected groundwater beneath the processing plant area to Corridor Creek and GTB and the low COPC concentrations seen in bores adjacent to Corridor Creek and GTB, contaminated runoff and/or groundwater discharge from the processing plant area are not expected to be of significant concern for surface water after closure.

Groundwater monitoring within the processing plant has increased in recent years to support future assessments. The assessment that the Process plant area is not expected to be an area of concern is being reviewed as part of the update to the post closure solute transport modelling and will be detailed in subsequent MCPs and the MTC Pit 3 closure application.

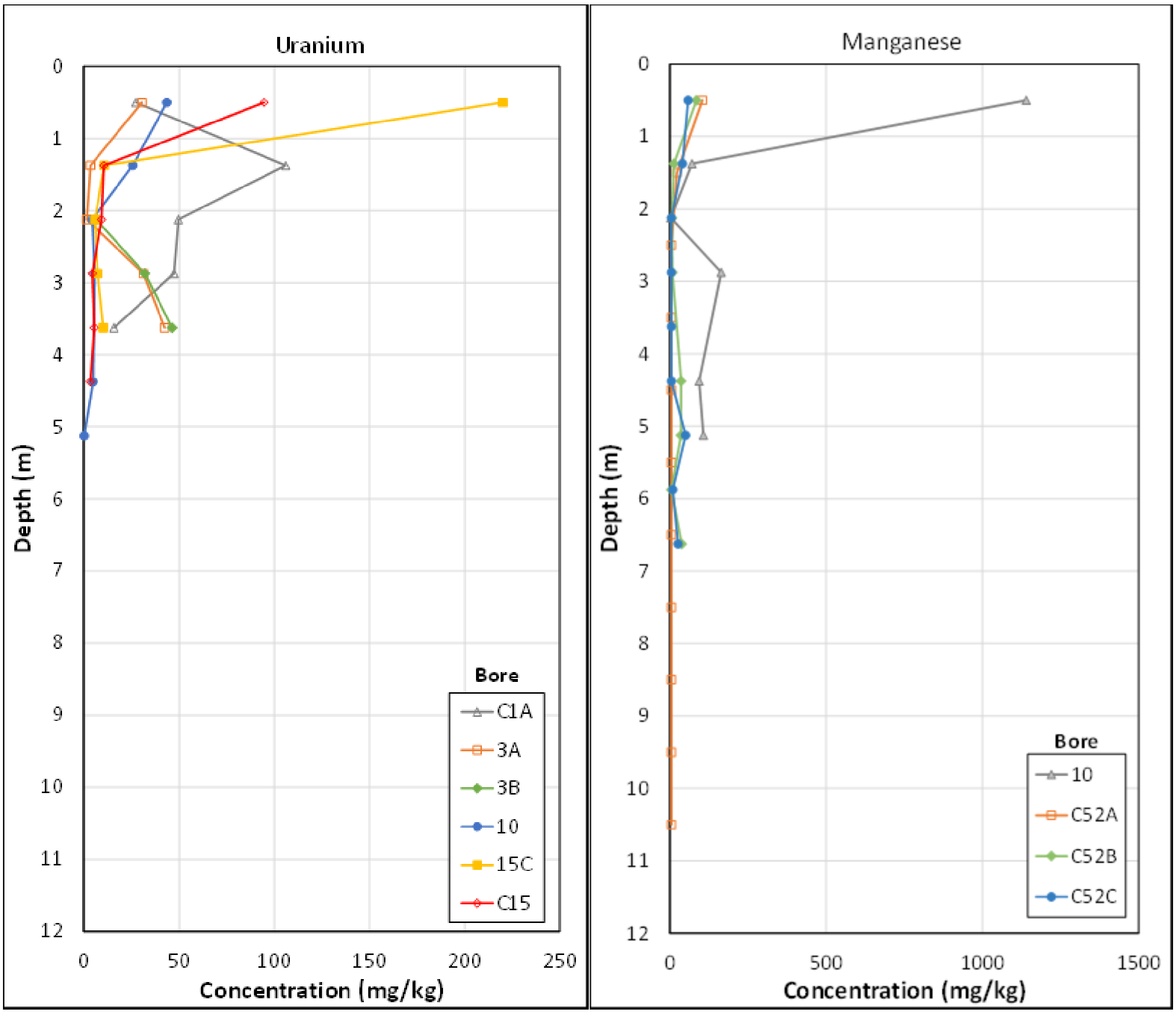


Figure 5-61: Uranium and manganese soil concentration versus depth; generally decreasing over depth (INTERA 2016)

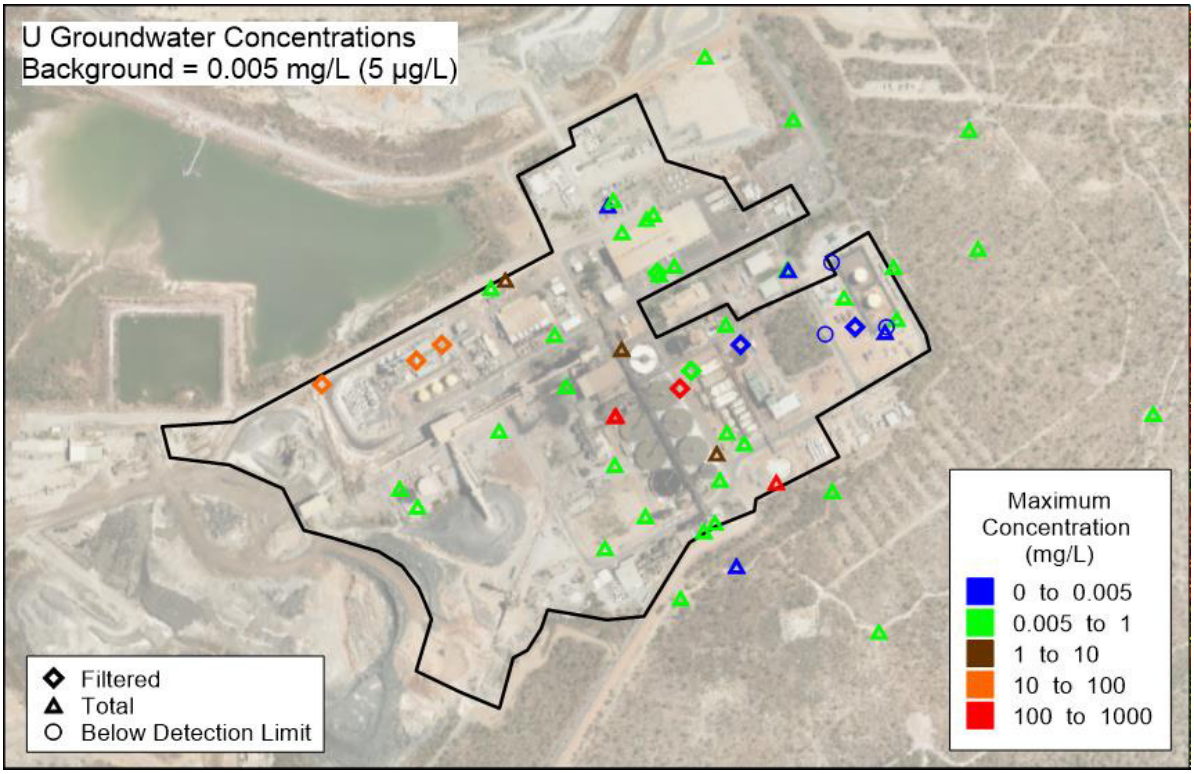


Figure 5-62: Maximum uranium in groundwater (data from 2006 – 2015) (INTERA 2016)

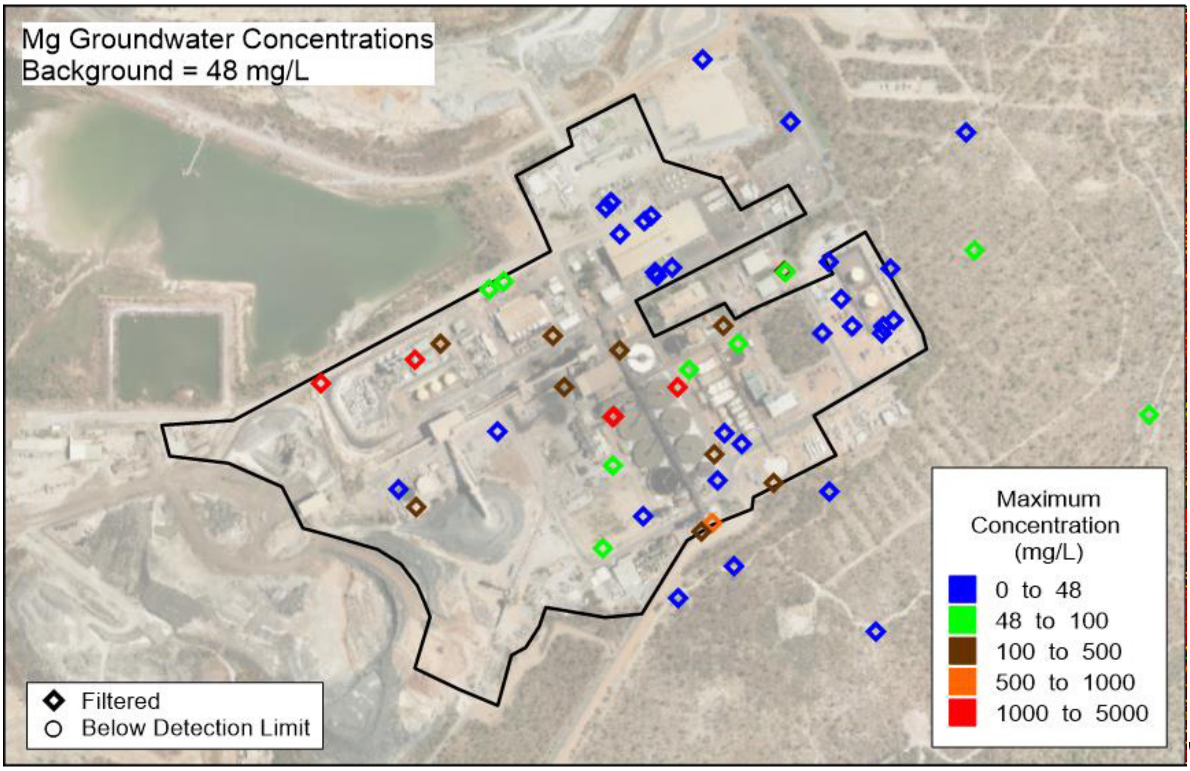


Figure 5-63: Maximum magnesium in groundwater (data from 2006 – 2015) (INTERA 2016)

LAAs conceptual model

The five areas of land application distributed across the Ranger Mine area are the Magela LAA (MLAA) and MLAA extension; the Djalkmarra LAA (east) and Djalkmarra LAA extension (west); the RP1 LAA and RP1 LAA extension; the Jabiru East Land Application Area (JELAA); and the Corridor Creek LAA (Figure 5-64).

As described in Section 5.5.2.4 uranium and radium-226 have been shown to be retained in the shallow soil; however, any future transport into surface water by erosion and runoff would be diluted to very low levels by the large creek flows. Irrigation with the dilute water produced by the treatment plants and natural recharge has been flushing out the conservative COPCs in recent years and will continue to do so prior to closure (Figure 5-64, Figure 5-65 and Figure 5-66). For all LAAs, the groundwater chemistry is expected to show limited to no impacts by the time of site closure.

The remediation of contaminated sites will be assessed and managed in accordance with the closure criteria outlined in Section 8.

The assessment that the LAA area not expected to be an area of concern is being reviewed as part of the update to the post closure solute transport modelling (section 5.5.2.10) and will be detailed in subsequent MCPs and the MTC Pit 3 closure application.

Ranger 3 Deeps conceptual model

Reclamation of the R3D decline and ventilation shaft will require backfilling with cemented aggregate fill and waste rock, which are potential COPC sources. Numerical modelling of COPC migration from closure of the entire proposed R3D mine concluded that solute loading to Magela Creek will be negligible. Therefore, leaching from the much smaller volume of backfill planned for the existing R3D workings (decline and ventilation shaft) will have no impact on the creek. Recovery of hydraulic heads to pre-excavation conditions in the deeper groundwater system will be expected to occur after closure as the hydrogeologic system re-equilibrates. No Long-term impact from depressurisation caused by excavation and dewatering of the exploration decline and shaft is expected.

Further refinement of the R3D conceptualisation was undertaken in 2018 by INTERA to assess the expected hydrological conditions for the R3D decline once the dewatering pumps were turned off and the decline and ventilation shaft were flooded. This is discussed in further detail in Section 5.4.3.9. The further assessment by INTERA in 2018 supports the INTERA 2016 conceptualisation and solute transport modelling.

Figure 5-64 Location of LAAs and associated monitoring bores

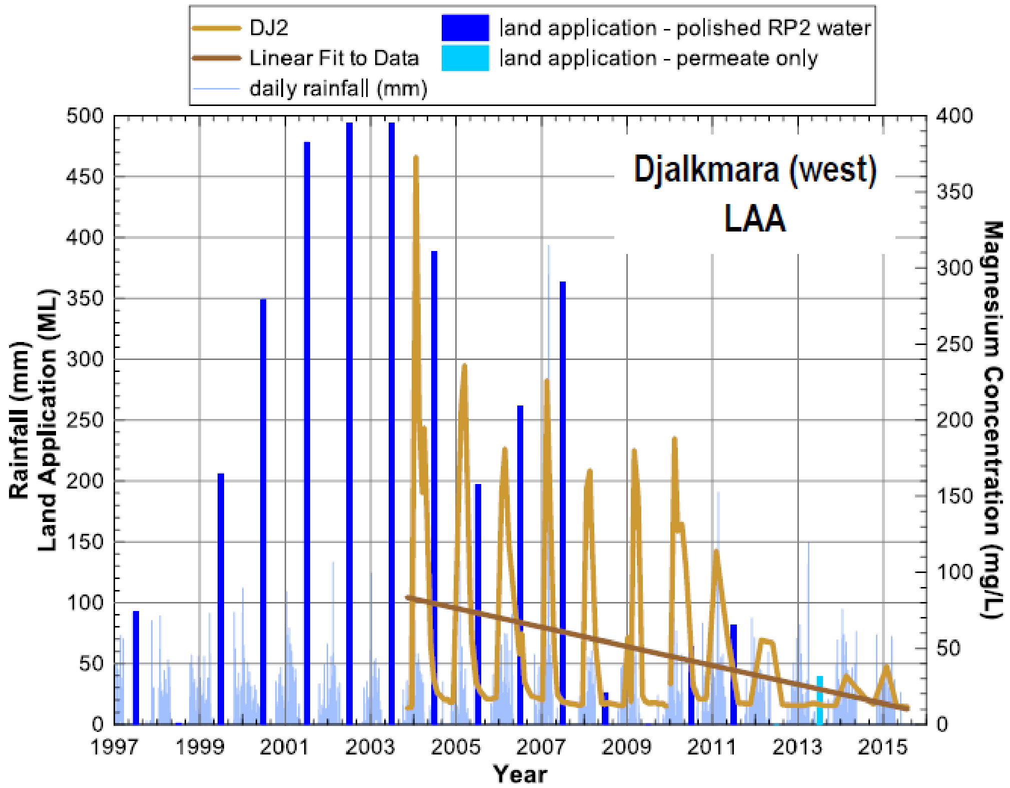


Figure 5-65 Magnesium observed in groundwater where irrigated with pond water (bore DJ2)

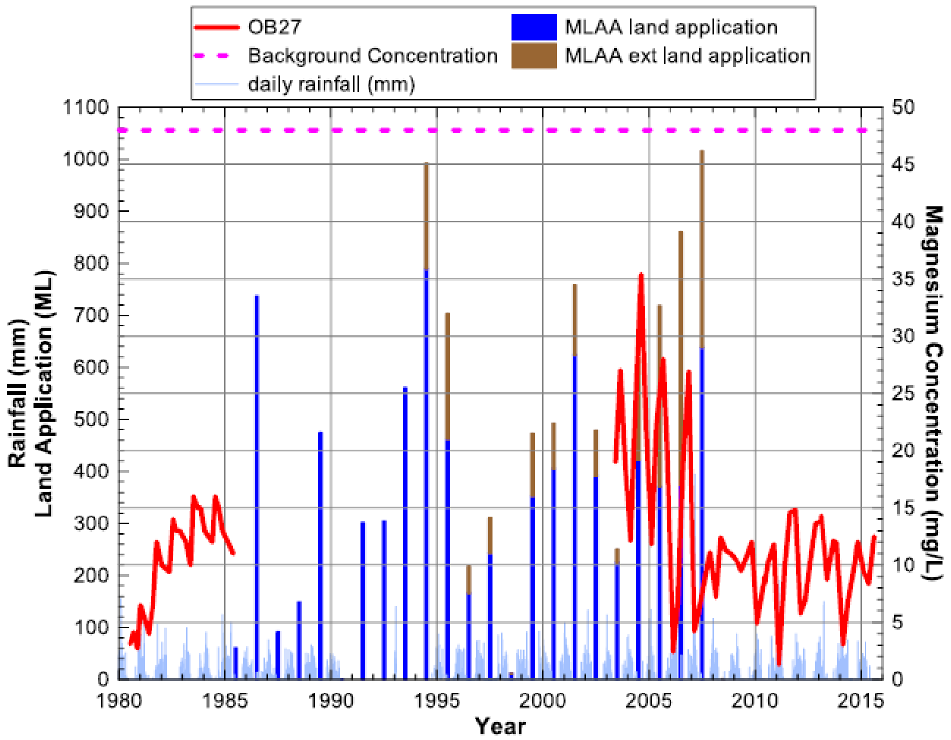


Figure 5-66: Surface water sulfate concentrations in bores OB27, MC27, and MC27 Deep (top) and bores MC12, MC12 Deep, 23562, and 83/1 Deep (bottom)

Landform waste rock conceptual model

Landform waste rock will leach COPCs, with concentrations for runoff much lower than those for groundwater that infiltrates to the water table through the waste rock. COPCs from the landform waste rock will migrate to Coonjimba Creek, Gulungul Creek and Magela Creeks and the Corridor Creek tributary by the runoff transport pathway and the groundwater discharge pathway. Estimated Mg loading from runoff is very small compared to that estimated for the groundwater pathway. Total estimated Mg loading from runoff and groundwater for landform waste rock, including that within the footprint of Pit 1 and Pit 3, is about 78 % of the historical average mine-derived Mg loading for the 1999 to 2012 period and is similar to the natural average Mg loading carried in Magela Creek surface water past the monitoring station upstream of the Ranger Mine for the 1999 to 2012 period.

Additional monitoring bores were drilled in the waste rock stockpiles in late 2018 and early 2019 (Section 5.4.3.11). Samples were waste rock were collected to inform the solute source term and have been analysed, results are currently undergoing review as part of the solute source term update to support the post closure solute transport modelling will be discussed in subsequent MCPs.

Conclusion

The Ranger Conceptual Model describes the elements of the Ranger Mine hydrogeologic and surface water environment that are important to understanding groundwater and surface water flow and solute migration within and out from the Ranger Mine at the appropriate time and space scales. Conceptual models were developed for the regional scale, sitewide scale, and the scale of individual areas of interest/concern where the COPC sources are located. The Ranger Conceptual Model provides a scientific framework based on the available evidence by which ERA can assess and implement decommissioning and closure activities consistent with regulatory environmental controls and rehabilitation requirements.

Updates to the solute transport modelling based on the updated Ranger Mine Conceptual Model are currently underway and will be discussed in subsequent MCPs and detailed in the MTC Pit 3 closure application (Section 5.5.2.9 and 5.5.2.10).

5.4.3.2 Pit 1 solute egress modelling – conclusions

ERA commissioned INTERA to develop a Pit 1 solute egress model to quantify the potential impacts to Corridor Creek for 10,000 years after closure. Potential impacts are defined as the mass loading to Corridor Creek over time of COPCs from the waste rock, tailings, and expressed process water (or PTF) in Pit 1.

Building on the models by CSIRO in 2012 and 2014, INTERA in 2014 and the previous set of conservative conceptual and numerical modelling tools that were designed to evaluate the closure of Pit 3, ERA has developed a comprehensive solute egress model for Pit 1 (INTERA 2016).

Predictions of the shallow Mg plume evolution from Pit 1 waste rock over time revealed that vadose zone leaching causes elevated groundwater concentrations in the western Pit 1 backfill through the first 270 yrs, but concentrations return to background thereafter (INTERA 2014b). Groundwater Mg concentrations from waste rock are much less in the downgradient weathered rock and sediments of the Corridor Creek tributary for the 10,000-yr simulation period, and fall below 60 mg/L after about 300 yrs. The groundwater Mg plume from Pit 1 waste rock reaches the sediments of the Corridor Creek tributary within 25 yrs after the simulation starts, and then continues to move downgradient through those sediments until it equilibrates with the dilute recharge, surface water infiltration, and groundwater discharge. It is important to understand that the groundwater Mg concentrations from Pit 1 waste rock after 300 yrs would not be distinguishable from background groundwater Mg concentrations caused by leaching of the bedrock and weathered rock along and beneath the Corridor Creek tributary.

Compared to the waste rock source, Mg leaching from the Pit 1 tailings source creates a deeper Mg plume in the groundwater between Pit 1 and the Corridor Creek tributary. A dilute portion of the tailings Mg plume (less than 60 mg/L) reaches ground surface at the downgradient margin of Pit 1 and exits as groundwater exfiltration within the first 25 yrs, but the plume does not reach the Corridor Creek tributary until sometime in the next 25 yrs. Groundwater flow drives the subsurface plume downward into the MBL zone and then toward the Corridor Creek tributary.

The pit tailings flux source after 95% removal creates a shallow Mg groundwater plume that migrates out of Pit 1 with much higher concentrations than the Mg plumes from the waste rock backfill and tailings sources. The shallow pit tailing flux Mg groundwater plume reaches ground surface at the downgradient margin of Pit 1 by the second year, reaches the Corridor Creek tributary by 25 yrs, and falls below 60 mg/L at the tributary after 60 yrs.

In summary, modelling of solute transport revealed that COPCs in the Pit 1 waste rock backfill, tailings, and pit tailings flux will likely migrate to the Corridor Creek tributary during the 10,000-yr assessment period. In all cases evaluated, loading from pit tailings flux is expected to only persist for several decades. The peak Mg loading from the combined waste rock, tailings, and pit tailings flux is estimated to be 17,700 kg/yr and to occur at 10 yrs after closure, corresponding to the peak period of higher source strength concentration from the pit tailings flux. The reactive COPCs, comprising U, Mn, ^{226}Ra, TAN, NO_3-N, total-P, and ^{210}Po, will also migrate from Pit 1 to the Corridor Creek tributary, with negligibly small loadings for ^{226}Ra and ^{210}Po.

5.4.3.3 Pit 3 solute transport modelling

INTERA (2014a) developed a numerical modelling of solute transport in groundwater to assess the potential impact of solutes leaching from different backfill scenarios for Pit 3 closure. The modelling specifically focused on quantifying the timing and rates of solutes migrating from the brine and tailings deposited in Pit 3 to Magela Creek (INTERA 2014a). This modelling was further updated by INTERA in 2016 and is undergoing further review and update to support the MTC Pit 3 closure application, details of the updated modelling will be provided in subsequent MCP's and the MTC Pit 3 closure application (Section 5.5.2.10).

Pit 3 will continue to be a hydraulic sink during the decommissioning period, but eventually Ranger's post-closure groundwater flow system in the vicinity of Pit 3 will reach the topographically driven south-to-north flow expected for the final landform. Groundwater and COPCs from Pit 3 will then migrate toward Magela Creek, which is the nearest discharge area.

Together with the brine injected into the underfill, the tailings and waste rock used to backfill will act as sources of COPCs, leaching Mg, U, Mn, ^{226}Ra, TAN, NO3-N, total-P, and ^{210}Po after closure.

Vadose zone waste rock will initially leach Mg, U, Mn, and ^{226}Ra at higher concentrations during about the first 280 yrs after closure, but concentrations will decrease thereafter when the small amounts of pyrite present in the waste rock have been oxidized.

After closure, some groundwater and COPCs will discharge into Magela Creek when it is flowing. As the base flow discharge rate is many orders of magnitude smaller than the surface water flow rate, the mass flux from groundwater is expected to be diluted in the high flow, low concentration creek surface water. Groundwater and COPCs in the Magela Creek sediments are expected to continue to migrate within the sediments of the creek bed throughout the year, eventually discharging to surface water downstream at or before the confluence of Coonjimba Billabong with Magela Creek.

Groundwater and COPCs could be brought to the ground surface on the downgradient margin of Pit 3 by groundwater exfiltration. COPCs may form salts during the dry season that would later be transported to Magela Creek by overland flow during the wet season.

Modelling of solute transport using a number of conservative assumptions estimated the mass of Pit 3 Mg and other COPCs that will be transported into Magela Creek. Loading of Mg to Magela Creek from brine will be negligible, whereas the Mg loading from waste rock will always be much larger than that from tailings. Peak annual Mg loading to Magela Creek surface water from waste rock, tailings, and brine was estimated be about 30,000 kg/yr, which is a small fraction of the average surface water. Long-term Mg loading from the combined sources from Pit 3 is estimated to be even smaller, averaging 13,900 kg/yr. The reactive COPCs, comprising U, Mn, ^{226}Ra, TAN, NO_3-N, total-P, and ^{210}Po, will also migrate from Pit 3 to Magela Creek, with negligibly small loadings for ^{226}Ra and ^{210}Po.

Each of a wide range of analyses investigating uncertainties in the driving force and hydraulic properties and alternative CMs demonstrated that the total Mg loading from Pit 3 is unlikely to be much greater than the estimated peak and long-term loadings.

In conclusion, Pit 3 has been a hydraulic sink during the mine operation period and, therefore, not a source of COPC contamination to groundwater or surface water. Closure conditions for Pit 3 include COPC sources from brine, tailings, and waste rock emplaced in the pit. Numerical modelling indicates these sources will migrate to Magela Creek during the 10,000-yr assessment period.

Reactive transport modelling

Reactive transport modelling was undertaken by INTERA in 2014 to support the solute egress modelling. Results of the reactive transport modelling demonstrated that attenuation of uranium and manganese transport in the relatively conductive ancestral Magela sands would only be effective over times less than about 100 years and attenuation in the weathered rock would be effective over times less than 7,500 years. Results showed that radium-226 does not attenuate in any appreciable manner in either the ancestral Magela sands or the weathered rock.

Solute loadings for U, Mn, Ra-226, TAN, NO_3-N, total-P, and Po-210, from waste rock, tailings, and brine sources were estimated by conservatively assuming no attenuation and scaling the Mg loadings by the ratio of the long-term reactive solute concentrations. The scaling calculations showed that the solute loadings to Magela Creek from the Pit 3 brine reactive solutes will be negligible. Average annual long-term loadings to Magela Creek for uranium is approximately 55 kilograms per year for the combined waste rock and tailings sources. Average annual Mn loadings to Magela Creek from the combined sources is 750 kilograms per year. Mass loadings of radium-226 to Magela Creek from the combined sources are estimated to be roughly 3 milligrams per year (1.1×10^5 milli-becquerels per year). Solute loadings for TAN for the combined sources are 400 kilograms per year. Average annual NO_3-N loadings to Magela Creek from the combined sources is 150 kilograms per year. Solute loadings for total phosphorus for the combined sources are 19 kilograms per year. Loading from polonium was negligible for all simulations with source data.

Secondary uranium and magnesium minerals associated with the waste rock landform

In the solute transport model the source term for COPCs is generated from weathering of waste rock placed in the shells of Pit 3 and Pit 1, and over the post-closure landscape, before solutes egress with groundwater into the receiving environment. The magnesium and uranium source terms were based on empirical data, constrained in the long-term by possible weathering pathways that invoked the formation of secondary carbonates such as hydromagnesite [$Mg_5(CO_3)4(OH)_2 \cdot 4H_2O$] and/or clays such as saponite [$Ca_{0.1}Na_{0.1}Mg_{2.25}Fe^{2+}_{0.75}Si_3AlO_{10}(OH)_2 \cdot 4(H_2O)$] in the variably and permanently groundwater-saturated zones of the waste rock overburden. The transport of Mg and uranium along flow paths through adjacent soil considered possible attenuation through sorption, ion exchange and secondary mineralisation as discussed above in Reactive transport modelling.

In 2016 ERA investigated a Ranger Mine stockpile to identify secondary minerals formed after prolonged burial and exposure to weathering. The aim was to examine whether the secondary minerals assumed by the solute transport model, which immobilise Mg and uranium, are generated during weathering.

Available literature provided a strong knowledge base about source term and secondary mineral generation. It is established that in the variably water saturated zone of the stockpile chlorite [$Mg_5Al_2Si_3O_{10}(OH)_8$] breaks down rapidly in contact with natural rainfall and acids generated by pyrite [FeS_2] oxidation. The source term concentrations for uranium and phosphorus generated by the leaching of chlorite rock in experimental columns (Overall *et al.*

2001) were consistent with the concentrations required to precipitate saleeite [$Mg(UO_2)_2(PO_4)_2 \cdot 10(H_2O)$] as a secondary mineral. Another column experiment representing weathering of tailings containing chlorite in the permanently water-saturated zone (Puhalovich & Pugh 2007), observed that sulfate reducing bacteria mediated the mineralisation of magnesite [$MgCO_3$]. In that experiment the source terms for Mg, observed experimentally as well as in the field, were also consistent with the concentrations required to precipitate magnesite, a mineral related to hydromagnesite. This literature guided interpretation of stockpile weathering.

In the 2016 investigation, ERA collected weathered rocks and exfiltrated groundwater from recently exposed faces of the former core of a stockpile. The rock samples were analysed for secondary minerals, and the groundwater was tested for constituent elements associated with these minerals. A computer model was used to reconcile secondary minerals observed in the stockpile with element concentrations in the groundwater.

The outcome of the investigation was support for the 320 milligrams per litre maximum peak loading for Mg assumed by the INTERA (2014a) solute transport model. The investigation also confirmed several of the main secondary minerals assumed by the INTERA (2014a) model: kaolinite [Al2Si2O5(OH)4], goethite [FeOOH], illite [K0.6 Mg0.25 Al2.3Si3.5O10(OH)2]. palygorskite, a magnesium clay mineral [(Mg,Al)2Si4O10(OH)·4(H2O)] was observed, whilst hydro-magnesite or magnesite were not observed. It is considered that the variably water-saturated groundwater environment of the stockpile represents the future weathering environment of the upper waste rock zone of the final landform, but not the permanently groundwater-saturated lower waste rock zone that will occur in the shells of Pit 3 and Pit 1. This permanently saturated zone should support sulfate reducing bacteria, which is known to facilitate the mineralisation of magnesite (Puhalovich & Pugh 2007). Secondary hydro-magnesite could also form in this water saturated environment.

Some additional secondary uranium minerals were identified in the stockpile (saleeite, torbernite Cu(UO2)2(PO4)2·8-12(H2O)/metatorbenite Cu(UO2)2(PO4)2·8(H2O) and uranophane Ca(UO2)2SiO3(OH)2·5(H2O)). Uraninite (UO2) is likely to form in the permanently groundwater saturated zone. Because these minerals potentially could form additional geochemical sinks for uranium in the final landform that were not included in the solute transport model, this investigation confirms that the solute transport model is conservative for uranium.

ERA is currently reviewing the geochemical source term with respect to predicting the seepage of contaminants from the waste rock final landform and buried tailings. Updates to the waste rock landform source term will be detailed in subsequent MCPs and the MTC Pit 3 closure application. (Section 0)

5.4.3.4 Peer review of solute modelling

A peer review of the INTERA solute egress modelling, including sections on the calibration of the numerical flow model have been undertaken over the past two years by Dr Leslie Smith, Professor at the University of British Colombia, Canada (Smith, 2015, 2016). Dr Smith specialises in the peer review of project work at minesites and hazardous waste management facilities, contaminant plume migration and modelling, seepage analysis at dam sites, fluid flow and solute transport in fractured rock, peer review and performance assessment of low and high-level nuclear waste disposal programs, analysis and modelling of groundwater systems, well field developments, dewatering systems, and review of work plants on site characterisation.

The initial peer review in 2015 was to address feedback raised during the proposed R3D underground mine EIS consultation. The second peer review in 2016, was appended to the Pit 1 notification intended to assess the potential environmental impact of the Pit 1 closure design.

The scope of the initial peer review covered the development of the groundwater flow and solute transport models, calibration of the groundwater flow model, and the application of those models to predict solute loading to Magela Creek expected to occur in a 10,000 year period following closure of the R3D underground workings (Smith 2015). The review specifically considered the groundwater modelling in the context of the Australia Groundwater Modelling Guidelines. Dr Smith concluded in respect to alignment with the Australian groundwater Modelling Guidelines and overall modelling approach used by INTERA:

> *"In my opinion, subject to various observations provided I the body of this report, each of the ten questions listed in Table 9.1 [compliance checklist' can be answered in the affirmative … I consider the hydrogeologic models developed for the evaluation of groundwater impacts associated with the Ranger 3 Deeps Project to be well-suited for their intended purpose."*

The scope of the Pit 1 peer review covered the development of the conceptual models for groundwater flow and solute transport, construction of the simulation model, calibration of the groundwater flow model, and the application of the model to predict COPC loading to Corridor Creek over a 10,000 year period following closure of Pit 1. As in the case of the initial peer review Dr Smith considered the INTERA groundwater modelling in the context of the Australia Groundwater Modelling Guidelines. Dr Smith (2016) concluded in respect to alignment with the Australian groundwater Modelling Guidelines and overall modelling approach used by INTERA:

> *"The Australian groundwater modelling guidelines support a pragmatic approach to modelling and encourage consideration of simple modelling options where they are appropriate. In my opinion, considered in relation to the intended purpose of the model, the three-dimensional hydrogeologic model constructed to aid in the assessment of the closure plan for Pit 1 is based on a reasonable balance between the degree of complexity embedded in the model and the utility of the model. ERA took advantage of a number of approximations and assumptions to achieve acceptable efficiencies in model development, model calibration and model application. One of the principal uses of*

hydrogeological models is their use as a tool to gain site-specific, quantitative insight to the key factors that control the patterns and rates of groundwater flow and, in the case here, factors determining loading of COPC to Corridor Creek. In my view, the model and complementary discussion in the modelling report are used effectively to this end."

The independent review and analysis of the hydrogeologic models developed for the evaluation of groundwater impacts associated with R3D and Pit 1 were considered to be well-suited for their intended purpose.

In addition to the peer review undertaken by Dr Smith, calibration of the 3D groundwater flow model and solute transport modelling from the Pit 3 backfill have been independently (peer) reviewed by Juliette Woods (Principal Groundwater Modeller at South Australia Department of Environment, Water and Natural Resources).

5.4.3.5 Further work

ERA has requested that INTERA updated the existing Ranger Conceptual Model and post-closure solute transport modelling. The update to the Ranger Conceptual model was completed in April 2019 and is detailed in Section 5.4.3.1. Updates to the post-closure solute transport modelling are scheduled to be completed in 2020 following a number of supporting studies. Updates to predictions of post-closure solute transport modelling will be provided to in subsequent MCPs..

5.4.3.6 Assessment of post-closure Mg loading to Magela Creek from Pit 3 tailings

The objective of this modelling study (report 22 March 2019), conducted to support the Pit 3 Tailings Deposition Application, was to estimate peak magnesium (Mg) loading to Magela Creek for each of two Pit 3 tailings deposition options over a 10,000-year time period and to assess the sensitivity of predicted loading to changes in key parameters.

INTERA developed and applied a three-dimensional groundwater flow and transport model for post-closure conditions to estimate the peak loading of Mg to Magela Creek from Pit 3 tailings for the M3D2 and M2D2 deposition options. The model was constructed using the recent Ranger Conceptual Model (RCM) groundwater flow calibration and post-closure flow models. Tailings deposition characteristics were used in the modelling to account for updated tailings source concentrations, volumes and hydraulic properties specific to the M3D2 and M2D2 deposition options. The assessment included a sensitivity analysis that varied hydraulic conductivity (K) of the tailings, K of the excavation damaged zone, and the tailings Mg source concentration.

Peak Mg loading to Magela Creek using the base case model parameters for the M3D2 option is about 4,500 kg/year and that for the M2D2 option is about 8,800 kg/year. These predicted loadings represent about 3 and 7 %, respectively, of the mean historical natural loading of 135,000 kg/year in Magela Creek at station MCUS located upstream of the mine and about 3 to 5 % of the mean historical mine-derived loading of 178,000 kg/year. The estimated number of groundwater pore volumes passed through the tailings in 10,000 years are very small (about 0.8 for the M3D2 option and about 1.6 for the M2D2 option).

The resultant modelling predicted that Mg loadings to Magela Creek from Pit 3 tailings for the M3D2 and M2D2 deposition options and the sensitivity analysis represent a small fraction of the mean natural Mg loading in Magela Creek upstream of the Ranger Mine and of the mean historical mine-derived Mg loading.

5.4.3.7 Evaluation of extent and hydraulic properties of the MBL zone near Ranger Pit 1

The study (report dated 4 January 2018) objective was to undertake an investigation of the MBL zone between the Ranger Mine Pit 1 and Corridor Creek tributary. The objectives of the investigation were to refine the three-dimensional extent, estimate hydraulic conductivity and storage properties, examine how interpreted post-closure solute transport pathways may change as a result of changes to the interpretation, and estimate reduction in groundwater ingress to Pit 1 resulting from abstraction from MB-L bore pumping. The report details the data, methods, models and previous investigations used to re-evaluate the extent and properties of the MBL zone.

Compared to the MBL zone represented in the INTERA (2014a) model, the revised MBL zone extends further to the northeast and southeast, is reduced by about half in thickness, and has an increased hydraulic conductivity. The revised extent and properties for the MBL zone are not expected to change the predicted pathways for solute migration from Pit 1 tailings to the Corridor Creek Tributary. Further review of the impacts to groundwater flux between Pit 1 and Corridor creek as a result of the updated MBL zone conceptualisation is to be undertaken as part of the post-closure groundwater solute transport modelling.

The analysis indicated that the estimated percentage of process water pumped from Pit 1 that was sourced by groundwater ingress from the MBL zone reduced from 40 % in the 2015-2016 water season to 15 % in the 2016-2017 water season. The period during which bore MB-L was pumped corresponded to about half of the 2016-2017 water season and resulted in an estimated 58 % reduction (from 6.2 to 2.6 L/s) in the average rate of MBL zone groundwater ingress into the pit. The water balance analysis confirms that pumping bore MB-L reduces groundwater inflow into Pit 1 from the MBL zone.

The findings and assessments from this study were used to support to the Ranger Conceptual Model update completed in March 2019.

5.4.3.8 Assessment of effect of tailings deposition on flow from Pit 3

The SSB raised concern regarding the environmental effects of the current method of tailings deposition into Pit 3, prompting ERA to request INTERA to assess the effect of tailings deposition and consolidation on the lateral flow of tailings pore water from the pit. Rapid deposition of tailings results in excess pore pressure in the tailings pore fluid. Consolidation of tailings, and coincident reduction in tailings hydraulic conductivity (K), occurs as these excess pore pressures dissipate. INTERA developed two two-dimensional cross-section groundwater flow models to simulate conditions at the end of tailings deposition to assess the flow of this expressed fluid. The cross-section locations were selected to coincide with groundwater flow paths between the pit and Magela Creek.

Both cross-section models showed that tailings pore fluid primarily flows directly into the overlying process water. The remainder flows into the excavation-damaged zone (EDZ) located around the pit or into the underdrain located in the pit between the tailings and underlying underfill. From the underdrain and the EDZ, essentially 100 % of the tailings pore fluid flows along the EDZ and into the process water.

The modelling results demonstrated that:

- there was negligible outflow of tailings pore fluid from Pit 3 or the EDZ into the surrounding formations: almost 100 % of tailings pore waters entering the underdrain and EDZ flows to the process water overlying tailings
- the tailings deposition method currently used by ERA does not pose an environmental threat from lateral flow of tailings pore fluid during the period of tailings deposition.

5.4.3.9 Evaluation of hydrological conditions after halt of pumping in the Ranger 3 Deeps decline

The study (report date 22 March 2018) objective was to assess the expected hydrological conditions for the R3D decline once the dewatering pumps are turned off and the decline and ventilation shaft flood. The following aspects were addressed:

- time taken for water level to rise in the decline to -20 m AHD after pumping has stopped
- pumping rate required to maintain the water level in the decline at -20 m AHD
- time required for the groundwater system to reach equilibrium after pumping stops
- impacts of not grouting the four standpipes located in cuddies along the decline
- approach and value of monitoring the water-level rise in the decline and shaft
- groundwater assessment and conceptualisation after mine closure.

Three-dimensional groundwater modelling was implemented to match inflows to the decline during and since excavation and to predict the water-level rise in the decline after dewatering ceases. Modelling results indicate that the time for the water level in the decline and ventilation shaft to reach -20 m AHD after pumping stops is about 490 days (about 1.3 years). Observed inflows from the base of the weathered zone into the decline range from 0.5 to 1.5 L/s in the dry and wet seasons, respectively, and flows into the ventilation shaft range from 0.5 to 1 L/s in the dry and wet seasons, respectively. Based on these observed data, pumping rates required to maintain the decline water level at -20 m AHD were estimated to range from 1 L/s during the dry season to 2.5 L/s in the wet season. The time required for the decline and shaft to flood above -20 m AHD to near equilibrium water-level conditions at 18 m AHD is estimated to be short (several months) after all pumping ceases and may occur concurrently with the backfilling of waste rock in the decline.

Shallow groundwater heads at the water table are expected to recover to natural conditions within several years after the upper parts of the decline and shaft are backfilled. Groundwater gradients will be downward in the vicinity of the decline portal and the ventilation shaft and, therefore, upward movement of groundwater from four remaining standpipes, if left ungrouted, will not occur. Downward flow along the decline into deeper bedrock units is expected to be negligible and, therefore, installation of bulkheads to further limit this flow is considered unnecessary.

The long-term impact of depressurisation from excavation and dewatering of the exploration decline and shaft on the local groundwater system and Magela Creek will be negligible. Therefore, the R3D decline, and ventilation shaft are not considered a potential area of concern after mine closure.

5.4.3.10 Predictive modelling of Ranger post-closure solute loading with uncertainty analysis

ERA has requested INTERA carry out groundwater modelling to predict transport of COPCs from minesite sources and COPC mass loading to surface waters over the next 10,000 years as a step to demonstrating achievement of environmental outcomes. Inputs to the groundwater flow and solute transport models (i.e., model parameters) will have some uncertainty, as will the model predictions of COPC mass loading to surface water.

A summary excerpt from the scope of work developed by INTERA is provided below. At the time of preparation of this report, works were still underway on the project and results were not available for publishing. Details on the project execution and results will be detailed in subsequent MCPs and the MTC Pit 3 closure application.

This scope to conduct a constrained uncertainty analysis on groundwater COPC loading to surface water receptors was developed using our experience and the scientific literature for uncertainty analysis and groundwater modelling (Freeze *et al*. 1990; Moore and Doherty 2005; Doherty *et al*. 2007; Tonkin and Doherty 2009; Doherty *et al*. 2010; Barnett *et al*. 2012; Anderson *et al*. 2015; Doherty 2015; Watermark Numerical Computing 2019; White 2018). The scope is consistent with and informed by the recent guidance from Middlemis *et al*. (2019) and Middlemis and Peeters (2018) for conducting uncertainty analyses of groundwater models.

The overall objective is to develop probabilistic predictions of solute loading from Ranger Mine sources to Magela, Corridor, Coonjimba, and Gulungul creeks in the 10,000 years following mine closure. Solute loads to the creeks are to be calculated for 20 COPC: magnesium (Mg), uranium, manganese, radium-226, total phosphate, nitrate as nitrogen, total ammonia as nitrogen, polonium-210, iron, copper, lead, cadmium, zinc, chromium, vanadium, calcium, nickel, selenium, aluminium, and sulfate.

INTERA have proposed to incorporate model parameter uncertainty together with calibration data constraints into an uncertainty analysis of COPC loading using a 3-step approach. The steps comprise preparing inputs to the constrained uncertainty analysis, carrying out the uncertainty analysis to predict future COPC loads, and compiling the load predictions for use in assessing potential impacts by ERA.

INTERA's proposed approach adopts the Monte Carlo method to generate equally probable realisations of model inputs and combines it with a framework based on Bayes rule (Bayesian framework) to constrain model inputs using calibration data. In the Monte Carlo method, model inputs are defined as random variables with probabaility distribution functions (PDFs) that are randomly sampled to create a set of equally probable realisations, which, when used in a predictive model, yield a set of model results with which to estimate a PDF of predictions. The Bayesian framework provides the theoretical and operational means to take initial estimates of model parameter PDFs and use other information, such as the observations of groundwater heads used to calibrate the Ranger sitewide groundwater flow model described in INTERA (2019a), to update the PDFs so that their ranges of values yield model results consistent with the other information or observations.

INTERA will predict loads from all or nearly all COPC sources using the null space Monte Carlo (NSMC) method (Tonkin and Doherty 2009; Doherty *et al*. 2010; Navarro Nevada Environmental Services 2010; Doherty 2015). The NSMC uncertainty analysis will be conducted using the three-dimensional numerical groundwater calibration flow model (INTERA 2019a) updated in the previous step together with the three-dimensional numerical groundwater flow and transport predictive models for the sources. INTERA has experience with the NSMC method, having used it to assess uncertainty in plume migration from underground nuclear testing (Navarro Nevada Environmental Services 2010) and more recently in 2018 to estimate post-closure risks from closure of a uranium mine in central New Mexico (INTERA 2018).

The NSMC method provides an efficient means to generate prediction PDFs from posterior parameter PDFs created using the prior parameter PDFs, calibration data set, and the calibration flow model. Random sampling of the prior PDF for each model parameter will produce a large number of sets of prior parameter values, called prior parameter realisations, which will be updated using the PEST null space tool and the PEST calibration tools to create sets of posterior parameter values (Watermark Numerical Computing 2019). These resulting posterior parameter realisations are then run in the predictive model to create COPC loads over time (e.g., horsetail plots like those shown in Figure 2a). This means that both the three-dimensional numerical calibration and predictive models must be run a large number of times. INTERA recently upgraded its Austin computational cluster from 48 to 144 nodes, which should assist in managing the relatively long current model run times and large number of simulations.

Carrying out the NSMC uncertainty analysis process comprises the following tasks, referred to below as NSMC tasks 1 through 7.

1. develop prior PDFs for all input parameters in the calibration and predictive models.
2. review prior PDFs with ERA and stakeholders.
3. construct and test predictive groundwater flow and transport models.
4. generate random sets of parameter values from prior PDFs (i.e., generate the prior parameter realisations).

5. use PEST null space and calibration tools to update prior parameter realisations using the calibration data and calibration model to produce posterior parameter realisations.
6. run the predictive models using the posterior parameter realisations.
7. compile and combine, if necessary, results of predicted COPC loads.

Development of prior PDFs in NSMC task 1 is required for each model parameter. This is a vital step for all model parameters used in the calibration and predictive models.. The roughly 50 input parameters for the calibration flow model include:

- horizontal and vertical hydraulic conductivity for each HLU
- specific yield and specific storage for each HLU
- parameters for boundary conditions representative of the active mining period such as groundwater recharge rates, evapotranspiration (ET) extinction depth and maximum rate, stages for creeks and retention ponds, conductance values for pit drains and creek general head boundaries (GHBs)

Additional input parameters for the predictive models include:

- horizontal and vertical hydraulic conductivity for pit backfill and landform waste rock HLUs
- effective porosity for all HLUs
- boundary condition parameters for the post-closure period including groundwater recharge, ET, and creek and billabong GHBs
- parameters characterising source concentration and leaching rates

Given the numbers of HLUs and boundary conditions and the number of parameters needed for each, INTERA expects that prior PDFs will be needed for roughly 100 to 200 input parameters. The prior PDFs will be described using theoretical distributions derived from the available site-specific data, past model results, and INTERA's expert judgement. Potential theoretical distributions include uniform and normal distributions and their logarithmic transforms. For example, the horizontal and vertical hydraulic conductivity input parameters may be represented as log normal PDFs because their values for a single HLU often span more than an order of magnitude. The means of the prior PDFs are equal to the calibration values for parameters in the calibration solution space and to the estimated means for parameters in the calibration null space. INTERA recommends that ERA and INTERA jointly develop the prior parameter PDFs in NSMC task 1 and then discuss them with stakeholders in NSMC task 2 before proceeding with the uncertainty analysis. These discussions between ERA, INTERA and the SSB commenced in December 2019 and will continue throughout the modelling project.

The predictive models for COPC sources will be constructed and tested in NSMC task 3. At present, INTERA plans to create a single predictive model for all but two sources, called the main predictive model. Model testing will include investigating numerical convergence, representation of each COPC source, suitability of model gridding, and reasonability of model results. A separate variable-density model will be created and tested to predict COPC loading from Pit 3 brine placed in the Pit 3 underfill.

Groundwater flow boundary conditions in the predictive model domain are assumed to be steady. Transport boundary conditions may be steady for some sources and time varying for others. The starting time for the predictive simulations corresponds to the time when groundwater flow is in equilibrium with climatic and surface water conditions; which has been estimated to occur during the first few decades after mine closure. This assumption is important to achieve the objective of developing probabilistic predictions of solute loading from Ranger Mine sources to Magela, Corridor, Coonjimba, and Gulungul Creeks in the 10,000 years following mine closure.

NSMC task 4 will create random samples of model parameter values (realisations) from the prior parameter PDFs created and finalised in NSMC tasks 1 and 2. We propose to use an appropriate random sampling algorithm such as that found in PEST (Watermark Numerical Computing 2019) or similar routines to generate a large number of prior parameter realisations.

NSMC task 5 is the core of the NSMC process and can be a computationally demanding task. The goal is to produce posterior parameter realisations that do calibrate the groundwater flow model. Each prior parameter realization will first be reprojected into the null space using the PEST PNULPAR tool to create the posterior parameter realisations. INTERA plans to run each reprojected realisation in PEST calibration mode with the singular value decomposition PEST tool, which should reduce the run time required (Doherty 2015).

In NSMC task 6, the posterior parameter realisations created in NSMC task 5 will be run in the post-closure predictive models created in NSMC task 3 to produce predictions of COPC loads over time. Results from each predictive model will be similar to one of the curves on the horsetail plot depicted in.

For the last NSMC task, INTERA will examine the horsetail plots for all predictive models over time and combine them into total COPC loads at times of interest. INTERA will also compile the results into the formats needed by ERA to assess potential impacts.

The predicted total COPC loads from groundwater over time cannot be directly compared to an indicator of environmental impact. The predicted COPC loads will be used to assess potential impacts for threshold COPC concentrations in creek surface water through integration with the Ranger Surface Water Model currently undergoing update. The total COPC loads at a chosen probability level for selected times from the groundwater uncertainty analysis would be used as inputs to a surface water model of the creeks.

5.4.3.11 Drilling and installation of monitoring bores in the waste rock stockpiles

During December 2018 and January 2019 ERA undertook a hydrogeological drilling program to drill and construct 9 monitoring bores in various locations through the existing waste rock stockpiles at the Ranger Mine. The objective of the monitoring bores was to support the understanding of source concentrations of COPCs from the waste rock stockpiles to inform groundwater modelling being undertaken by INTERA. (Section 5.5.2.6)

Drilling of the bores was undertaken by J and S Drilling services, with hydrogeological site support provided by INTERA (SP_OB_PL01 through SP_OB_PL03) and Coffey (SP_OB_PL04 through SP_OB_PL09). Following completion of drilling the bores were unable to be developed, a plan to develop the bores is currently being scoped for execution in the 2nd half of 2019.

Groundwater level and quality monitoring of these bores has commenced by the site water management team. Data obtained from monitoring will be used to inform the sitewide groundwater solute transport modelling being undertaken by INTERA for completion in 2020 (Section 5.5.2.10)

5.4.4 Surface water modelling

Over the decades following the creation of the post-mine final landform the site vegetation will mature, and in time the site is expected to largely merge in with the surrounding environment. However the buried tailings and waste rock resulting from the mining process will (with the effect of rainfall, runoff and groundwater movement over the coming millennia) lead to the gradual release of a range of COPCs into the environment. An assessment of the COPC loads likely to be released from the site over the next 10,000 years has been undertaken in a previous study.

The purpose of the surface water modelling is to assist with planning and supporting the approvals required to rehabilitate the minesite by providing estimates of the concentrations of nominated COPCs in receiving surface waters over a period of 10,000 years following the rehabilitation of the mine. The area of interest is the Magela Creek catchment, from the rehabilitated minesite down to Mudginberri Billabong.

A surface water model developed by Williams *et al.* (2013) was previously used to evaluate COPC reporting downstream of the Magela Creek and Gulungul Creek confluence after mine closure. This evaluation applied the surface water model in a PCSWMM model platform, which increased the original model functionality by using an industry standard, GIS compatible, model platform. The original model, developed for an earlier version of the final landform design, was updated to represent the current landform design (V5) and the whole of site conceptual model (INTERA 2016). In 2017 Water Solutions commenced a new, independently developed surface water model to predict the concentrations of COPCs in surface waters of the Magela Creek catchment over the next 10,000 years. The model development was completed in 2020. Further updates are planned to the Water Solutions developed surface water model (Section 5.5.2.11) to include updated solute loadings from groundwater solute transport modelling currently being undertaken by INTERA (Section 5.5.2.10). Results will be detailed in future MCP.

Solute transport modelling (INTERA 2016) has indicated that rainfall entering the waste rock cover will influence solute egress, with 10 percent recharge of the groundwater-shed being from rainfall (INTERA 2016). Furthermore, higher source strength concentrations of COPCs in the waste rock landform predicted to occur between years 50 to 270, and ceasing after year 270, is also expected to influence solute egress. Over the long term (270 to 10,000 years), solute generation will involve groundwater reacting with waste rock, and mixing with slow egress of buried tailings source load some 5,500 years after mine closure. These source terms were predicted by INTERA (2016), and were used in the surface water model. The source terms and solute transport modelling is currently undergoing significant update which when completed will supersede the values and predictions reported in INTERA (2016). Details on this update are provided in Section 0.

The following sections present the surface water modelling development for solute egress modelling from the rehabilitated minesite. The configuration, calibration and simulation of the Ranger Surface Water Model (RWSM) has been undertaken in four major stages.

1. RSWM was configured and calibrated to simulate flow in the study area
2. the RSWM was then configured and calibrated to simulate water quality in the study area
3. the daily site loading time series were developed, based on estimated groundwater discharges to the surface water system, to represent the expected discharge of COPCs from the rehabilitated site over the next 10,000 years.
4. Five scenarios were simulated using the model; a No Mine scenario for reference, and scenarios at the Year 1, Year 20, Year 270 and Year 10,000 time horizons after mine closure. A set of probabilistic statistics have been developed describing flow and COPC concentrations for the 18 modelled COPCs at five key output locations upstream and downstream of the mine on Gulungul and Magela Creeks (GS28, End RPA, GS12, GCLB and GS18) and also including Coonjimba, Georgetown, Gulungul and Mudginberri Billabongs (Figure 1).

ERA is in the process of undertaking further updates to the RSWM. This updated information will be included in the next iteration of the MCP. More information is provided below and current supporting study information in provided in Section 5.5.2.11

5.4.4.1 Flow configuration and calibration

Key characteristic of the flow configuration and calibration of the RSWM are summarised below:

- The study area was subdivided into 15 subcatchments based on the creek network, gauging stations and major points of interest, with the key points of interest and subcatchments in the central part of the model shown on Figure 5-67 Key RSWM study area locations, Water Solutions (2020)
- Daily streamflow estimates were derived from data recorded at five key gauging stations, GS28, GS01, GS09, GS12 and GS18 (Figure 5-67), and used as the key

recorded time series against which the model flows were calibrated. The available periods of record varied from 8 to 47 years, with all recorded data being in the period 1971 to current.

- 129 years of daily rainfall estimates were obtained from the SILO database for each of the 15 sub-catchments, and 129 years of daily evaporation estimates were derived based on recorded American Class A pan evaporation data at the Jabiru Airport weather station.
- Rainfall and evaporation estimates were converted to runoff using the AWBM rainfall runoff model, with low flow losses added to ensure that dry seasons were adequately simulated.
- Reach transmission losses were included to simulate losses from flow as it travels along the creek channels included in the model.
- Channel routing, using the Watershed bounded network model (WBNM) routing methodology, was included to simulate the attenuation of flow as it travels along the modelled creeks.
- Three backwater billabongs (Georgetown, Coonjimba, and Gulungul Billabongs) were included in the model, with the focus on matching their behaviour over the dry season. The backwater billabongs were positioned to accept inflow from their own sub-catchment and backflow from Magela Creek, with a low flow bypass included for low level Magela Creek flows. Storage curves were derived for each billabong based on available survey data, and seepage rates were estimated based on calibration to available level records over the dry season.
- Three first flush channel storages were included in the model upstream of Mudginberri Billabong, to provide a reasonable match to the average timing of first flows into the billabong.
- One named on-line billabong was included in the model, Mudginberri Billabong, at the downstream end of the study area. A storage elevation-volume-area curve was derived for Mudginberri Billabong based on available survey data, and a spillway rating curve was developed based on the rating curve used for GS18. A conceptual groundwater/side storage was included in parallel with Mudginberri Billabong that absorbs a portion of large inflows in the first part of the wet season and provides a better match to the recorded levels over the wet season.

The flow calibration was evaluated using a range of statistics and plots, including annual statistics, average monthly flow plots, daily flow exceedance plots, billabong levels and daily flow plots. Three key plots are shown below to illustrate the calibration achieved: Figure 5-68 shows the mean monthly flows at GS28, on Magela Creek upstream of the mine, demonstrating that the model is matching the typical wet - dry seasonal pattern of flows. Figure 5-69 shows the daily flow exceedance plot at GS09, on Magela Creek next to the mine, demonstrating that the model is providing a good match to recorded flow rates across the flow regime. Figure 5-70 shows the modelled and recorded levels in Mudginberri Billabong (GS18), demonstrating a good match to recorded water levels at the downstream end of the model.

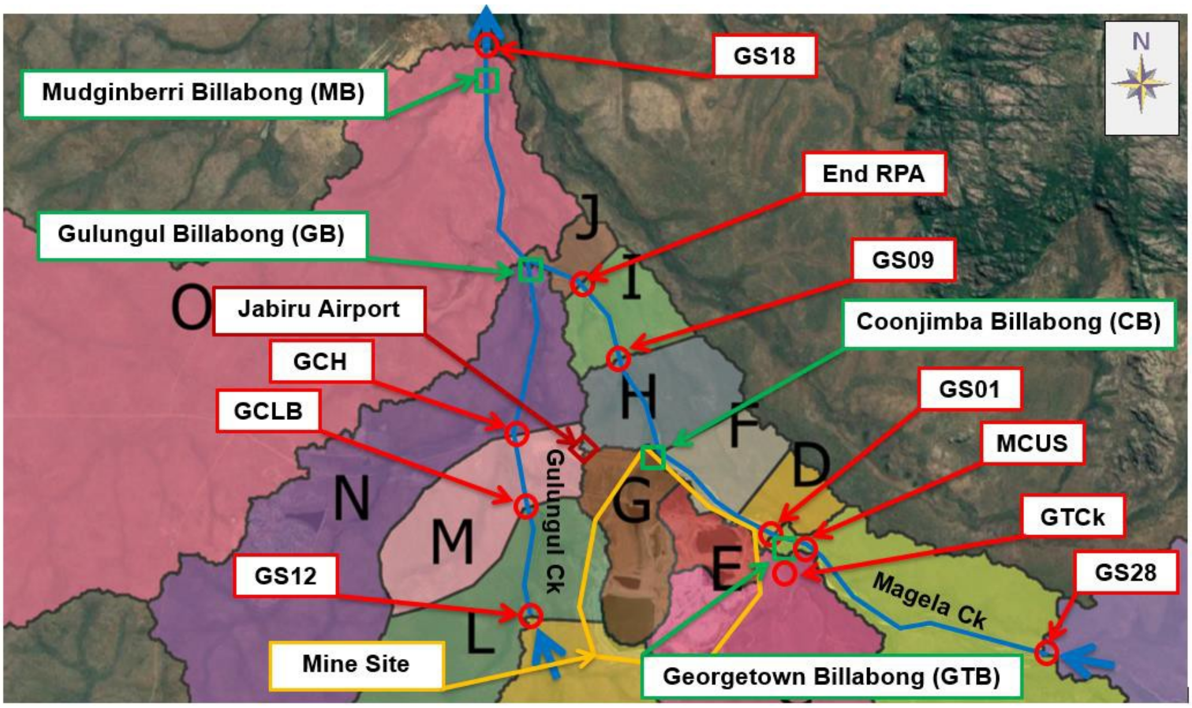


Figure 5-67 Key RSWM study area locations, Water Solutions (2020)

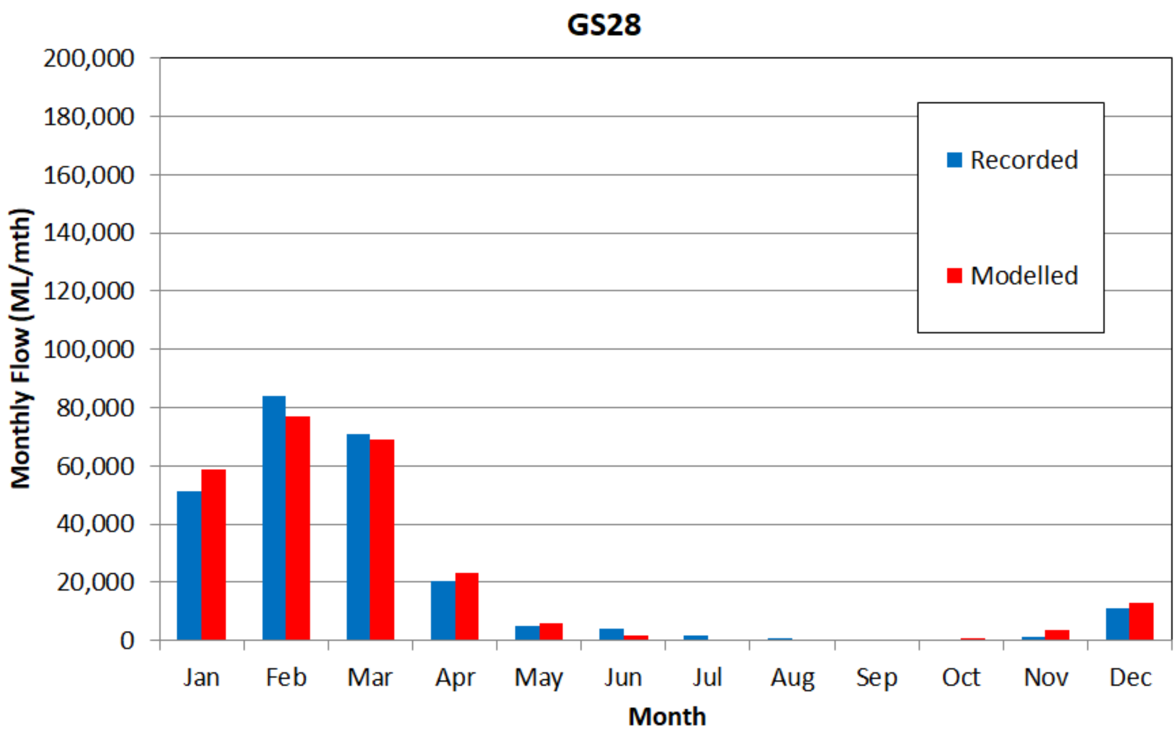


Figure 5-68 RSWM mean monthly flow - GS28, Water Solutions (2020)

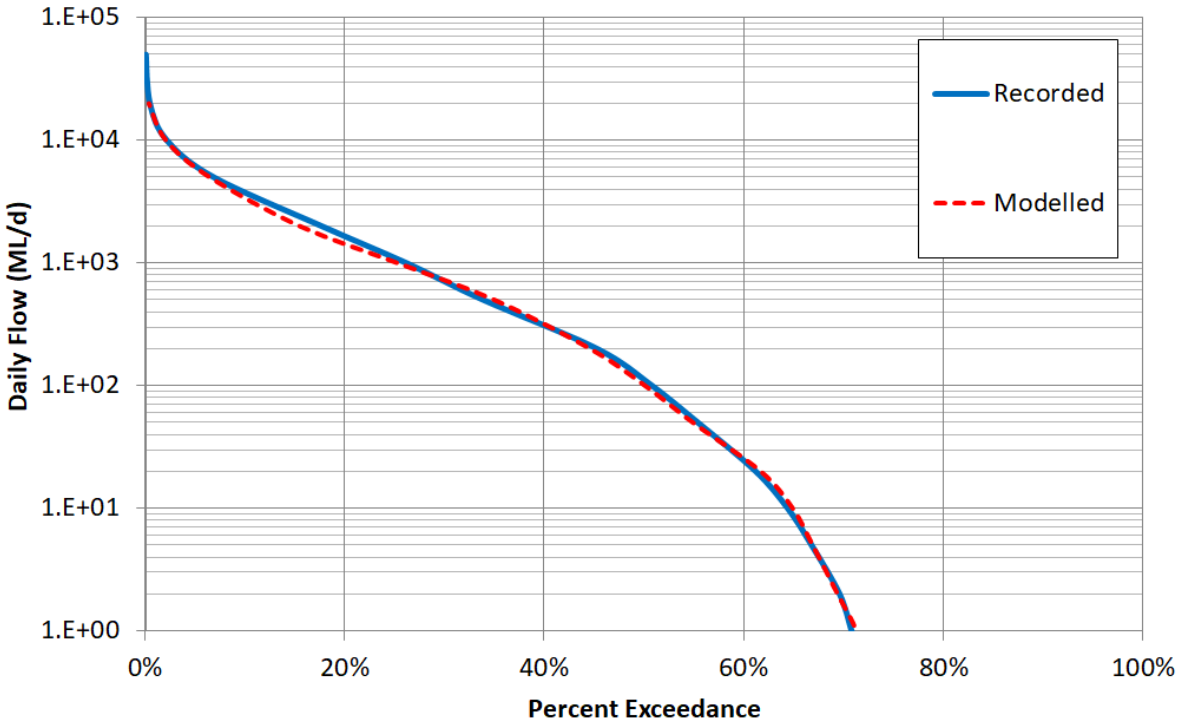


Figure 5-69 RSWM daily flow exceedance - GS09, Water Solutions (2020)

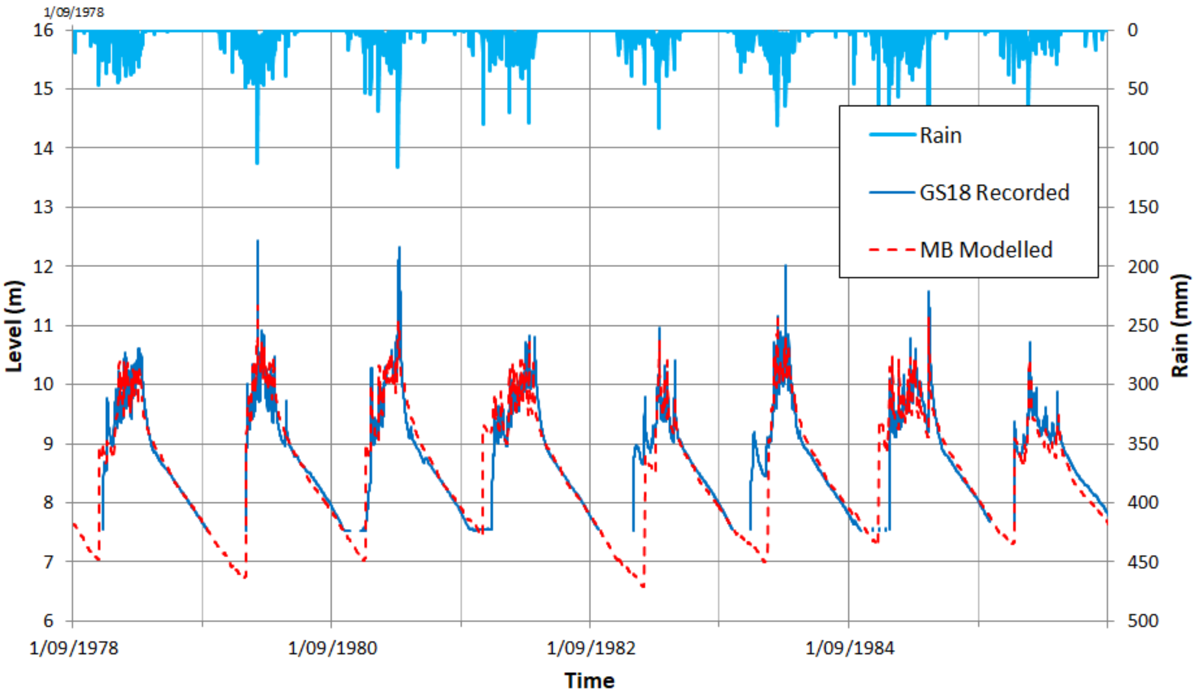


Figure 5-70 RSWM Mudginberri Billabong storage levels, Water Solutions (2020)

5.4.4.2 Water quality configuration and calibration

Key characteristics of the water quality configuration and calibration of the RSWM are summarised below:

18 COPC were modelled, as listed in The last element required in the configuration and calibration of the model was to estimate the 129 year daily time series of TSS loads for the site. TSS loads are expected to peak in Y1 and then settle down to background levels by Y20 with the growth of vegetation and the consolidation of material at the site.

- Table 5-27
- COPCs were assumed to behave conservatively in flow, i.e. conservation of mass applies.
- The derivation of initial estimates of natural catchment loading was based on a review of previous research
- Recorded water quality data were available for 10 locations in the study area, obtained from a range of sources including ERA, the Supervising Scientist and the NT Government. The available periods of record varied from a single recorded point for

some COPCs at some sites, up to many years of data, with all recorded data being in the period 1971 to 2018.

- The model was configured with the initial estimates of natural catchment loads and the results reviewed against the available data. Based on this review a suite of six natural runoff water quality relationships were developed:
  - Flat Concentration,
  - First Flow,
  - First Event,
  - Exhaustion,
  - Flat Load
  - a flow vs concentration rating curve approach.

The developed suite of relationships was applied, singly or in concert, to each COPC iteratively until an adequate calibration was achieved. The resultant relationships and key parameters are summarised in The last element required in the configuration and calibration of the model was to estimate the 129 year daily time series of TSS loads for the site. TSS loads are expected to peak in Y1 and then settle down to background levels by Y20 with the growth of vegetation and the consolidation of material at the site.

- Table 5-27 and Table 5-28.

The recorded data available for the water quality calibration tended to be widely scattered, of varying accuracy, and with extensive data at detection limits, which meant that it was difficult to develop summary statistics or plots without introducing bias. Thus the water quality calibration was conducted based on review of time series plots of modelled and recorded data.

5.4.4.3 Derivation of site loading time series

With the flow and natural water quality processes in the model well established through the flow and water quality calibration summarised above, one further task was required before the model simulations could be run and assessed - To estimate the additional COPC loads likely to come from the rehabilitated minesite over the specified10,000 year period.

Four key time horizons within the 10,000 period were selected, Y1, Y20, Y270, and Y10,000, each representing a period of time when peak delivery of COPCs is expected to be generated by at least one of the rehabilitated mine sources.

Average annual estimates of COPC loads at the four nominated time horizons were derived from previous studies by INTERA for all COPCs except TSS. A summary of the derived total site load for each COPC is provided in the table below (Table 5-29)

These average annual estimates were disaggregated to daily values over the 129 year simulation period using a method based on typical groundwater contributions to the surface water system, based on advice from INTERA. Figure 5-71 below provides a sample of one of the daily site loading traces developed using the determined methodology (for Mg at the Corridor Ck site loading location), and Figure 5-72 provides an appreciation of the annual variation in COPC loading resulting from the developed methodology.

The last element required in the configuration and calibration of the model was to estimate the 129 year daily time series of TSS loads for the site. TSS loads are expected to peak in Y1 and then settle down to background levels by Y20 with the growth of vegetation and the consolidation of material at the site.

Table 5-27 RSWM Natural runoff water quality relationships parameters, Water Solutions (2020)

COPC Description		Relationships Used and Parameters
Name	Symbol	
Magnesium	Mg	• Flat Concentration - 0.2 mg/L • Flat Load - 7.3 g/d/ha
Calcium	Ca	• Flat Concentration - 0.15 mg/L • Flat Load - 5.0 g/d/ha
Nitrate	NO3-N	• Flat Concentration – 3E-3 mg/L • First Flow - 0.197 mg/L
Manganese	Mn	• Flat Concentration – 4.5E-3 mg/L • Exhaustion - 0.01 mg/L, end date 15 January
Uranium	U	• Exhaustion - 4E-5 mg/L, end date 31 August
Ammoniacal Nitrogen	NH3-N (or TAN)	• Flat Concentration – 5E-3 mg/L • First Flow - 1E-3 mg/L
Orthophosphate	PO4-P	• Flat Concentration – 2.5E-3 mg/L • First Flow - 12.5E-3 mg/L
Copper	Cu	• Flat Concentration – 2E-4 mg/L
Lead	Pb	• Flat Concentration – 2.5E-5 mg/L
Cadmium	Cd	• Flat Concentration – 2.5E-5 mg/L
Iron	Fe	• Flat Concentration - 0.1 mg/L • First Flow - 0.18 mg/L
Zinc	Zn	• Flat Concentration – 4E-4 mg/L
Chromium	Cr	• Flat Concentration – 3E-4 mg/L
Vanadium	V	• Flat Concentration – 3.5E-4 mg/L • First Flow - 1E-4 mg/L
Nickel	Ni	• Flat Concentration – 1E-3 mg/L
Radium	Ra226	• Flat Concentration – 60E-12 mg/L • First Event – 120E-12 mg/L
Polonium	Po210	• Flat Concentration - 0.031E-12 mg/L • First Event - 0.037E-12 mg/L
Total Suspended Solids	TSS	• Exhaustion – 1.5 mg/L, end date 31 August • Flow v Concentration, see Table 2

Table 5-28 RSWM TSS flow vs concentration relationship, Water Solutions (2020)

Flow (ML/d/ha)	TSS Concentration (mg/L)
0	0
1.6E1	0
1.6E2	1
1.6E3	5
1.6E4	35
1.6E5	50
1.6E6	50

Table 5-29 Source loads at time horizons - total site loads, Water Solutions (2020)

COPC Description		Average Annual Load (kg/a)			
Name	Symbol	Y1	Y20	Y270	Y10000
Magnesium	Mg	2.74E+04	1.38E+05	1.51E+05	7.12E+04
Calcium	Ca	4.60E+03	2.26E+04	2.72E+04	1.25E+04
Nitrate	NO3-N	1.51E+02	6.82E+02	9.43E+02	4.16E+02
Manganese	Mn	1.08E+03	1.04E+04	4.21E+03	2.99E+03
Uranium	U	1.01E+02	4.58E+02	6.30E+02	2.78E+02
Ammoniacal Nitrogen	NH3-N (or TAN)	5.36E+02	4.14E+03	1.62E+03	1.19E+03
Orthophosphate	PO4-P	1.91E+01	8.72E+01	1.18E+02	5.24E+01
Copper	Cu	1.51E-01	1.07E+00	3.91E-01	3.02E-01
Lead	Pb	8.96E-03	3.16E+00	5.14E-01	6.12E-01
Cadmium	Cd	1.23E-02	8.72E-02	3.19E-02	2.47E-02
Iron	Fe	2.30E+01	4.67E+03	7.73E+02	9.10E+02
Zinc	Zn	7.84E-01	1.34E+01	3.27E+00	3.07E+00
Chromium	Cr	3.70E-02	2.60E-01	9.54E-02	7.37E-02
Vanadium	V	5.04E-03	3.54E-02	1.30E-02	1.00E-02
Nickel	Ni	6.16E-02	6.90E+00	1.18E+00	1.36E+00
Radium	Ra226	4.77E-06	2.30E-05	2.93E-05	1.32E-05
Polonium	Po210	1.18E-10	8.33E-10	3.04E-10	2.35E-10

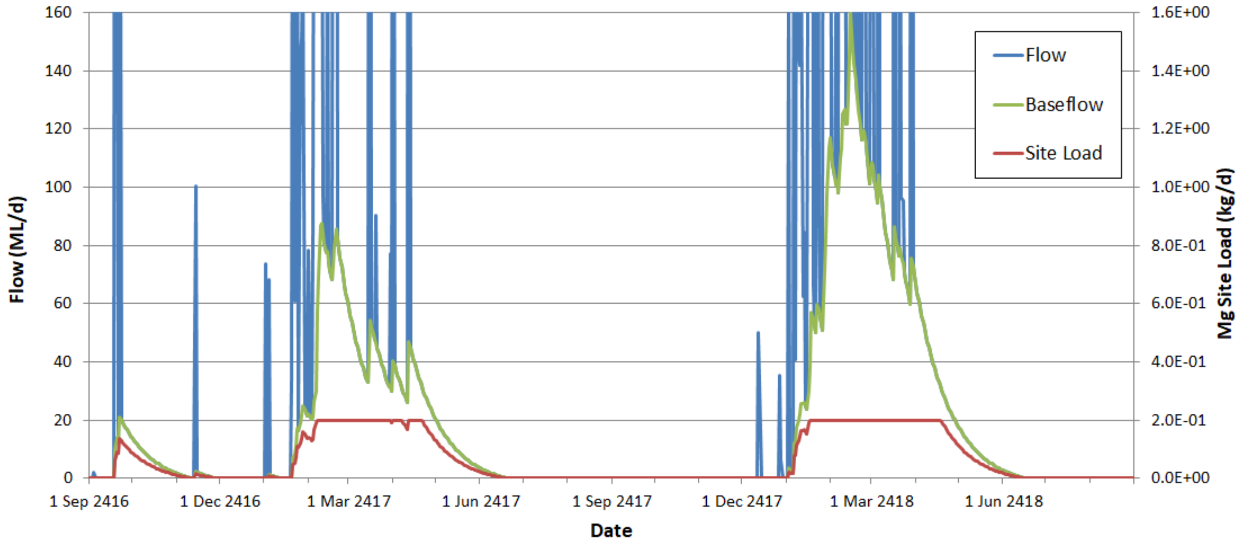


Figure 5-71 Example site loading trace (Corridor Creek - Magnesium), Water Solutions (2020)

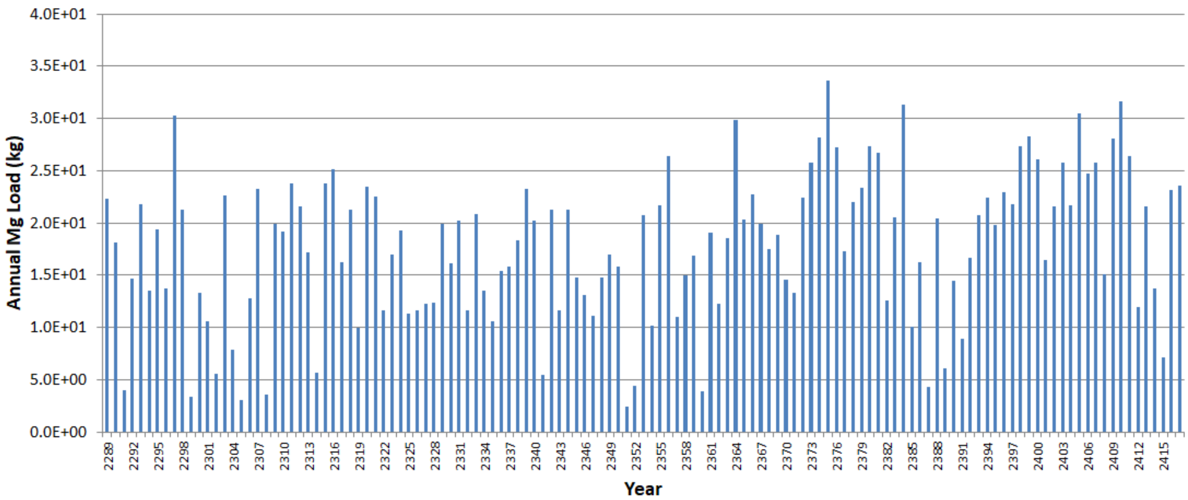


Figure 5-72 Example Annual COPC loading pattern (Corridor Creek - Magnesium), Water Solutions (2020)

Based on suspended sediment data collected from the trial landform at the mine, a Y1 average annual rehabilitated catchment TSS concentration of 120 mg/L was adopted. The derived natural catchment TSS concentration rates were scaled up to match this average annual concentration. Figure 5-73 below provides a sample of the derived TSS site loading concentrations, showing that the estimated rehabilitated site TSS discharge is significantly higher than estimated natural catchment discharge.

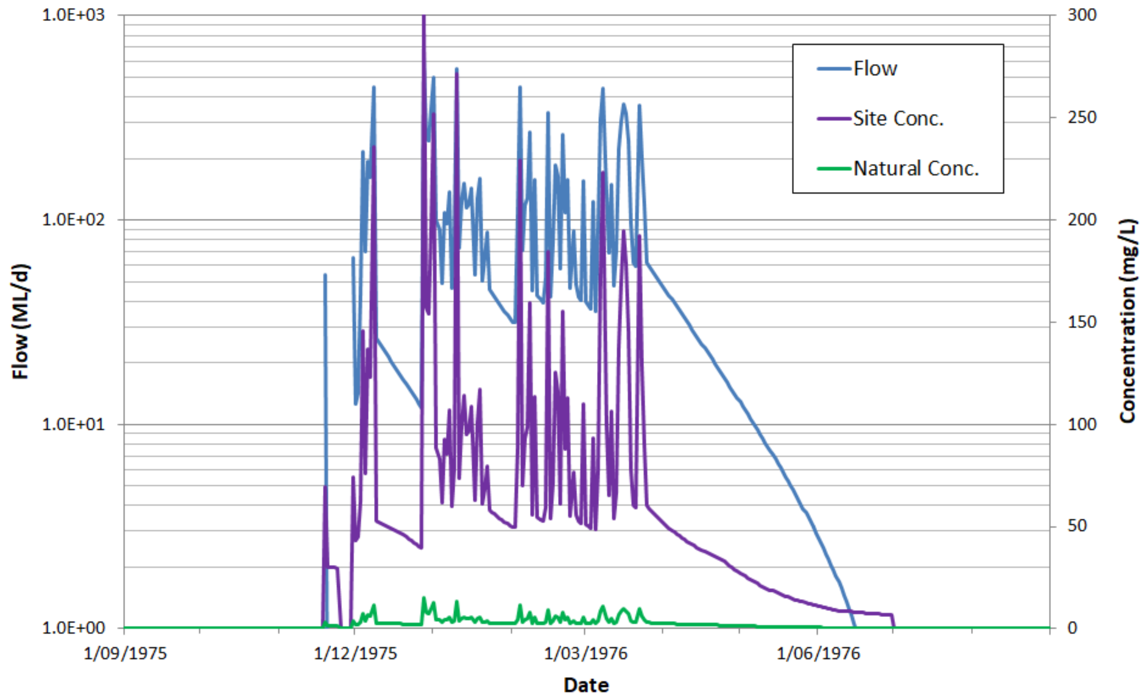


Figure 5-73 Sample site TSS loads, Water Solutions, (2020)

Figure 5-74 below provides an appreciation of the variation in annual TSS loading over the 129 year simulation period that results from the application of the developed methodology. The annual TSS loads vary substantially, with the largest TSS discharge associated with the 2006-7 water year, the year that contains the largest flood on record.

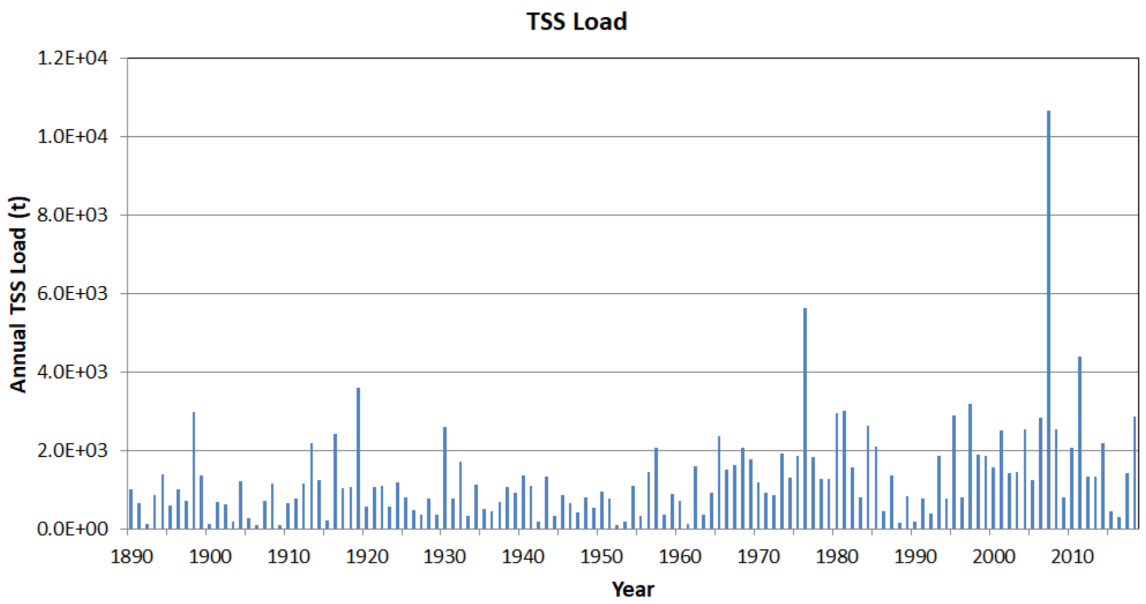


Figure 5-74 Annual TSS loading Pattern, Water Solutions (2020)

5.4.4.4 Simulations

Five scenarios were simulated using the configured and calibrated model. The first modelled scenario is the case used for model calibration, referred to as the 'No Mine' case as it represents just the loads from natural catchment sources, that is, no loads are included from the minesite. (This scenario has been included in the results to assist in understanding the results for the other four scenarios.) The other four scenarios are the selected four time horizons Y1, Y20, Y270 and Y10000.

A standard set of results at five key reporting locations (GS28, GS12, End RPA, GCLB and GS18 (Figure 5-67) has been developed for each scenario in order to provide a concise understanding of the results produced by the model. Other reporting locations include billabongs as per Figure 5-67 This includes statistics on the model flow rates, COPC mass loads and COPC concentrations.

The mean annual flow at each key location in all scenarios is shown in the table below. All five scenarios have the same flows, with the only difference between the five scenarios being the site COPC loads that are applied.

Table 5-30 shows that the mean annual flow increases from GS28 to End RPA and from GS12 to GCLB, reflecting the inflows from the catchments between these locations. However the mean annual flow at GS18 is less than the combined mean annual flow at End RPA and GCLB. This reduction is due to the considerable volume of breakouts and losses in the lower reach of Magela Creek above Mudginberri Billabong. In all, some 39% of the tributary inflows to the model are lost to surface flows in the main channel of Magela Creek, either via seepage, evaporation, breakouts or storage effects in the model.

Table 5-30 Mean annual surface water flow, Water Solutions (2020)

Location	Mean Annual Flow **(ML/a)**
GS28	1.97E+05
End RPA	2.42E+05
GS12	2.09E+04
GCLB	2.79E+04
GS18	2.26E+05

Figure 5-75 shows the mean monthly flows over the 129 simulated years at the five key locations. This figure shows the expected wet – dry season pattern. Monthly flows tend to increase from GS28 to End RPA and from GS12 to GCLB, but flows at GS18 are generally less than the sum of the flows at End RPA and GCLB. A monthly shift can also be observed - flow at GS18 is considerably less that upstream in the early wet season, but is comparatively higher late in the wet season, reflecting the filling up of the various billabongs, bed sands, floodplain stores, etc., allowing more of the upstream flow to make it past Mudginberri Billabong later in the wet season.

Figure 5-76 shows the daily flow exceedance over the 129 simulated years at the five key locations. This figure shows that GS12 and GCLB are fairly similar, being relatively close together, and that End RPA and GS18 are similar, with End RPA being physically located much closer to GS18 than to GS28.

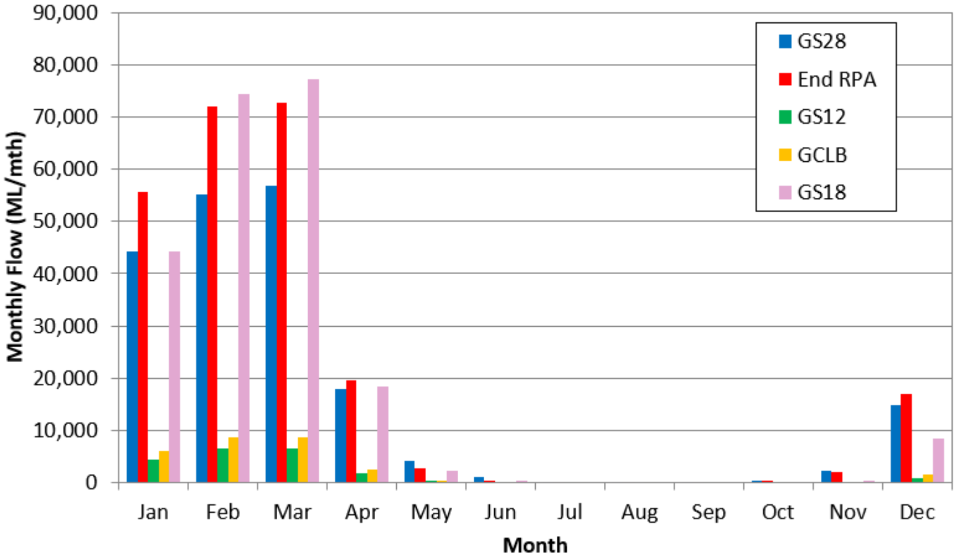


Figure 5-75 Mean monthly flows, Water Solutions (2020)

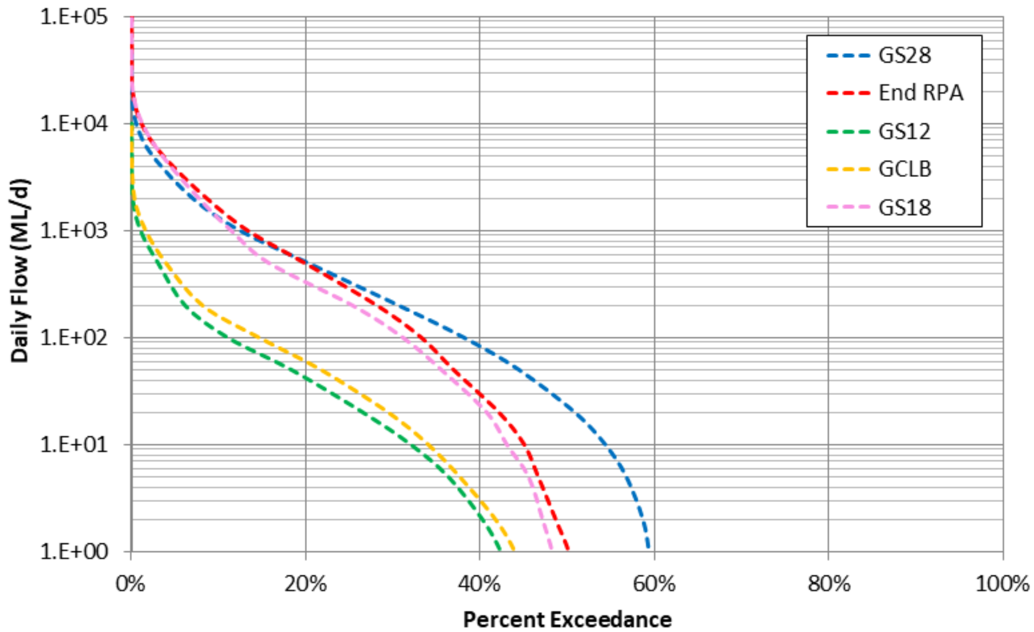


Figure 5-76 Daily flow exceedance, Water Solutions (2020)

Table 5-31 shows that site loads for some COPCs are of a similar order to natural loads (e.g. Mg), while others are much larger than natural loads (e.g. U) or much smaller than natural loads (e.g. Cu).

A number of potential improvements or extensions to the model have been identified during the project, and the model provides results that allows future work to more closely focus on areas of likely concern. The results produced by the RWSM are considered preliminary by ERA and not being used for evaluation against closure criteria. The RSWM model is currently undergoing further updates to address key stakeholder feedback, address improvements identified through development of the model, and included updated post closure solute transport loadings predictions (Section 5.5.2.11). Results from the RSWM update will be provided in the MTC Pit 3 closure application.

Following completion of the update to the RSWM in late 2020, multiple projects, including assessments of sediment accumulation, human diet and health, ecosystem vulnerability, release water pathways and cumulative aquatic risks can be conducted to assess if water quality closure criteria/objectives will be met. This will include additional studies such as assessing the traditional diet, risks associated with the predicted water quality, and predictions of accumulation of uranium into sediments. This will also inform decisions on what is as low as reasonably achievable (ALARA) on the RPA. Updates to the RSWM will be provided in future versions of the MCP.

Table 5-31 Mean annual COPC loads in model inputs, Water Solutions (2020)

COPC Description		Natural Load (kg/a)	Site Load (kg/a)			
Name	Symbol		Y1	Y20	Y270	Y10000
Magnesium	Mg	1.29E+05	2.74E+04	1.38E+05	1.51E+05	7.12E+04
Calcium	Ca	9.29E+04	4.60E+03	2.26E+04	2.72E+04	1.25E+04
Nitrate	NO3-N	2.36E+03	1.51E+02	6.82E+02	9.43E+02	4.16E+02
Manganese	Mn	2.28E+03	1.08E+03	1.04E+04	4.21E+03	2.99E+03
Uranium	U	1.05E+01	1.01E+02	4.58E+02	6.30E+02	2.78E+02
Ammoniacal Nitrogen	NH3-N (or TAN)	1.85E+03	5.36E+02	4.14E+03	1.62E+03	1.19E+03
Orthophosphate	PO4-P	9.99E+02	1.91E+01	8.72E+01	1.18E+02	5.24E+01
Copper	Cu	7.36E+01	1.51E-01	1.07E+00	3.91E-01	3.02E-01
Lead	Pb	9.20E+00	8.96E-03	3.16E+00	5.14E-01	6.12E-01
Cadmium	Cd	9.20E+00	1.23E-02	8.72E-02	3.19E-02	2.47E-02
Iron	Fe	3.79E+04	2.30E+01	4.67E+03	7.73E+02	9.10E+02
Zinc	Zn	1.47E+02	7.84E-01	1.34E+01	3.27E+00	3.07E+00
Chromium	Cr	1.10E+02	3.70E-02	2.60E-01	9.54E-02	7.37E-02
Vanadium	V	1.29E+02	5.04E-03	3.54E-02	1.30E-02	1.00E-02
Nickel	Ni	3.68E+02	6.16E-02	6.90E+00	1.18E+00	1.36E+00
Radium	Ra226	2.42E-05	4.77E-06	2.30E-05	2.93E-05	1.32E-05
Polonium	Po210	1.19E-08	1.18E-10	8.33E-10	3.04E-10	2.35E-10
Total Suspended Sediment	TSS	1.86E+06	1.19E+06	0.00E+00	0.00E+00	0.00E+00

5.4.5 Aquatic ecosystem assessment & framework development

ERA contracted BMT Ltd. to define a process to interpret modelling results against regulatory requirements. The broad aim of the project is to develop a practical and transparent framework to assess effects of COPCs on receiving environments within the RPA during the closure phase, with an initial focus on magnesium.

The project is in its third phase. The first two phases involved review of existing information and stakeholder meetings to identify preliminary indicators for all primary environmental objectives and draft environmental and community values (ECVs) for different water types on and off the RPA (BMT WBM 2017, BMT 2018). More information on the supporting study in Section 5.5.2.16)

The third phase of the project developed a Vulnerability Assessment Framework (VAF) to aid the interpretation of modelling results, with a focus on the potential effects of magnesium on ECVs of the mine area.

Ecological vulnerability assessment fills the knowledge gap that exists between laboratory and field effects experiments on a sub-set of species or assemblages (i.e. the information underpinning the SSB Rehabilitation Standards) to understanding risks to higher levels of organisation and/or to other species and species groups (De Lange *et al.* 2010). Ecological vulnerability assessment considers not only the direct sensitivity of organisms to a stressor, but also trophic and habitat relationships and therefore the potential for indirect flow-on effects.

The VAF involved the following steps:

- identification of ECVs, including 'key species' that are important from biodiversity and cultural perspectives, as well as important habitats and other groups
- selection of a set of ecosystem components and processes based on the approach outlined in the 'National Framework and Guidance for Describing the Ecological Character of Australia's Ramsar Wetlands' (DEWHA 2008)
- development of conceptual models of key processes and linkages with ECVs
- preparation of conceptual diagrams to illustrate and summarise key ecological processes operating in the study area. The process diagrams provide a basis for examining potential timing of mining releases (i.e. exposure) and key biological processes in this project phase.
- assessment of the direct (i.e. toxicity) and indirect (i.e. food resources and habitats) sensitivity of ECVs to magnesium; (iv) assessment of the adaptive capacity of ECVs.
- consideration of sensitivity at the individual organism level, and how this translates to vulnerability at higher organisation levels (the local species population, assemblage, community/habitat and/or ecosystem level) as well as the capacity of biota to recover

Vulnerability is based on the consideration of following elements (De Lange *et al.* 2010, Weißhuhn *et al.* 2018):

- level of exposure to stressors – which will be predicted by the surface water modelling project
- sensitivities to stressors such as magnesium, both in terms of direct effects and indirect flow-on effects to habitat and or food resources. This requires consideration of the biological traits of biota, and the structural and functional relationships between the organisms, and the abiotic environment
- capacity to recover following a perturbation, such as exposure to a contaminant. This is also known as resilience or adaptive capacity

The level of exposure will be predicted by the surface water modelling. Scoring matrices and descriptions were developed to categorise sensitivity and resilience. These were based on multiple information sources including ecotoxicology assessments and field studies, local and national literature, and expert elicitation from an independent expert panel.

The scoring of sensitivity and adaptive capacity for the selected ecosystem components was undertaken independently by the expert panel and project team. Scoring results were received in June 2019 and a draft report distributed to the expert panel in late 2019. Finalisation of the report is pending rescoring to include several new lines of evidence on magnesium effects produced by the SSB (draft summary received July 2020). Re-scoring of ecosystem sensitivity to magnesium is planned for Q3 2020 to provide information to inform the Pit 3 application.

5.5 Supporting studies

ERA, in collaboration with stakeholders, has prepared a list of Key Knowledge Needs (KKNs) to address gaps within closure planning. Both ERA and the SSB will implement the KKN projects, either independently or cooperatively depending on the project

The list of KKNs as updated in May 2020 is provided as Appendix 5.4

This section provides summaries of the closure supporting studies and is arranged into the overarching study areas below to align with the KKN themes where practical.

- Landform
- Water and Sediment
- Health Impacts of Radiation and Contaminants,
- Ecosystem establishment.

5.5.1 Landform

This section provides summaries of the completed studies relating to landform development.

KKN title	Project title
LAN2: Understanding the landscape-scale processes and extreme events affecting landform stability	Assessment of impact on stability of the rehabilitated landform from identified landscape-scale processes
LAN3: Predicting erosion of the rehabilitated landform	Rock Size Distribution on Pit 1 final landform
	Monitoring of Pit 1 Landform Shape, Stability and Consolidation
	Pit 1 Monitoring of Sediment Discharge

5.5.1.1 Landform evolution modelling

A number of landform studies have been undertaken to address key closure issues and risks, including removal of all site infrastructure and backfilling of pits, containment of tailings and erosion of the final landform. These studies, including those completed by both ERA and the SSB on the trial landform (TLF), have informed the overall design and predicted performance of the current final landform design.

Once the two mined-out pits have been backfilled with tailings and waste rock, the landform and surface cover will be built to the final approved design. The final landform aims to simulate the hill slope environmental processes that determine the sustainability and diversity of ecosystems in analogous undisturbed environments. The land use values ascribed to the mine area by the Traditional Owners are also being considered in the design. These values relate to restoring safe access to the site to allow cultural uses that occurred before mining.

The design of the final landform has been determined from a digital terrain model of natural analogue areas with the aim of producing a landform with similar indices of erosion and runoff distribution to the natural landscape (Hollingsworth & Lowry 2005). The shape of the current final landform is largely determined by the requirement to maintain pre-mining drainage and catchment areas and to ensure stability in either the current climate/rainfall regime or the predicted regime that may result from climate change. The TSF walls and western edges of the southern and western stockpiles sit atop high ridgelines in the pre-mining landscape. These ridges form prominent features of the final landform and, combined with a reinstated ridgeline over Pit 1, restore catchment areas to close equivalents of their pre-mining form. Topography of the final landform is similar to the pre-mining landform; maximum elevation after consolidation increases from 38 metres pre-mining to a final landform maximum of 44 m Australian height datum (AHD).

Initial landform development was based on landform design criteria (Hollingsworth & Lowry 2005, Hollingsworth & Meek 2003, Hollingsworth *et al.* 2003a, Hollingsworth *et al.* 2003b) and described in the ERA 2005-06 Closure Model, which was subsequently issued to stakeholders (McGovern 2006). The final landform design described in McGovern (2006) continues to be revised to ensure that it takes into consideration changing stockpile material grades, volumes and locations.

The preliminary slope analysis performed on final landform version 5 (FLv5) shows very gentle slopes across the landform with maximum slopes, measured from the ridgelines to the edge of the disturbed area, ranging in grade from approximately 2 percent to 5 percent (Figure 5-77). A slope analysis was also completed as part of the erosion and sediment control design work. This showed slopes vary from about 1 in 30 (3 %) to 1 in 200 (0.5 %), with the larger catchments tending to have lower slopes, although this is not always the case. This has not changed significantly in the latest version of the final landform, FLv6.2 and it continues to meet the original design intent (Section 9.4.5).

Each version of the landform has been subjected to landform evolution modelling by the SSB to assess the performance of the landform against closure criteria. The landform evolution modelling undertaken by the SSB (Lowry & Saynor 2015) applied a modified version of the CAESAR-Lisflood landform evaluation model (Coulthard *et al.* 2002, Coulthard *et al.* 2013) to assess the geomorphic stability of the final RPA landform over timeframes ranging from decades to millennia.

The CAESAR-Lisflood is an enhanced version of the CAESAR landform evaluation model that had previously been used to assess the geomorphic stability of the Ranger Mine TLF. The key data inputs used by the CAESAR-Lisflood landform evaluation model were a digital elevation model (DEM), rainfall and surface particle size. The catchment areas used for assessing the Ranger Mine conceptual landform are shown in Figure 5-78.

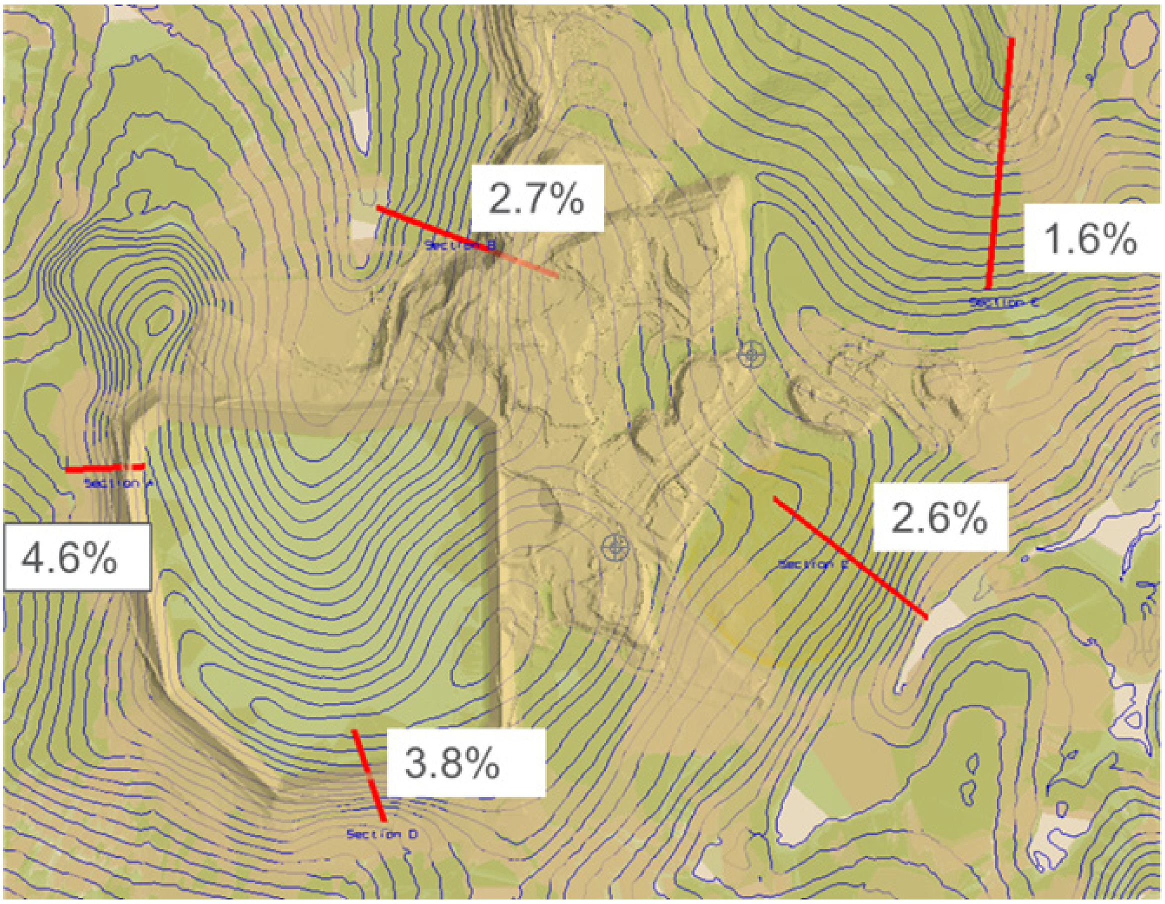


Figure 5-77: Preliminary slope analysis looking at the steepest slopes

The model has, to date, been conservative in nature, having only minimal vegetation on the surface for the entire 10,000-year period, and currently excludes any orthodox storm water and erosion control structures to reduce bedload yields. However, more recently the SSB has incorporated a grass cover layer.

The modelling conducted in 2013 on the fourth version of the landform (Lowry *et al.*, 2013) identified a number of potential erosion issues across Pit 1 and Pit 3 tailings. The landform was subsequently redesigned to version five (FLv5) based on the results of this model and assessed by the SSB (Supervising Scientist 2016b). The SSB subsequently recommended in January 2016 (Supervising Scientist 2016b) that the landform design be modified to reduce the chance of deep gully formation, particularly in the Djalkmarra Creek and Corridor Creek catchments. The Supervising Scientist (2016b) put forth the following options for consideration:

- modification of the slopes within the affected catchments
- application of an armoured surface to sections of the catchment to make the surface more resistant to fluvial erosion and runoff
- armouring the toe of the landform in the area currently occupied by the road around the south-east edge of Pit 3

The study (Lowry & Saynor 2015, Supervising Scientist 2016b), predicted both the locations of gully formation and the broad scale erosion and deposition across the landform with long-term denudation rates being calculated. The results show most of the deposition occurs in the first 100 years with erosion ongoing throughout the model. Denudation rates decrease over time and are found to approach the published background denudation rate for the region.

Modelled denudation rates after 10,000 years provided by the SSB are:

- Coonjimba: 0.05 mm per year
- Corridor Creek: 0.03 mm per year
- Djalkmarra Creek: 0.02 mm per year
- natural background: 0.01 – 0.04 mm per year

Predicted erosion for simulated periods of up to 10,000 years in the Corridor Creek and Djalkmarra catchments has been shown in Figure 5-79 and Figure 5-80, respectively. These modelled results indicated an exponential decline in erosion/gully formation, but also the potential formation of gullies up to 9 m deep in areas of the landform that are close to buried tailings. These will be the locations for the design of drainage channels and other erosion mitigations to minimise the potential impact on landform stability and revegetation success.

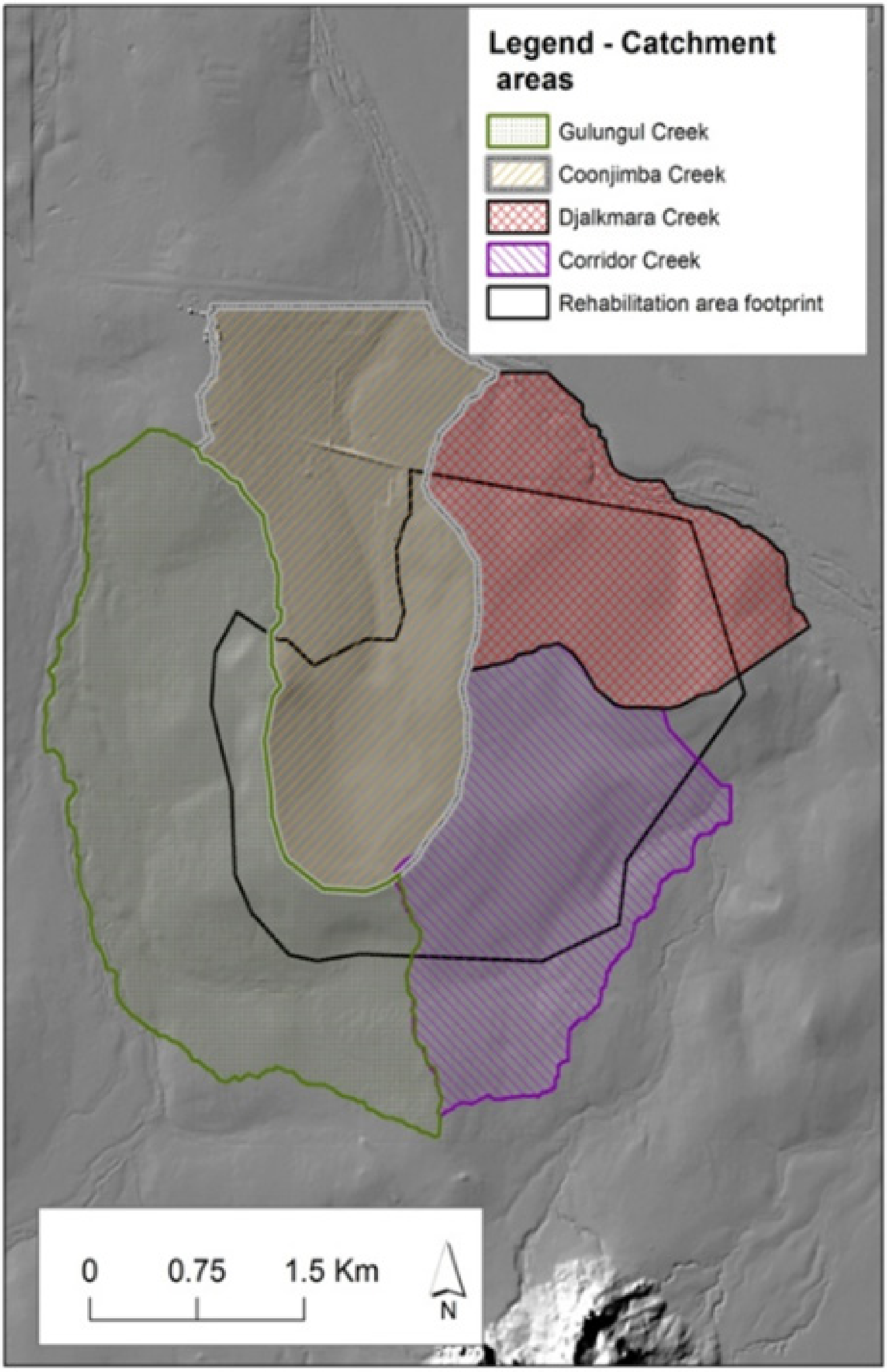


Figure 5-78: Catchment areas – Ranger Mine conceptual landform (Lowry & Saynor 2015)

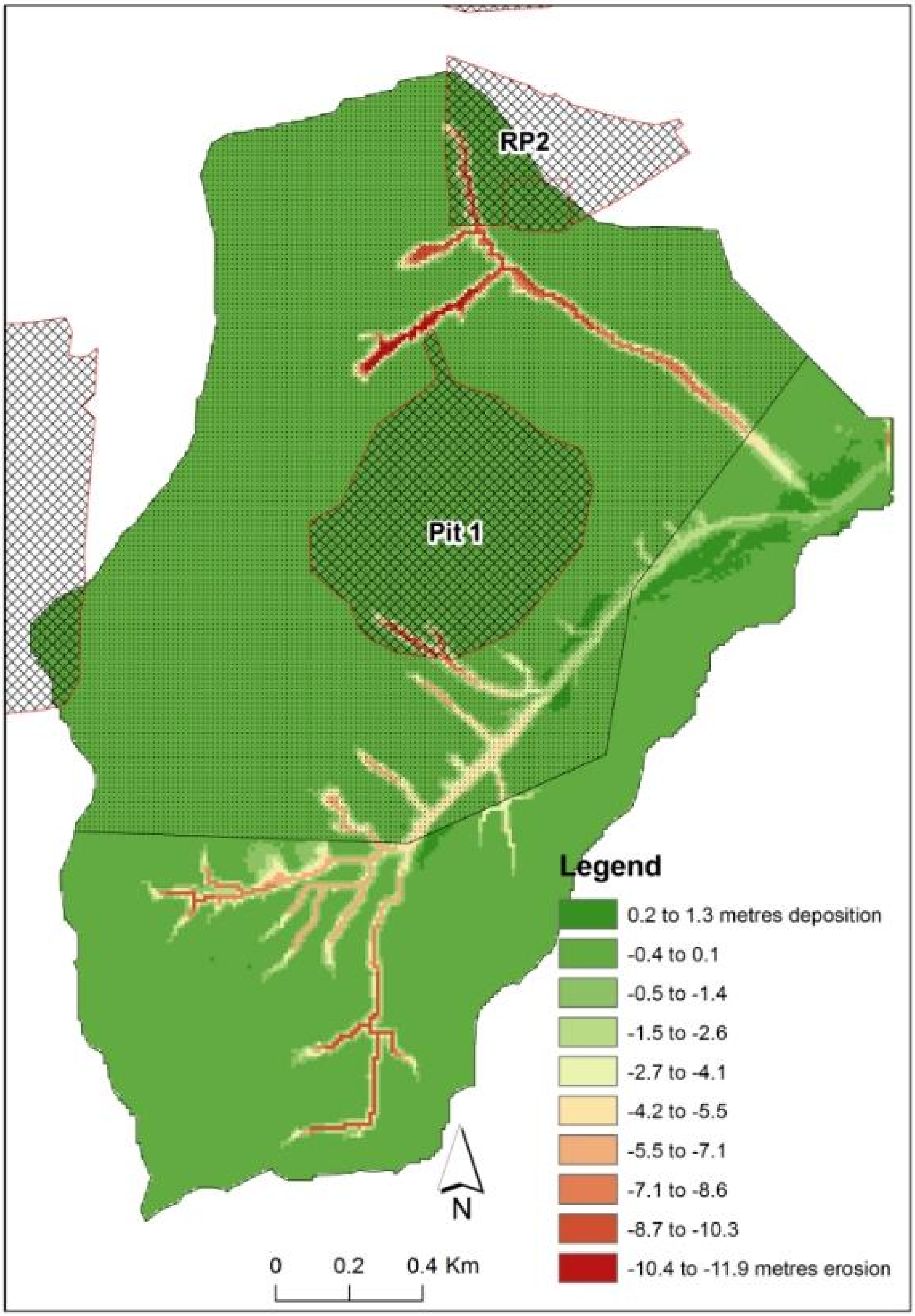


Figure 5-79: Corridor Creek catchment – extent of erosion/deposition zones after simulated period of 10,000 years (Supervising Scientist 2016d)

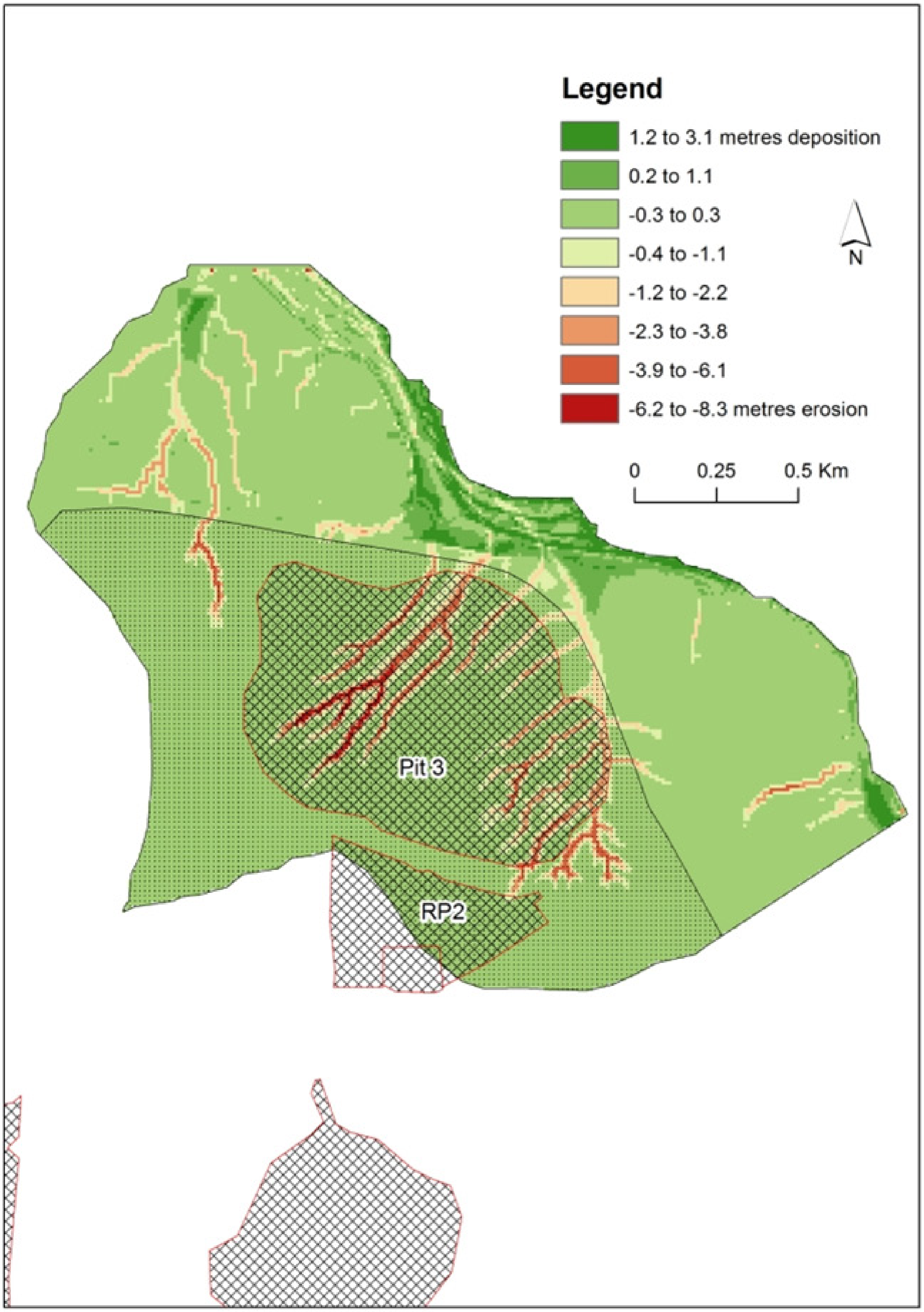


Figure 5-80: Djalkmarra catchment – extent of erosion/deposition zones after simulated period of 10,000 years (Supervising Scientist 2016c)

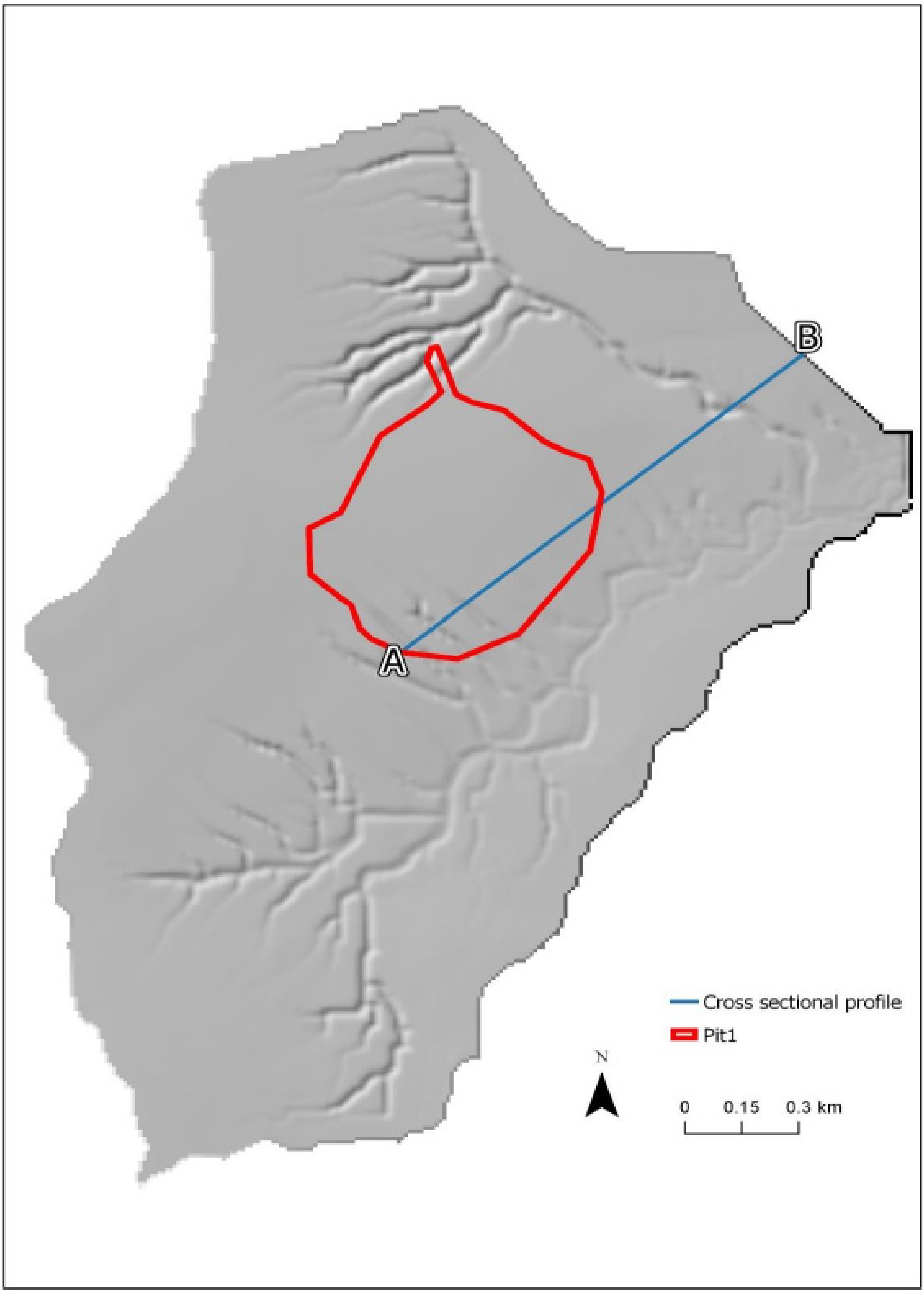


Figure 5-81 Surface of Corridor Creek catchment after a simulated period of 10,000 years under an extreme dry-rainfall, grass cover only scenario (Supervising Scientist 2019)

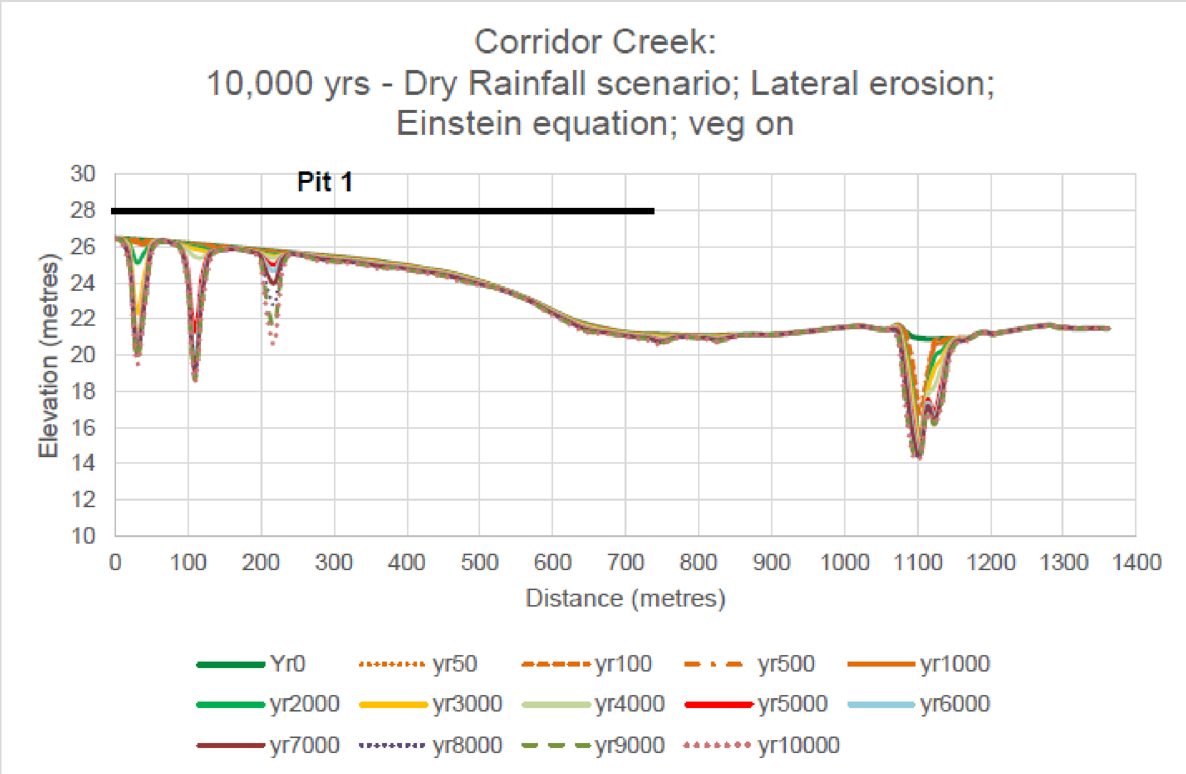


Figure 5-82 Profile across Pit 1 (extent of Pit 1 shown by black line) after a simulated period of 10,000years under an extreme dry-rainfall, grass cover only scenario (Supervising Scientist 2019)

A number of limitations of the modelling work were identified by the SSB. The following improvements are being implemented to ensure model outputs are both plausible and scientifically defensible. These improvements include:

- the development of a stochastic synthetic rainfall dataset to generate a series of unique rainfall scenarios which may occur within a period of 10,000 years. This has allowed uncertainty in predictions to be better accounted for and will provide a range or probability of likely outcomes.
- an enhancement of the effect of vegetation community growth (vegetation has a major effect on the erosion potential of the landform surface) on landscape evolution within the landform model. The vegetation parameter values used in the CAESAR-Lisflood model have been better defined and continue to be reviewed to better account for the effects of developing vegetation cover over the area of the Ranger minesite.
- consideration of the role of fire, given its role in the northern Australian landscape and potential to disrupt or prevent the development of specific vegetation communities
- integration of a dynamic vegetation model linking soil moisture to biomass growth

- implementation of an effective weathering function into the model to reflect the natural rate of both physical and chemical weathering and to ensure the models do not prematurely predict sediment exhaustion from the environment
- Based on the modelling and advice from the SSB, changes to the final landform design surface were made to address concerns in key areas and incorporated into the final landform version FLv6.2. This included the diversion of all major drainages away from the pits and areas identified in the modelling predictions. The DEM Version FLV6.2 was provided to the SSB in 2018 for assessment on the performance of selected catchments of the landform, using the CAESAR-Lisflood landform evolution model (LEM). The SSB conducted a number of simulations on the current FLV6.2 landform in order to assess, at an early stage, erosion characteristics over the Pit 1 catchment, and whether the landform is adequate for assessment of the final landform against closure criteria. The SSB provided their feedback in a memorandum dated 21 February 2019, with additional advice provided in Technical Advice #010 on 13 September 2019. The most recent advice provided by the SSB is summarised below.
- Initial simulations run up to 1,000 years across the Corridor Creek catchment indicated that gullies deep enough to expose tailings are unlikely to form across the surface of Pit 1 within a simulated period of 1,000 years. Subsequent simulations have since been run to model a range of scenarios in the Corridor Creek catchment for a simulated period of 10,000 years.
- Simulations of an extreme dry-rainfall scenario, over a 10,000-year period, predict several gullies with approximate depths of up to 8 metres may form across the southern edge of the Pit 1 surface with gullies at the deepest point at a depth of about 19mAHD. This simulation predicts that there remains up to 13m of waste rock between the bottom of the predicted gullies and the predicted tailing surface provided by settlement monitoring (Figure 5-81 and Figure 5-82). This scenario included the presence of grass cover, which serves to reduce the effect of erosion, but does not include the establishment of a full vegetation community.
- By applying an armoured surface to this same Pit 1 surface at the initiation of gully formation at year 1,000, it was found that further gully growth or formation was prevented within the subsequent 1,000 year simulated period (Figure 5-83).
- Annual denudation rates for the extreme dry-rainfall scenario of the Corridor Creek catchment were predicted fall into the range of background rates within 10,000 years, of 0.04mm/yr +/- 0.03 (Figure 5-84).

The SSB stated that additional rainfall scenarios are now being modelled, for periods up to 10,000 years, including extreme wet-rainfall scenarios. Further assessments are also required of the FLV6.2 landform outside of the Corridor Creek catchment, thereby identifying locations on the final landform may require additional mitigation such as surface armouring, to eliminate any significant gullying. Results of these simulations will be presented in subsequent versions of this MCP, once completed.

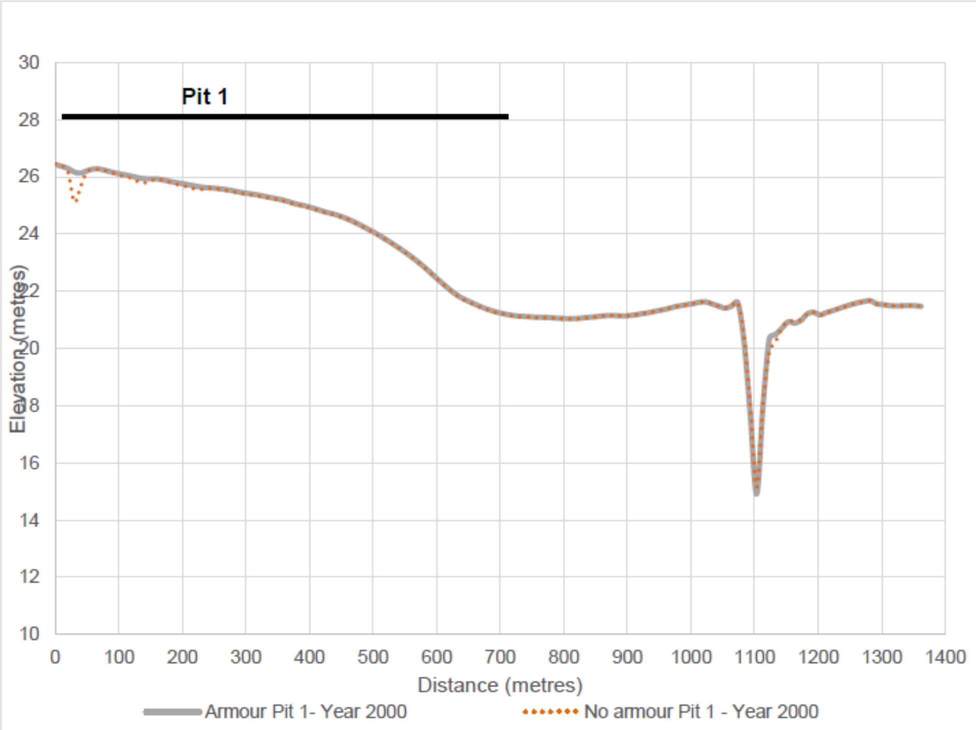


Figure 5-83 Effect of armour versus unarmoured surface on gully formation in the Corridor Creek catchment (Supervising Scientist 2019)

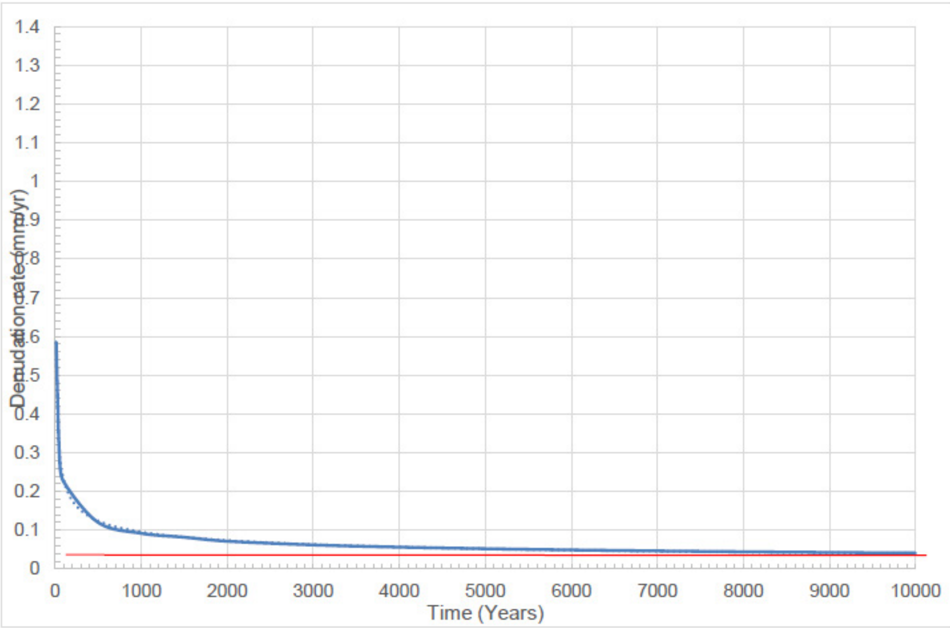


Figure 5-84 Modelled denudation rate over a simulated period of 10,000 years under an extreme dry-rainfall, grass cover only simulation. The red line represents the background denudation rate (Supervising Scientist 2019)

The results of the simulations to date provide a guide for future enhancements both to the landform design and to the landform evaluation model software. Existing results combined with the proposed work will provide increased confidence that the CAESAR-Lisflood model will be able to correctly predict the potential paths for evolution of a rehabilitated landform once it has been constructed.

The SSB has advised ERA that landform erosion modelling results are indicative only and should not be used to provide precise locations or depths of potential gully erosion, as such this information has only been used to guide the development of the final landform.

In mid-2019 ERA engaged a Rio Tinto hydrologist to build capacity in the assessment of closure landforms using the CAESER-Lisflood landform evolution modelling software. ERA is currently evaluating closure landforms and completing sensitivity testing of key model parameters including climate sequences, rainfall losses, particle size distribution and vegetation cover. This project has allowed for faster evaluation of landforms, and a better understanding of the modelling process and the implications for erosion outcomes dependent upon both landform design and parameter choice.

As mentioned above, the landform design is an iterative process. Design of drainage channels and other erosion mitigations is ongoing to minimise the potential impact on landform stability and revegetation success. ERA's ongoing engagement with a Rio Tinto hydrologist will assist ERA in understanding whether incremental changes in landform design are achievable and/or beneficial, and to better provide input into the final evaluation of landform stability at closure (denudation and formation of gullies).

5.5.1.2 Final landform material properties

The bulk material movement will be completed by moving all material with potential for environmental impact to the bottom of the mined-out pits where extensive solute modelling studies show it will be contained without any significant negative impacts on the natural environment. The final landform material is proposed to be low uranium content 1s waste overburden rock which is found in select stockpiles on the Ranger Mine. The remainder of the landform and pit backfill material will be made up of a mixture of 2s and 1s waste rock. Refer to Section 2.2.1 for details of the rock grading and content.

Table 5-32 shows the indicative particle size distribution for the 1s waste rock material taken from the Ranger Mine TLF (Saynor & Houghton 2011). ERA have also completed particle size distribution analysis for larger mineralised material in the Ranger Mine stockpiles, for various grades of material ranging from 2s to 7s, using fragmentation software. Figure 5-85 provides the results of this analysis.

Table 5-32: Particle size distribution for waste rock landform (by sieve analysis)

Sample name	% Sample > 2 mm	% Sample < 2 mm	% Sample < 63 µm	Total sample mass (g)
Minimum	50.4	21.3	20.9	3,922
Maximum	78.7	49.6	4.3	9,422
Average	63.1	36.9	9.6	6,198

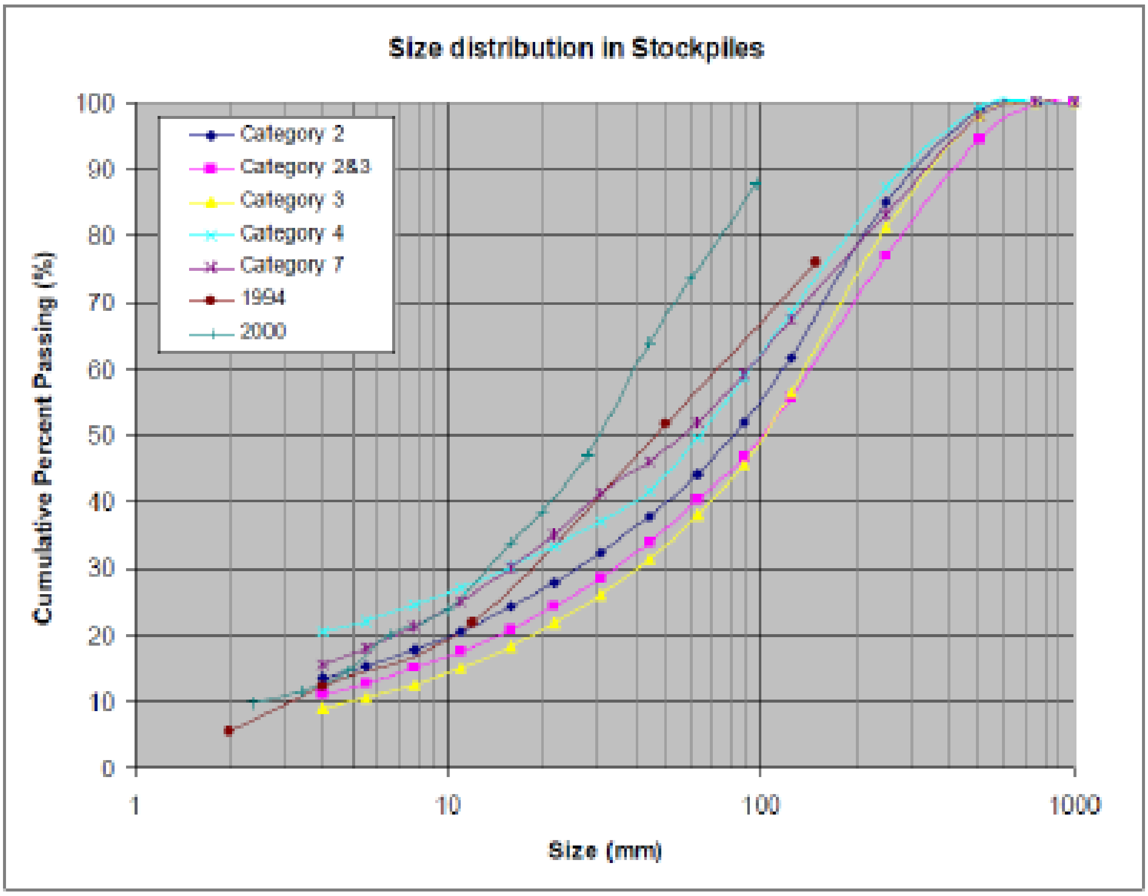


Figure 5-85: Particle size distribution for waste rock in stockpiles (by fragmentation software)

Hollingsworth *et al.* (2003a p 4-5) describes the significant number of studies that have been completed on the waste rock in stockpiles on-site, particularly in relation to soil formation. An excerpt from this report (excluding references) is provided as follows:

> *"Much of the rock material exposed on the surface of the stockpiles weathers rapidly to form rudimentary soil materials. A stony armour surface develops within five years, together with an underlying vesicular silty crust, analogous to desert pavement soils. This effectively seals the surface and is responsible for low infiltration rates. Below the*

compacted surface layer, the stockpiles can have very low bulk densities and consequently appreciable deformation and settlement was anticipated in the long term.

The chemistry and mineralogy of waste rock material has been analysed and rapid weathering and physical degradation of waste rock on the surfaces of the stockpiles has been observed. This weathering is compared with the end products of weathering in the soils and saprolite of the natural landscape.

A number of distinct 'mine soil' types have been recognised on the waste rock stockpiles. These include:

- *unweathered and weathered rock without profile development*
- *stony/gravelly desert-like pavement and an intergranular surface vesicular crust; with or without an A0 horizon*
- *stony/gravelly desert-like pavement and an intergranular surface vesicular crust overlying a vesicular loamy or silty crust horizon; with or without an A0 horizon*
- *stony/gravelly desert-like pavement and an intergranular surface vesicular crust overlying an altered, reddened B horizon with a weak tendency to become gravel-free and contain introduced fines and salts; with or without an A0 horizon*
- *bisequal soil; with or without an A0 horizon (surface litter layer)*
- *pseudo-acid sulfate soil with vesicular loamy crust; occurs in shallow depressions where seasonally perched water tables occur.; with or without an A0 horizon, and*
- *pseudo-acid sulfate soils without a vesicular crust, associated with alluvial fans on the banks of retention ponds.*

Incipient soil features develop within two years of construction of the waste rock stockpiles. Colour mottling (due to increased hydromorphy), variations in soil texture (as a result of water erosion of fine material), structure development, decrease in pH (due to pyrite oxidation) and sulfate weathering were recognised. Acid mine drainage risk has been generally low. Rock analyses of orebody 1 material indicated that total S levels in the samples of waste rock and ore were, with few exceptions, less than 0.04 percent, corresponding to very low potential acid sulfate risk. However, individual rock samples from the '7P' ore stockpile contained 3.51 percent S and exhibited conspicuous acid leaching and weathering features. This would account for the pseudo-acid sulfate soils that have been described.

Higher risks of acid generation in drainage water were identified with orebody 3 material. The more reactive behaviour of orebody 3 material has had implications for stockpile management. There are clear implications from the behaviour of this material in the future for the management and selection of materials that are suitable for finishing the final landform.

Mine soils were more fertile than the natural undisturbed soils of the area, and stockpiled natural soils, in terms of plant seedling growth. However, both P and N were deficient for optimal plant growth. In addition, glasshouse bioassays of mine soils indicated that symbiotic micro-organisms (rhizobia and mycorrhizal fungi) were absent or poorly represented in mine soils, other than those with a vegetation assemblage. It was found that there was no preferential (active) uptake or accumulation of U by plants. Also, all mine soil samples contained high exchangeable Mg levels and high concentrations of exchangeable K and S were measured in pseudo-acid sulfate soils."

Table 5-33 and Table 5-34 show the edaphic properties measured for the rehabilitated waste rock landform and the analogue natural landform (Hollingsworth 2010).

Table 5-33: Rehabilitated waste rock landform properties

Depth	**Rock content**	**Soil texture**	**Dry bulk density**	**Infiltration rate**	**Saturated hydraulic conductivity**	**Plant available water content**	**Soil penetration resistance**
	%		**kg.m^{-3}**	**mm.hr^{-1}**	**mm.hr^{-1}**	**mm.m^{-1}**	**MPa**
Soil							
0 – 0.5 m	>60	Sand	1.4 – 2.3	1 - 10	1,000	10	>3
0.5 < 1.5 m	50 < 60	Sandy loam	>1.6		1 - 10	50	
>1.5 m					>1,000	10	
Landform	**Recharge rate**	**Runoff coeff.**	**Relief**	**Catchment area**	**Slope**		
	10 – 25% of rainfall	>50%	<5 m	11 ha	0 – 3%		

Table 5-34: Analogue landscape properties

Soil depth	Gravel content %	Soil texture	Dry bulk density kg.m^{-3}	Infiltration rate mm.hr^{-1}	Saturated hydraulic conductivity mm.hr^{-1}	Plant available water content mm.m^{-1}	Soil penetration resistance MPa
0 – 0.5 m	>60	Sand to sandy loam	1.1 – 1.7	300 – 4,800	1,000	10	>3
0.5 < 1.5 m	50 < 60	Sandy loam – sandy clay loam	>1.6		60 – 4,500	50	
1.5 – 2.0 m	>60	Sandy loam	>1.8		0.4	50 – 100	
2.0 – 3.0 m					0.08	50 – 100	
Landform	**Recharge rate**	**Runoff coeff.**	**Relief**	**Catchment area**	**Slope**	**Leaf area index**	
	5 – 10% of rainfall	>20%	<30 m	1,500 – 5,000 m^2	1 – 5%	0.8 – 1.6	

5.5.2 Water and sediment

This section provides summaries of the completed studies relating to Water and sediments as well as selected completed and ongoing KKN related studies. Some studies inform multiple KKNs and have only been included once to avoid repetition.

KKN title	Project title	Status	Section
WS1: Characterising contaminant sources on the RPA	Background COPCs in Groundwater	Completed	5.5.2.1
	Aquatic Sediments	In Progress	5.5.2.2
	Acid Sulfate Sediments Conceptual Model	Completed	5.5.2.3
	Interpreting Soil Assessments for Land Application Areas	In Progress	5.5.2.4
	Non-aquatic contaminated sites sampling	Completed	5.5.2.5
	Stockpile Drilling	Completed	5.5.2.6
	Solute Source Update	In Progress	0

KKN title	Project title	Status	Section
WS2: Predicting transport of contaminants in groundwater	Literature Review on Contaminant Mobility	Completed	5.5.2.8
	Update Groundwater Solute Transport modelling and Conceptual Model	Completed	5.5.2.9
	Post closure Solute Transport modelling with uncertainty analysis	In Progress	5.5.2.10
WS3: Predicting transport of contaminants in surface water	Surface water modelling	In Progress	5.5.2.11
	Surface water groundwater interaction	In Progress	5.5.2.12
WS5: Determining the impact of contaminated sediments on aquatic biodiversity and ecosystem health	Acid Sulfate Sediments management options	In Progress	5.5.2.13
	Surface Water Pathway Risk Assessments (Release pathways onsite).	Planned	5.5.2.14
WS6: Determining the impact of nutrients in surface water on aquatic biodiversity and ecosystem health	Eutrophication Risk Study	In Progress	5.5.2.15
WS7: Determining the impact of contaminants in surface and ground-water on aquatic biodiversity and ecosystem health	Aquatic Ecosystem Assessment & Framework Development	In Progress	5.5.2.16

5.5.2.1 Background COPCs in groundwater

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- WS2. Predicting transport of contaminants in groundwater
- WS7. Determining the impact of contaminants in surface and ground-water on aquatic biodiversity and ecosystem health
- RAD2. Radionuclides in aquatic ecosystems
- RAD9. Impacts of contaminants on human health

Background COPCs require characterisation in order to identify the natural range in concentrations in different HLUs across the site. This will inform the post-closure solute transport modelling projects, solute source Area / Concentration conceptual model and surface water modelling projects

Groundwater and surface water modelling are key requirements to support the Pit 3 capping and backfill application to MTC. (Project is discussed in Section 5.2.7)

Previous studies on background COPCs in groundwater at the Ranger Mine were completed by Esslemont (2015, 2017). The key objectives of this study were to better define a list of site-specific background dataset and to derive background concentration limit/threshold for each of the COPC.

Scope and approach

- review of historical studies to provide justification for focussing on the previously selected COPC
- database collation and initial screening: Download of comprehensive dataset from ERA and initial review and screening to remove data not useable in the assessment.
- identification and extraction of background dataset
- review of data quality objectives
- ensure representative data are queried and obtained for appropriate locations and times
- identification of important data characteristics and patterns that need to be considered in the full evaluation
- screening of data for acceptable quality considering analytical methods, method detection limits, presence of laboratory qualifies and metadata
- visualisation of data
- development of descriptive data statistics
- evaluation of data gaps
- assessment of data types, metadata, completeness through time and space for the corresponding hydrolithologic units
- evaluation of sample size and frequency to ascertain the likelihood that the existing data are sufficient to characterise background concentrations with the desired level of acceptability
- development of background dataset
- justification of inclusion or exclusion of data points from the site specific background data set using a compilation of several lines of evidence. This includes temporal analysis, population partitioning, geochemical analysis and chemical fingerprinting

- integration of all the lines of evidence to develop the background dataset with consideration for the conceptual hydrogeological model
- derivation of background COPC concentration limits and background threshold values
- active monitoring of the project through regular engagement with the consultant Environmental Resources Management (ERM).

Results and conclusion

The project was completed in June 2020 with delivery of the report *Ranger Uranium Mine Background Evaluation* dated 5 June 2020. In support of the report ERM developed nine interactive html dashboards allowing for full interrogation of the dataset and statistical analysis undertaken to develop the background threshold values. ERM presented via teleconference to stakeholders at the Ranger Closure Consultative Forum on 19th June 2020 where the report and supporting appendices were provided to stakeholders for review and feedback.

The completed project effectively refined the COPC list and identified the background dataset, established site-specific background datasets where minimum data criteria were met, and established background threshold values (BTVs) for COPCs in groundwater at the Ranger Mine. Further information on this project is described in section 5.2.7

Feedback was received from the SSB via email in July 2020. The SSB advised that, where sufficient data was available, they are in agreement with the COPC background threshold values that have been derived. Where there was insufficient data to develop a COPC background threshold value a suitable approach is required, either a low confidence value or future assessment following collection of additional data. Follow up engagement with the SSB has commenced and an approach is being developed to address this data gap.

Feedback from the DPIR was received on 26 August 2020 and was in agreement with comments made by the SSB.

5.5.2.2 Aquatic sediments

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- WS5. Determining the impact of contaminated sediments on aquatic biodiversity and ecosystem health
- RAD9. Impacts of contaminants on human health

Aquatic sediment sampling is required to understand any potential ecological impacts related to mine contaminated sediments. This will inform ALARA-BPT assessments which in turn inform the decommissioning requirements for onsite waterbodies.

An Independent Surface Water Working Group (ISWWG), established by ERA and the GAC to review surface water management and monitoring at Ranger Mine, made 15 recommendations (Hart & Taylor 2013). One recommendation related to sediment monitoring:

> *"A sediment monitoring program be re-introduced. In doing so due consideration needs to be given to the technical challenges in designing a program to reliably evaluate possible adverse environmental impacts during the operational phase of the mine, while providing benchmark data to detect possible impacts after closure."*

Parry (2016), recommended a sampling and analyses program based on leading practice and a review of historical data from earlier investigations of billabong sediments. The recommendations, agreed to by a stakeholder working group, were trialled in 2015 and implemented and refined in 2016 (Esslemont 2016). The sediment sampling conducted in 2016 was reported by Esslemont and Iles (2017).

These reports contain a well described pre-closure baseline dataset and demonstrate that there has been no sediment contamination in off-site billabongs as a result of mining. Given the improved water quality leaving the minesite in recent years the risk of sediment contamination off the RPA occurring now is negligible.

Metal contamination of onsite billabongs has not increased in recent years and the formation of acid sulfate soils (ASS) is now the recognised priority hazard to sediments in water bodies on the RPA. Therfore, the focus has now shifted away from routine monitoring of on and off site sediments to a targeted program to understand the ASS issues.

Sediment monitoring was undertaken to investigate acid occurrences in Coonjimba Billabong (Esslemont & Iles, 2015 and Esslemont, 2016). A review of this work contained recommendations for sediment sampling to improve the understanding of the ASS status and risks (Baldwin, 2017). This lead to the development of an ASS conceptual model for the minesite which will underpin the design of the ASS sampling program for 2020.

The objectives for this project are to:

- collect and analyse data from a sediment sampling program
- provide an inventory and assessment of sediment contamination (including ASS status) in waterbodies on the minesite (relative to reference sites) to inform closure risks and decommissioning plans.
- document the decommissioning plans in the Final Landform application
- inform future aquatic ecosystem monitoring that may be undertaken between 2020 and 2024

Scope & approach

Sediments from billabongs on the RPA will be sampled and analysed for COPCs identified in Parry 2016 and additional analytes identified for assessing the ASS risk.

The sampling locations are being finalised based on a review of the ASS conceptual model and recommendations from the SSB and their consultant. The sampling locations will be reviewed with stakeholders. Parameters have been previously agreed to by stakeholders. The need for sampling in future years will be based on the outcomes of the 2020 campaign and future risk assessments.

The sampling and analysis plan was reviewed by stakeholders during development (2018 – 2020).

Delays to sampling due to the permitting process for off-site locations, and delays in finalising the ASS conceptual model resulted in improvements to the sampling plan. The updated sampling, analysis and quality plan will be discussed with stakeholders prior to sampling. Stakeholders will review and evaluate draft reports prior to finalisation.

5.5.2.3 Acid sulfate sediments conceptual model

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- WS5. Determining the impact of contaminated sediments on aquatic biodiversity and ecosystem health

Historical sampling and assessment results have identified both potential acid sulfate sediment (PASS) and actual acid sulfate sediment (AASS) in Coonjimba Billabong (Esslemont & Iles 2015, Esslemont 2016). ASS in Retention Pond 1 has also been identified in the past (Esslemont 2016). In addition, CSIRO mapping (2011) identified a high probability of ASS presence in some areas on the minesite, including Georgetown Billabong, TSF, RP1, Coonjimba Billabong, former Djalkmarra Billabong and Magela Creek.

Subsequently, in order to assess the potential for, and risk from, ASS formation at the RPA, ERA engaged ERM to undertake an assessment based on the historical and current operational activities.

A preliminary site wide conceptual model has been developed, based on a collation and review of historical topography, groundwater and surface water data, and existing soil and sediment sampling result (ERM 2020a). The objective of the model is to further understand:

- source dynamics of ASS formation at the site
- mechanisms of PASS exposure and oxidation to form AASS
- potential pathways for acidification products (dissolved metals, acid and sulfate) from ASS sources areas
- surface water and groundwater receptors that may receive such acidification products

- potentially complete source-pathway-receptor (SPR) linkages

The following sections present the general methodology of the ASS assessment and key findings from the ERM assessment.

Scope and approach

The assessment involved a desktop review of site-specific reports on ASS, ground and surface water quality datasets, water level, historic rainfall, water management practices and consolidated GIS analysis to identify areas that met the conditions required to potentially form ASS.

The key differentiated terminologies adopted in this assessment, as shown in Figure 5-86, include:

- potential acid sulfate sediments: sediments that contain sulphides in a reduced condition and have the potential to generate acid if oxidised
- actual acid sulfate sediments: sediments that have oxidised to release acid, sulfate, and/or metal load
- areas where PASS or AASS have been confirmed based on sediment sampling or other assessment
- areas where the potential for ASS to have formed are identified in this assessment based on elevated concentration, water-logged conditions and other attributes

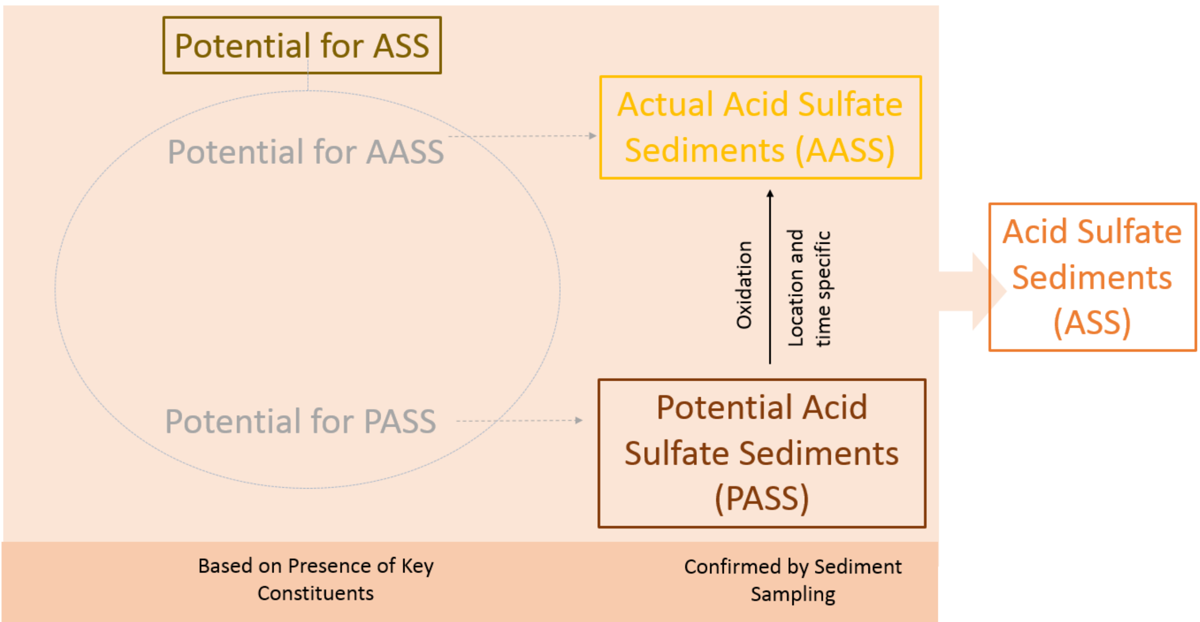


Figure 5-86 ASS terminologies (ERM 2020)

The conceptual model was developed using the structure shown in Figure 5-87, with section references as in ERM 2020. There are three key constituents that contribute to the potential formation of ASS: the potential water-logged conditions, elevated sulfate concentration (≥10 mg/L), and sufficient organic matter to establish the chemically reducing environment. Two former conditions can be interpreted from the consolidated historical data. However, due to the lack of data available for organic matter, a non-limiting environment is assumed in this assessment.

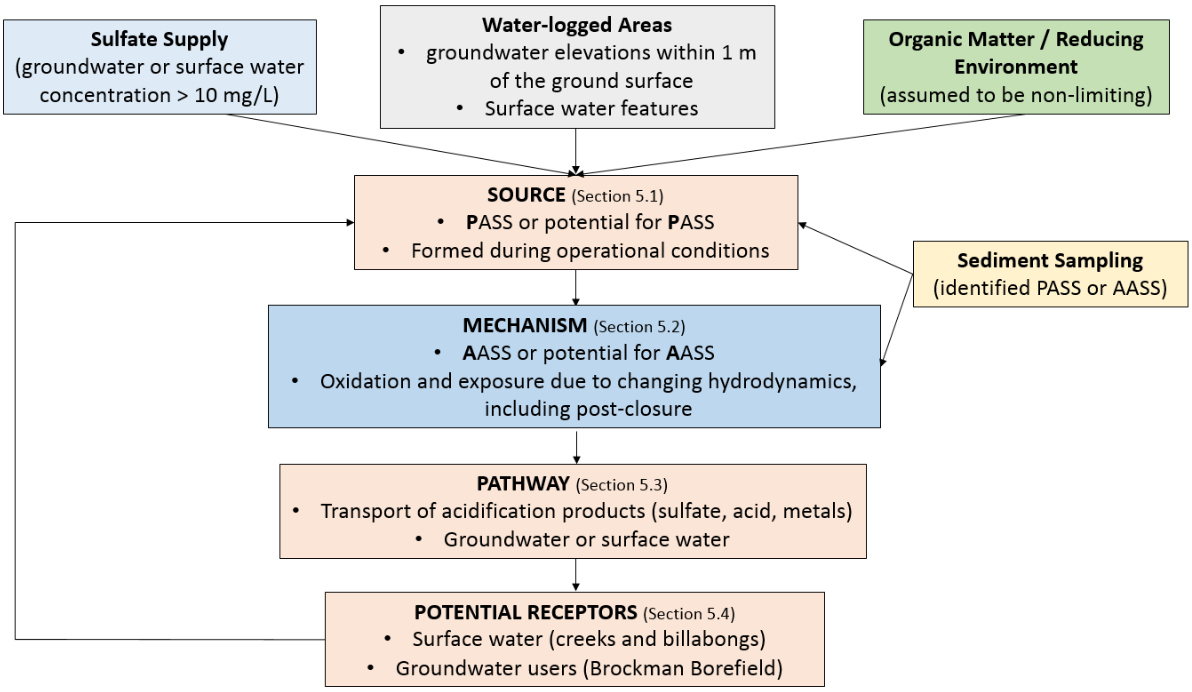


Figure 5-87 Development of preliminary site wide ASS conceptual model (ERM, 2020)

Considering the high seasonal variation in water quality and quantity, the preliminary site wide assessment was based on certain temporal periods for data interpretation to consider local seasonal behaviour of surface water and groundwater, and hydrodynamic changes resulted from water management activities. Six different time periods were assessed:

- wet-wet and following dry season
- dry-wet and following dry season
- wet season and following dry season corresponding to the onset of ASS conditions

The maximum sulfate concentrations in surface and groundwater and maximum groundwater elevations were selected from datasets for locations across the site for these periods as a conservative approach. The screened surface and groundwater datasets were consolidated and entered into GIS to identify areas with overlap of attributes required for ASS formation. The areas meeting these conditions are identified as “sources”, i.e. areas with potential for ASS formation.

Areas of sulfate supply and potential receptors for ASS products were also identified for each of the PASS source areas to develop a source-pathway-receptor linkage model.

Results and conclusion

The following results were produced for each time period being assessed:

- a set of sulfate concentration and groundwater elevation maps for each of 2 groundwater zones
- a map for each time period showing the intersection of sulfate concentrations ≥10 mg SO_4/L at or within 1 meter of the surface

Figure 5-88 summarises the results of these outputs, plus surface water where maximum concentrations of sulfate were ≥10 mg/L, in a preliminary ASS conceptual model. Note that areas shown as “not considered” are those areas where no or limited groundwater data were available for the periods of assessment. These areas will be considered in the next stage for the ASS assessment.

There are several areas conservatively considered to represent PASS or potential for PASS sources areas. These include the Coonjimba Creek/Coonjimba Billabong alignment, Magela Creek, Corridor Creek, and Gulungul Creek, where sulfate concentrations higher than 10 mg/L in groundwater occurred together with water logged conditions, or sulfate concentrations in surface water drainage lines and surface water bodies were higher than 10 mg/L.

The yellow shaded areas are considered a source (potential ASS area) in at least one of the 6 time periods assessed. Note that only a few small areas were identified as sources in all 6 time periods.

In many of the identified source areas AASS or PASS may not be present. A mechanism is required to shift from potential source area to PASS and further onto AASS. For example, potential source areas may be limited in organic matter, and thus no PASS or AASS can be formed. On the contrary, natural or mine-related changes to the hydrodynamic at the site may expose PASS that has the potential for oxidation and release of acidification into the surrounding environment and form AASS. For example, Coonjimba Billabong and areas along the Coonjimba Creek are identified as a PASS source area, where past acidification events were observed with both AASS and PASS have been identified along the alignment.

Figure 5-89 summarises the source-pathway-receptor linkages for the ASS conceptual model, with the source areas, the pathways for transportation and the potential receptors identified.

Several operational areas were identified as sulfate supply areas in regards to sulfate concentration in surface water and groundwater. These areas include the TSF and surrounding run-off collection sumps, process plant area, Sed2B, Corridor Creek Wetland filter, RP1 wetland filter, Western Stockpile and LAAs. Some of these sulfate supplies will not be present after closure. Others are included in the post-closure contaminant source conceptual model and the potential for them to be ASS sources will be assessed in the next steps.

ERA

The main surface water receptors that have the potential to be exposed to and impacted by oxidation of PASS and AASS include Coonjimba Creek, Coonjimba Billabong, Corridor Creek and Gulungul Creek (Figure 5-89).

The uncertainties in this stage of the assessment arise from accuracy of the DEM topographic surface, and the limitation of data availability in some areas for the periods analysed. In addition, there is uncertainty with temporal variation, as only maximum sulfate concentrations during the early wet season is adopted in this assessment; whereas a sustained increase above 10 mg/L sulfate is required to form ASS.

To confirm the presence of AASS and potential risk to the receptor areas now and following closure, a sampling program and risk assessment will be conducted in the near future, refer to Section 5.5.2.2

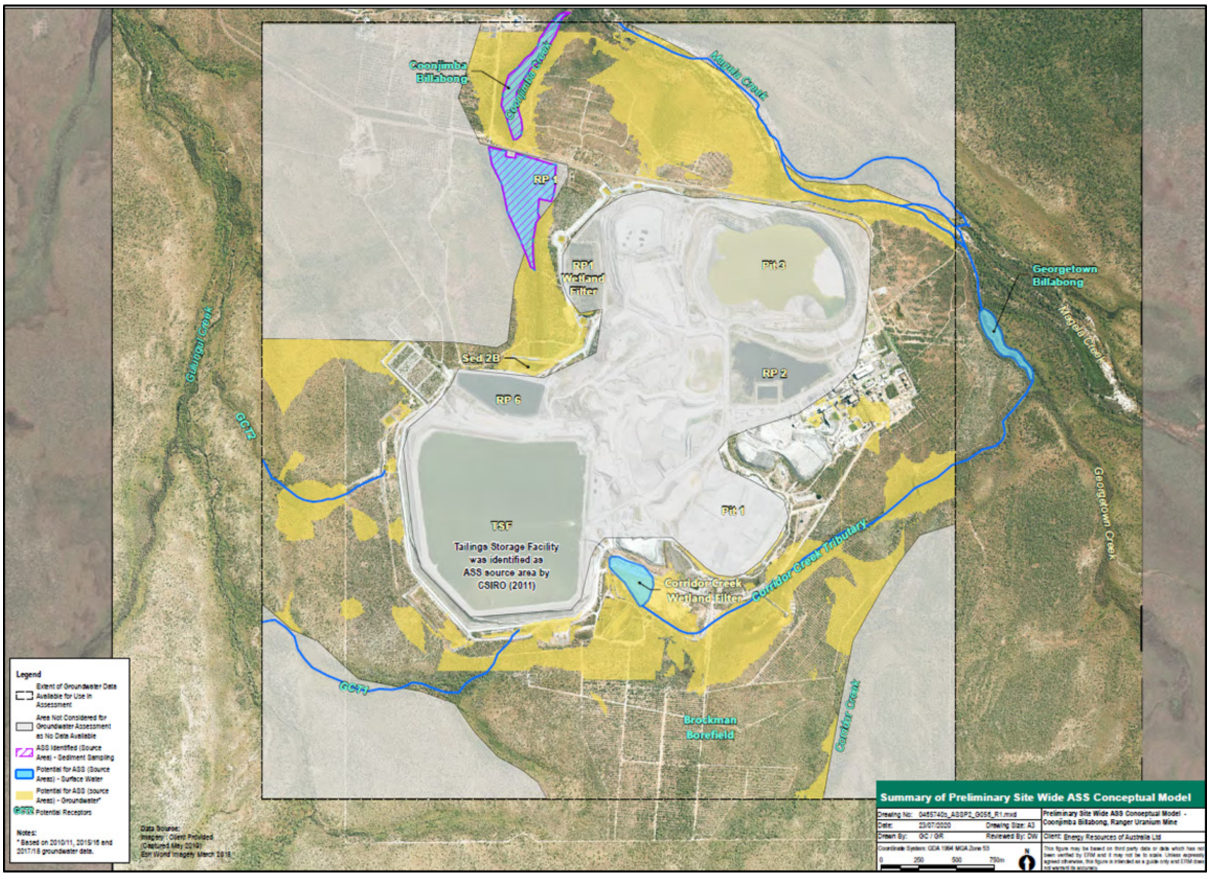


Figure 5-88 Summary of preliminary site wide ASS conceptual model

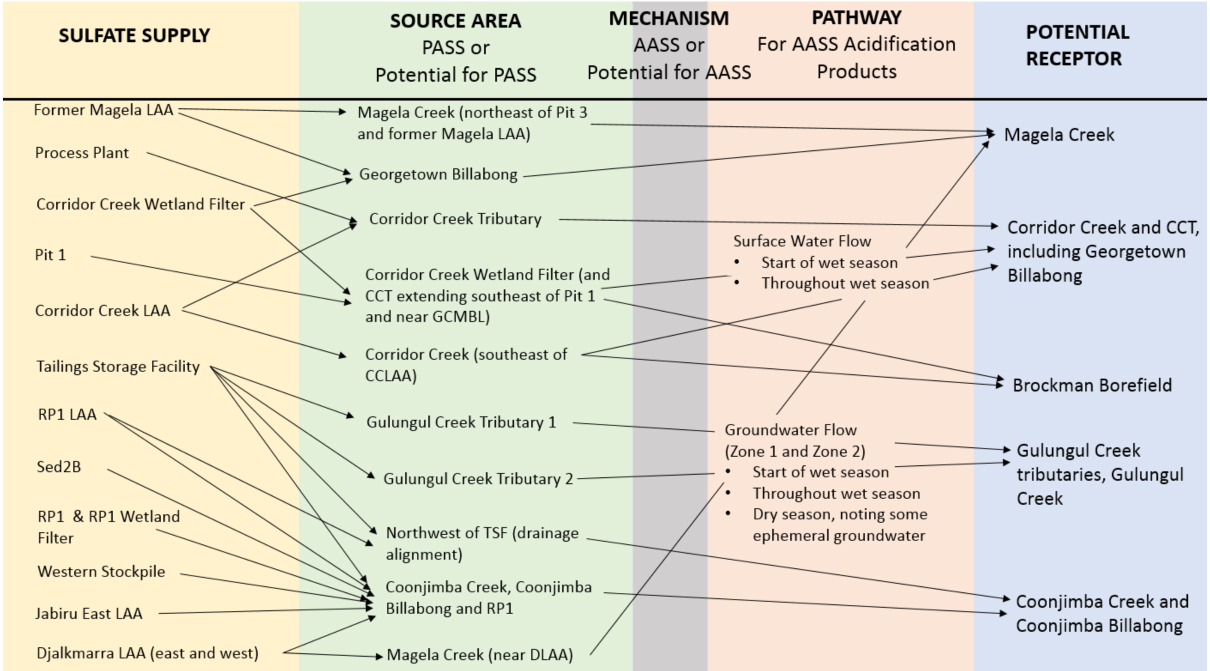


Figure 5-89 Summary of SPR linkages (ERM, 2020)

Following the development of the preliminary ASS conceptual model, ERA will investigate the risk associated with each conceptualised PASS source location. Targeted sediment sampling during 2020 dry season, along with the development of a location specific risk-ranking, are proposed to evaluate potential ASS formation in the sources areas identified. The risk-ranking for each identified PASS sources area will be based on location specific concentrations in surface water and groundwater, likelihood of hydrodynamic changes associated with closure, and the sensitivity of the potential receptor to acidification products. The risk assessment can then be used as a tool for monitoring regime development. An ASS model for closure conditions will be developed to inform closure risks and management strategies.

5.5.2.4 Interpreting soil assessments for land application areas

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- RAD9. Impacts of contaminants on human health

Previous assessments identified soils and sediments on the RPA that have become contaminated through treatment of pond water in wetlands and bunds, irrigation of pond water in the LAAs, the accumulation of low-level contaminants in waters passing through billabongs, and seeps and spills in the plant areas. An objective for closure is for soils to be remediated to a level where their environmental impact is ALARA.

LAAs have been used on the RPA since 1985 as a method of water disposal, primarily during the dry season. Types of water historically applied to the LAAs consist of:

- untreated pond water from RP2
- polished RP2 water – water that has passed through a constructed wetland filter
- managed release water
- permeate water – Water Treatment Plant permeate and Brine Concentrator distillate.

The LAAs have been designed to retain uranium in near-surface soils. Irrigated water disposed of at the LAAs has improved through time. There are eight LAAs at the RPA (Figure 5-90), spread across five areas. These consist of Magela LAA (MLAA) and MLAA extension, Djalkmarra LAA (east) (DLAA) and DLAA extension (west), RP1LAA and RP1LAA extension, Jabiru East LAA (JELAA), and Corridor Creek LAA (CCLAA). These cover a total area of 338 ha consisting of native and/or disturbed woodland or sparse woodland.

The behaviour of contaminants in the soils at Ranger and the contamination status of the LAAs has been studied extensively, with assessment available since 1979. Given the nature of the LAAs, soil investigations have largely focused on the upper 0.1m below ground level (BGL) of soils, however deeper samples (up to 6m BGL) have also been collected.

Recently, two sampling campaigns were undertaken in 2018 and 2019 to characterise the contemporary condition of soils within the LAAs (SLR 2018b, 2019). In 2020, a comprehensive literature review of the LAAs was undertaken (ERM, 2020 draft). All known data was also collated into an excel database, enabling data interrogation far easier than has been possible historically. This data is currently being analysed and a summary of findings will be provided in the next MCP.

A review of the information from the literature review and excel database is now underway to determine contamination of the LAAs. This will inform a BPT assessment, thereby informing the approach for remediation for each LAA, if required, based on ALARA. Detailed remediation plans, where needed, will be provided in future updates of the MCP.

The objective of this project is to understand what contaminants are present on the rehabilitated landform, whilst informing what COPCs to human health may exist. This will inform what level of remediation is needed for each LAA.

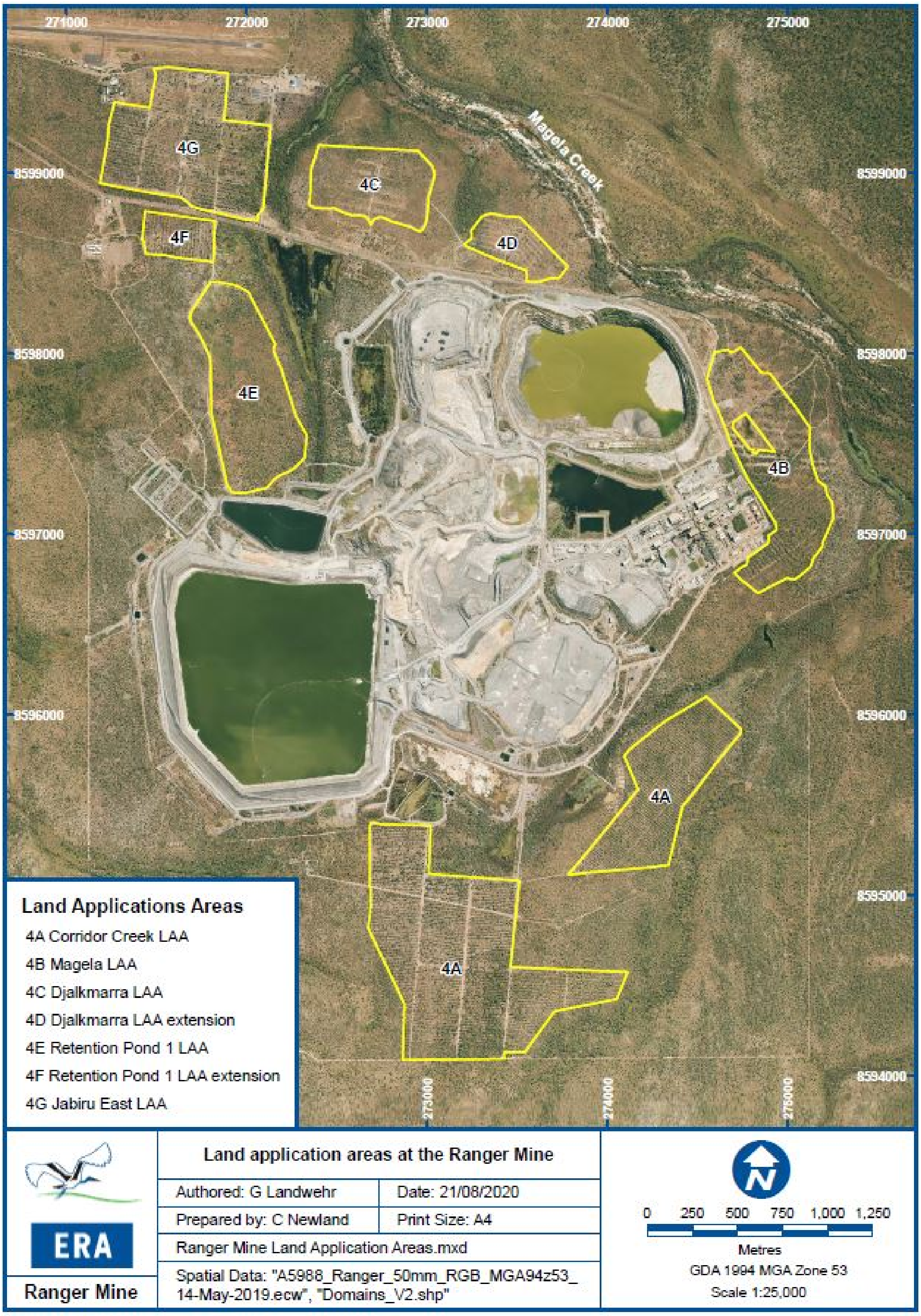


Figure 5-90 Land Application Areas at the Ranger Mine

Scope and Approach

The scope of this project is to:

- cohesively link all historical LAA soil investigations by undertaking a literature analysis;
- create a database of all LAA soil data available to enable analysis of results;
- understand the contamination and mobility of COPCs at each LAA;
- undertake a BPT assessment for each LAA to determine, if required, the level of remediation to be undertaken to ensure ALARA. BPT assessments will take the source-pathway-receptor exposure model into account when determining the final management option.

No additional sampling is planned at this stage to further inform this project. The current dataset is considered to be sufficient for informed decisions regarding the level of remediation (if any) required for each LAA. Historical LAA and 'background' soil data (up to 6m BGL) will be used to develop LAA conceptual site models and spatially map sediment concentrations.

The outcomes of the report will be reviewed and reported internally through the Water and Closure Operational Forum. Data will also be presented to stakeholders at the RCCF and/or MTC; whichever is sooner. Updates will be included in future updates of the MCP and KKN closeout evidence will be reported to stakeholder groups and ARRTC.

5.5.2.5 Non-aquatic contaminated sites sampling

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- RAD2. Radionuclides in aquatic ecosystems
- RAD9. Impacts of contaminants on human health

A comparative assessment of COPCs and their respective source(s) (e.g. waste rock, tailings/pore water, groundwater, soils) is needed, including consideration of any remnant 'hotspots' that may be present post-rehabilitation of the Ranger Mine. This information contributes to whole-of-site contaminant transport modelling to predict the Pit 3 backfill, post-closure water quality, and will inform the rehabilitation and risk management of the site.

Contaminated sites have been identified across Ranger Mine since the early 2000s (Hollingsworth, 2006) and since then, a significant number of targeted contaminated land assessments have been undertaken previously on the RPA at known contaminated sites between 2006 and 2016. Although the focus of previous assessments was predominantly on identifying groundwater contamination, soil and sediment profiles have also been assessed at known contaminated sites to define the lateral extent of contamination in the soils and sediments on the RPA.

The contaminated sites have been documented in a *Contaminated Land Risk Register* which has been developed and maintained by the site environment team at the Ranger Mine, in accordance with the operational *Hazardous material and contamination control plan* (ERA 2016). The *Contaminated Land Risk Register* identifies all sites where activities have occurred that have the potential to contaminate land. Section 9.4.1 describes the contaminated sites domain including the specific contaminated sites, grouped into major site areas, based on location and proposed remediation strategies. The major site areas are shown in Figure 5-91, Figure 5-92 and Figure 5-93.

As part of the feasibility study undertaken in 2018, a review of the *Contaminated Land Risk Register* was undertaken to provide a register (at that point in time) suitable for closure planning purposes. The review involved ensuring all areas of potential contamination were captured as well as aligning historical investigations undertaken to date, thereby developing a current knowledge based of site contamination. Sites were also classified according to risk (costs of remediation). Any new potentially contaminated land as a result of operational activities occurring after this review will be added to the *Contaminated Land Risk Register* by the site environment team and will be incorporated into closure investigations if required.

Following this review, a *Plume and contaminated site management plan* was developed during the feasibility study. The plan describes future work (site assessments and BPT assessments), post remediation validation assessments and post-closure monitoring. This plan was further reviewed for appropriateness in April 2019 to confirm whether broad remediation statements made during the feasibility study were suitable, i.e. supported by outcomes of previous studies and outcomes of the feasibility study, and a gap analysis was completed. Areas identified during the gap analysis as having insufficient data to adequately determine a remediation treatment option were detailed, including depth and COPCs for further investigation.

Additionally, to support the post-closure solute transport modelling, an assessment of potential groundwater contamination sources is underway and will be detailed in the Pit 3 Closure application. These potential groundwater contamination sources are the Process Plant Area, TSF, LAAs, and the waste rock stockpile of the operational period.

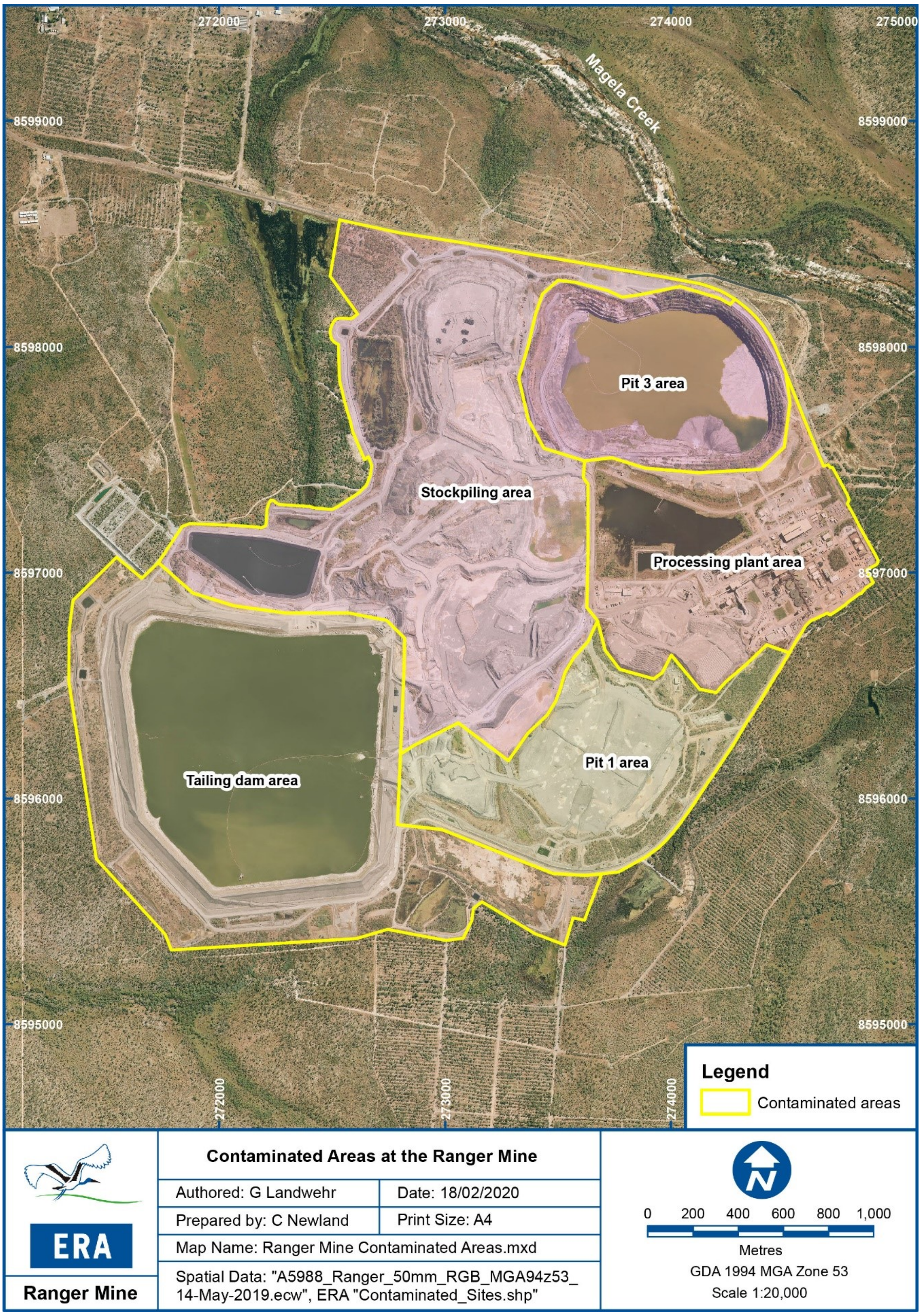


Figure 5-91: Ranger Mine area boundaries

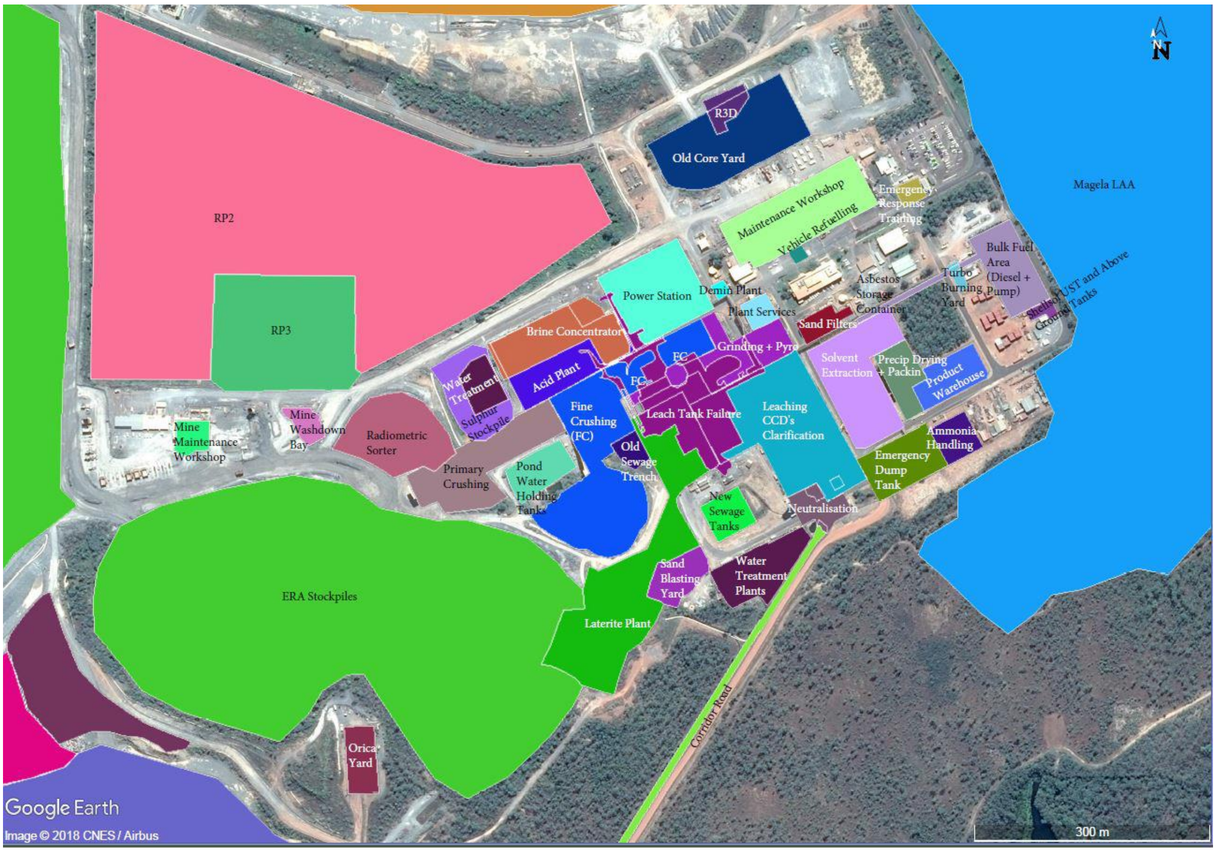


Figure 5-92: Processing plant area – contaminated sites register

Scope and approach– processing plant - soils

In order to understand the current state of the soils around the RPA, a contaminated sites drilling program was executed between November 2019 and January 2020 to sample soils, install groundwater monitoring wells and re-develop existing monitoring wells at targeted areas defined by the gap analysis undertaken in April 2019. A summary of knowledge gaps for the selected sites is summarised in Figure 5-37

The identified sites were sampled between November 2019 and January 2020 in accordance with the Australian Standards (AS 4482.2-1999 and AS 4482.1-2005). Soil samples were obtained using a drill rig equipped with a hollow stem augur. Soil conditions and descriptions were logged in the field and samples analysed for COPCs and other parameters of interest.

IN selecting the locations of the soil bores drilled as part of the drilling program (Figure 5-94 to Figure 5-100) ERA took into consideration, historical data and known gaps (as detailed in Table 5-35), nature and source of the contaminants and hydrogeology for each site.

A Sampling Analysis Quality Plan (SAQP) was developed to document the purpose and rationale of each location, target depth, sampling interval and COPCs of interest (ERA, 2020).

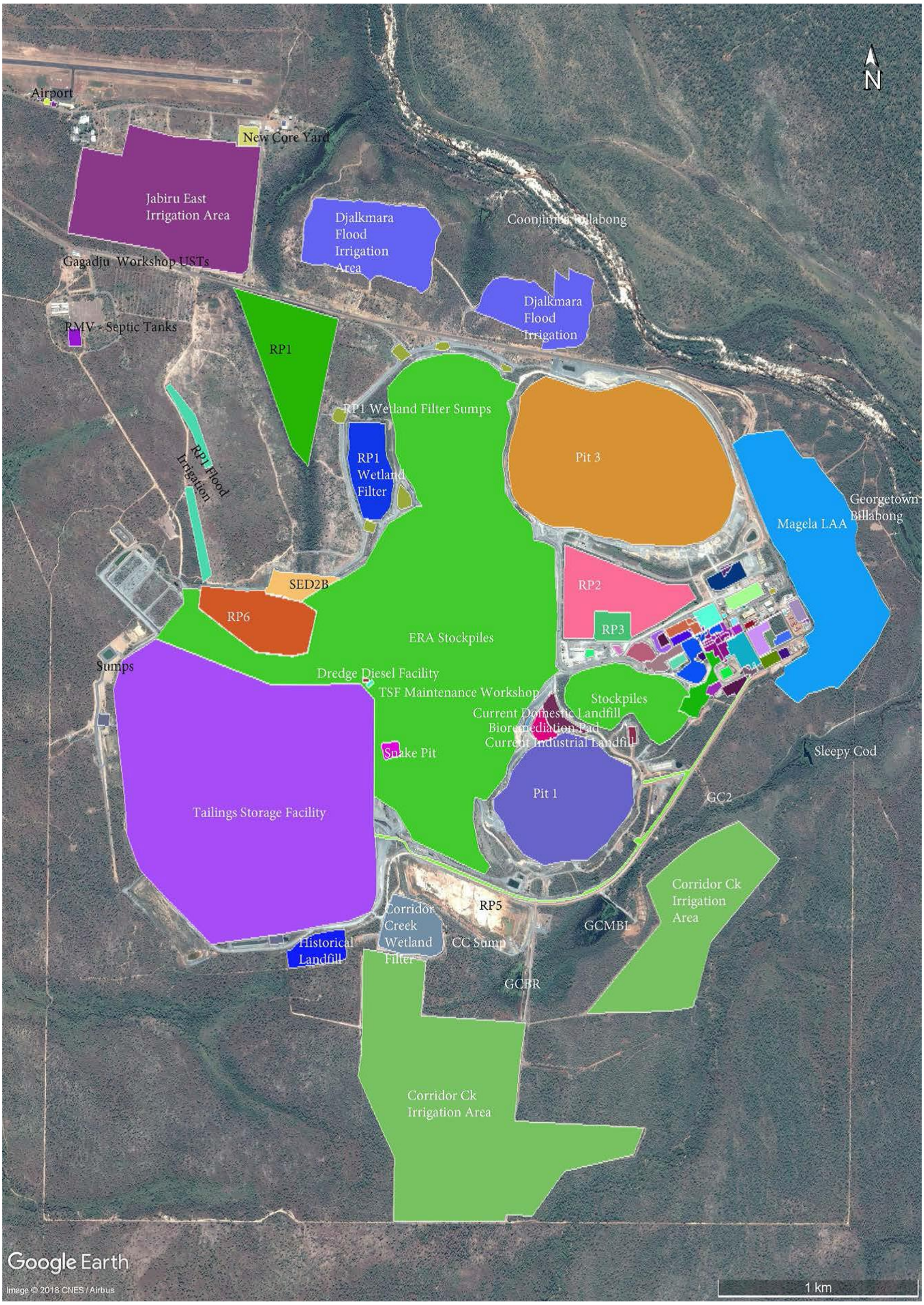


Figure 5-93: Major site area boundaries – contaminated sites register

Table 5-35: Summary of targeted Site Contamination Assessment of Knowledge Gaps.

Site	COPC	Knowledge Gap Actions
Historical Landfill	• TRH, BTEXN, PAH, Phenols, VOCs, Sulfate, Ammonia, Nitrate + Nitrite as N and Metals (Mn, U).	• Update data on current vertical extent of COPCs in soil. The primary depth of concern is within the top 4.5 m.
Emergency Dump Tank	• TRH, PAH, VOCs, Sulfate and Metals (Mn, U).	• Establish site-specific data to determine the vertical extents of COPCs in soil at the emergency dump tank. Depth of assessment up to 10 m BGL.
CCD Circuit	• Metals (Fe, U, Mn), pH, Sulfate, EC, TRH, cations and anions.	• Determine vertical extents of COPCs in soil beyond a depth of 3.65 m BGL.
Sulfur Stockpile and Acid Tank	• Metals (Mn, Cr, U, Fe), pH, sulfate and TRH.	• Determine vertical extents of COPCs in soil beyond a depth of 4 m BGL.
Power Station	• TRH, BTEXN, PAH, Sulfate, PCB, Metals (Mn + U)	• Determine vertical extents of COPCs in soil beyond a depth of 4.5 m BGL.
Shellsol Tank	• TRH, BTEXN, PAH and Phenols	• There is a limited data on vertical extents of COPCs in soil beyond a depth of 3.25 m BGL.
Bioremediation Pad	• TRH, BTEXN, PAH, VOCs and radionuclides	• There is currently a poor understanding of the vertical extent of COPCs in soil beyond depth of 0.4 m BGL.

Figure 5-94: Locations of soil bores drilled in the processing area at Ranger Mine

Figure 5-95: Locations of soil bores in the historic landfill, bioremediation pads and TSF walls at Ranger Mine.

Figure 5-96: Location of boreholes at the historic landfill area

Figure 5-97: Location of soil bores at the emergency dump tank and CCD circuit areas

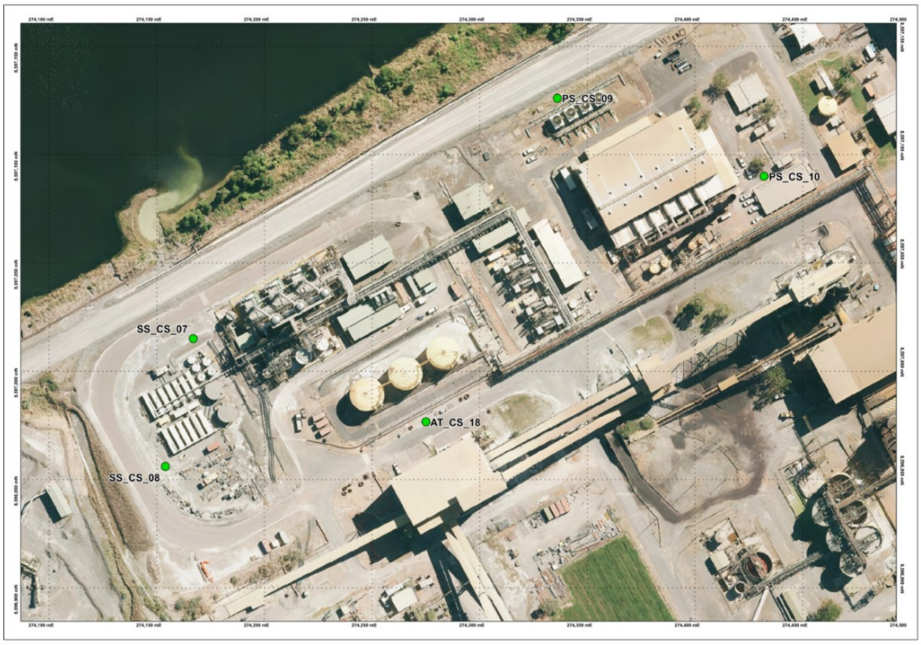


Figure 5-98: Location of soil bores at the former sulfur stockpile, acid tank and power station areas.

Figure 5-99: Location of soil bores at the Shellsol underground and above ground tanks

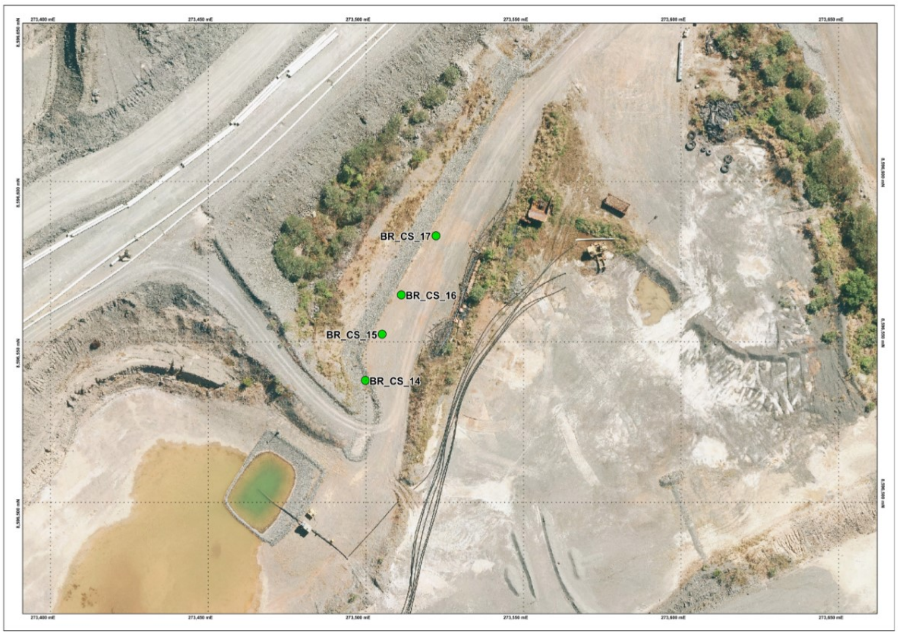


Figure 5-100: Location of soil bores at the bioremediation pad

Evaluation and reporting outcomes – processing plant: soils

Overall, contamination was found to be localised to infrastructure and top soils with limited groundwater impacts that are slow moving.

The following observations were made:

- Historic Landfill Area - Cr and Mn concentrations in LF_CS_01 showed increasing concentrations with depth with all other analytes displaying a decreasing or stable trend with increasing depth.
- Dump Tank Area - The profile was approximately linear in the area with no significant variation to the concentrations observed for most COPCs with increasing depth. An inverse relationship was observed between NH3-N and NO3-N indicating evidence of microbially mediated transformation processes in the soil.
- Counter Current Decanter (CCD) area - All COPCs concentrations except for those for NH3-N exhibit a decreasing trend down bore at the soil bores with the highest levels of contamination observed in the top 2.0 m of the soil profile.
- Former Sulfur Stockpile and Acid Tank – General decrease in COPC concentrations from surface to a depth of 5.0 m BGL with the steepest decrease observed at 1.5 m BGL. Cr trends increased with increased depth in SS_CS_07. At depths greater than 5 m BGL, Cu concentrations increased with depth in SS_CS_07 and Mn concentrations increased with depth in SS_CS_08.
- Power Station Area – Most of the COPCs that were analysed showed a sharp decrease at depths greater than 1.5 m BGL to stabilise at depths deeper than 2.0 m BGL. Hydrocarbon contamination was only detected in one soil bore at a depth of 0.1 m BGL in the power station area and in two bores to a depth of 1.5 m BGL at the former bioremediation pad area. There is no observable PCB contamination in the area.
- Shellsol Tank – There were no hydrocarbon impacts identified in the area and concentrations of COPCs were observed to gradually decrease with increased depth.
- Bioremediation Pad – Hydrocarbon impacts were identified at BR_CS_16, with a spike at 0.5 m BGL, persisting at low levels to a depth of 1.5 m BGL. Low level hydrocarbon contamination was detected at 0.1 m BGL in BR_CS_15. The contamination appears localised. No other impacts were observed at other bores sampled from this site.

Scope and approach– processing plant: groundwater

The Ranger Conceptual Model (INTERA 2016) noted that 38 contaminated sites have been identified in or near the processing plant area. Based on the dataset available, it appears that the majority of impacts to groundwater exist beneath the western portion of the process plant area, with lower levels of impact identified across the rest of the process plant and between the process plant and receptors to the south and east. The highest concentrations of sulfate in groundwater were identified in groundwater from bores 3B, 35 and 47 within the process plant area, extending to the south-east to OA09, OB241 and OB242; these concentrations are partially delineated by OA08, OA11, and OA10. The impacts of manganese in groundwater were not delineated between potential sources and receptors. Concentrations of COPCs above background in groundwater from bores towards Corridor Creek to the south and Georgetown Creek and GTB to the east are considered to be most likely to have been derived from irrigation activities in the former Magela LAA area (ERM 2020b).

Further review of the contamination extent and profile is underway to support the post-closure solute transport groundwater modelling for the Pit 3 closure application. This includes analysis and interpretation of all available groundwater laboratory analysis data from the processing area to support development of a three dimensional profile of contamination profile within the Leapfrog geologic modelling software that can then be incorporated into the solute transport modelling. Additionally, the contamination profile will be included within the uncertainty analysis of the groundwater modelling with results to be presented in the Pit 3 closure application and subsequent MCPs. A map showing all the bores with data that are being used to inform and develop the contamination profile for inclusion in the groundwater modelling is shown in Figure 5-101.

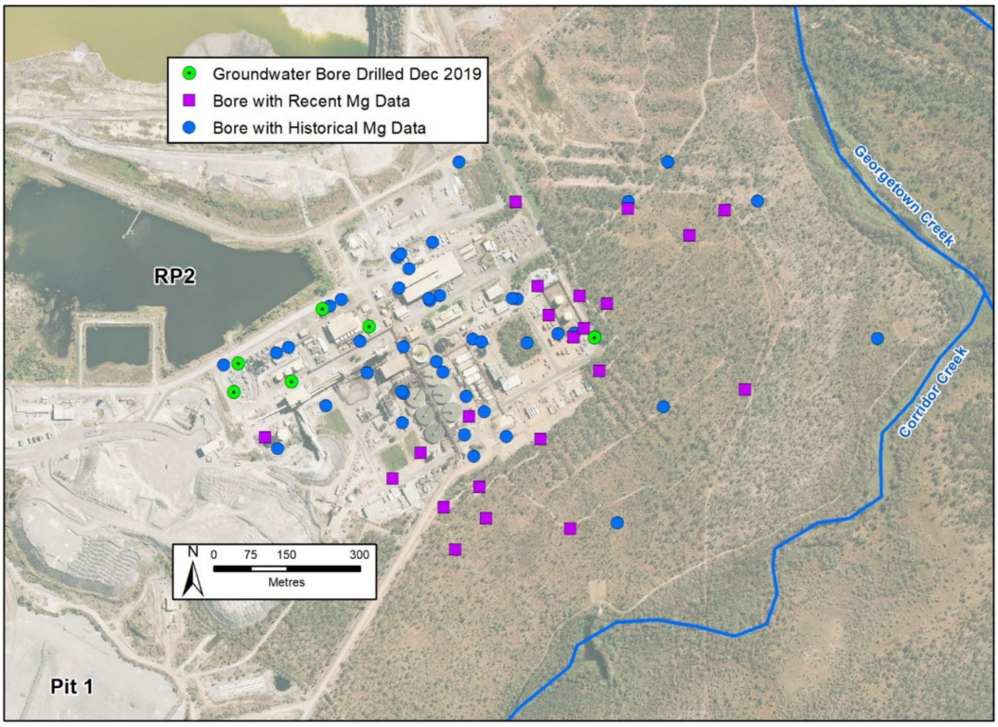


Figure 5-101 Monitoring bores used to inform development of groundwater contamination profile at the Processing Plant Area

Scope and approach– TSF

Gradual seepage from the TSF, since the time of its construction, has resulted in the formation of a groundwater contamination plume. The extent and behaviours of the plume have been investigated repeatedly over the years (Weaver 2010). Studies into the groundwater contamination below the TSF have been undertaken in order to support both the MTC application Ranger Mine Tailings Storage Facility – Subfloor Material Management and the Ranger Mine post closure solute transport modelling. The key elements of the studies involved sampling and analysis of the subfloor material below the TSF, a review of historical hydrogeological investigations, and a review of all available groundwater data surrounding the TSF.

To support the subfloor material management application INTERA (2020) modelled the extent and profile of the magnesium contamination below the TSF. This was undertaken by integrating and interpolating the available data within the Leapfrog geologic modelling software as shown in Figure 5-102.

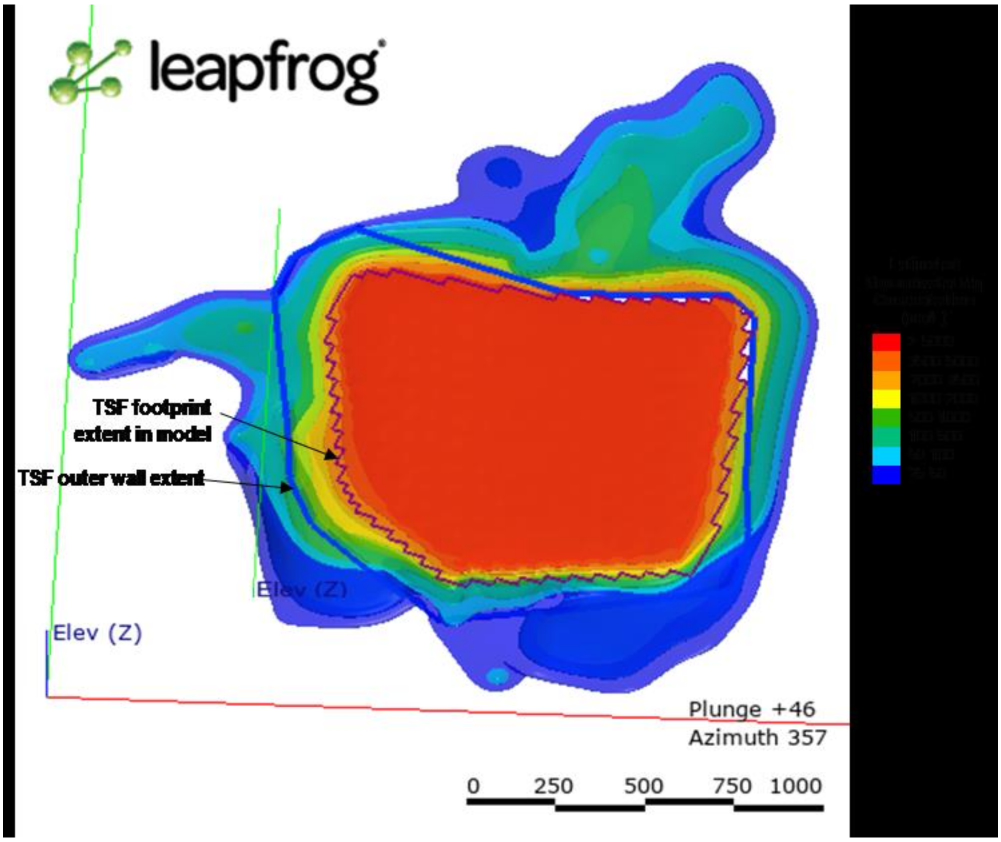


Figure 5-102 INTERA (2020) Leapfrog TSF Mg plume concentration and lateral extent

To support the post closure solute transport groundwater modelling with uncertainty analysis (section 5.5.2.10), the study to support the subfloor material management application was reviewed with the inclusion of more recent groundwater data, and laboratory analysis results from the subfloor-drilling program. The objective of this review is to include all COPCs required to be modelled to support assessment against the Ranger Mine closure criteria, and to define the uncertainty analysis parameterisation. The outcomes of this study will be detailed in the Pit 3 Closure application and future MCPs.

Scope and approach– Land Application Areas: soils

See section 5.5.2.4

Scope and approach– Land Application Areas: groundwater

Contamination that will be present in groundwater below the LAAs at closure is currently under investigation by INTERA. The purpose of this investigation is to define what COPCs will be above background concentrations in groundwaters proximal to the LAAs at closure. The results will be included as a source term within the post-closure solute transport modelling. Review of both historical and recent groundwater bore laboratory analyses is underway to identify what contamination has historically been present and to identify any trends in the groundwater COPC concentrations with consideration for both current and historical irrigation practices. The bores being utilised for this investigation are shown in Figure 5-103

Scope and approach– waste rock landform

To develop the post closure waste rock landform source term nine bores were drilled in December 2018 and January 2019 targeting groundwater below the waste rock stockpile. Monitoring of groundwater in these bores has been undertaken to quantify any contamination that may exist below the waste rock stockpile, and validate any geochemical modelling undertaken to inform the post closure waste rock landform source term. (Section 0) Analysis of the groundwater chemistry data is underway and will be presented in the post closure solute transport modelling to support the Pit 3 closure application as well as future MCPs.

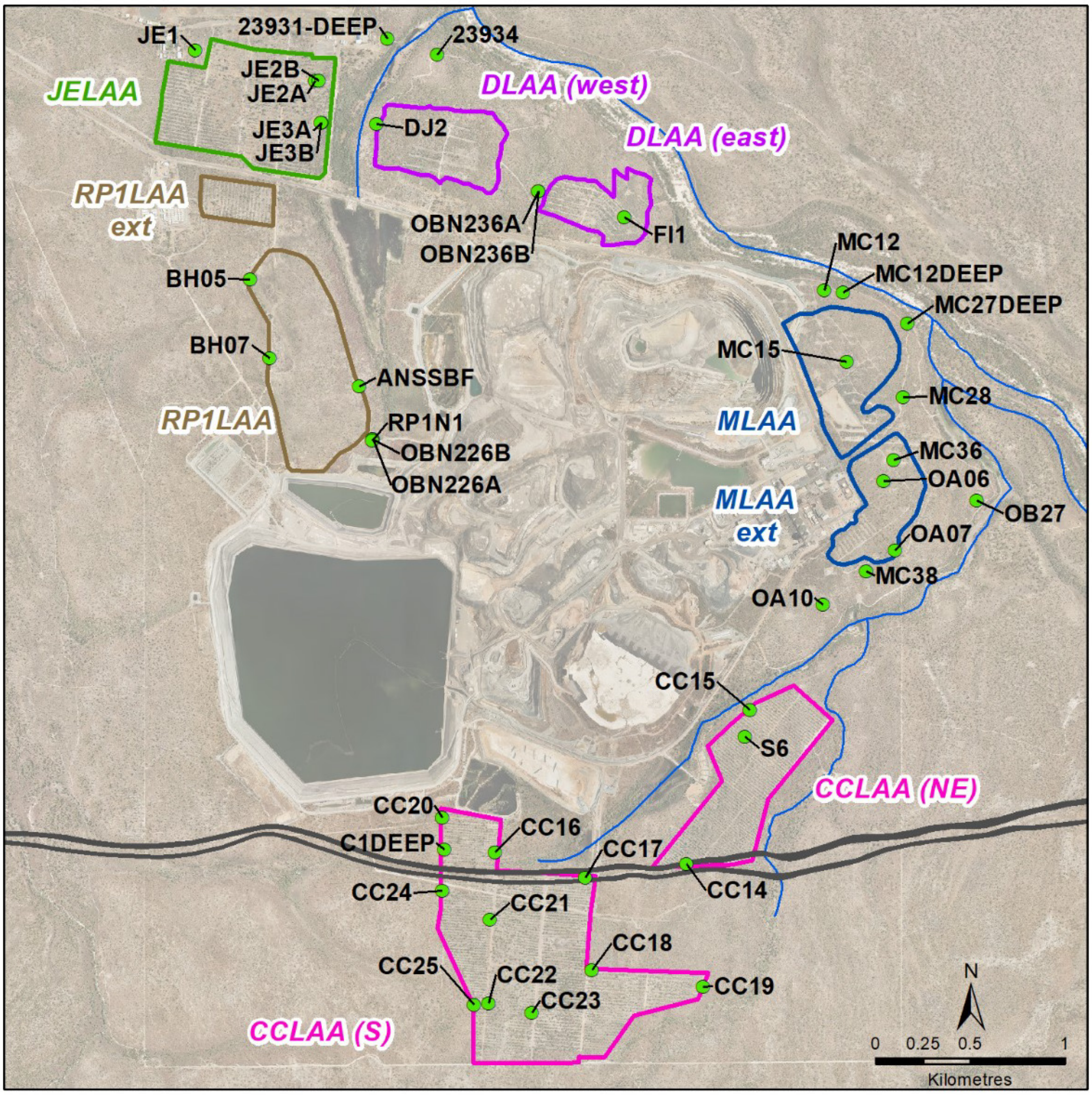


Figure 5-103 Land Application Area groundwater-monitoring bores for source term development

5.5.2.6 Stockpile drilling

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- WS2. Predicting transport of contaminants in groundwater

Monitoring of the bores, drilled as part of the stockpile drilling program, is required to inform the updated waste rock source term and subsequently the post-closure solute transport model with uncertainty analysis. The objective of this project is to collect groundwater level and chemistry data to inform the assessment of groundwater movement under the waste rock stockpiles and trial landform as well as support the validation of the waste rock source term for the post-closure solute transport model. See Section 0.

Scope and approach

This project required the regular collection of groundwater level and chemistry data from 9 bores drilled in the waste rock stockpiles and 1 bore in the trial landform (Figure 5-104). This project commenced after the completion of the bore drilling in January 2019. The project ceased mid 2020 once the data were collected.

Monitoring was undertaken by the site Water Management team on a monthly occurrence using their existing groundwater monitoring SAQPs.

Figure 5-104 Location of 9 stockpile bores and 1 TLF bore

Water level data will be provided to ERA's groundwater consultants on a regular basis when it is collected from the field. Details of the reporting outcomes will form part of the validation process for the post closure solute transport modelling with uncertainty analysis.

5.5.2.7 Solute source term update

This project relates to one KKN:

- WS1: Characterising contaminant sources on the RPA

A critical input to the post-closure solute transport modelling is the solute source term conceptual model. The solute source term conceptual model details the contaminants present, and the concentration or mass of the contaminants present for all the major contaminated locations on the RPA. The solute source term also includes reference to any geochemical processes that result in mobilisation of COPCs from the waste rock landform.

INTERA have previously developed a solute source term conceptual model for the major contaminant sources on the RPA for the 2014 and 2016 post closure solute transport modelling. The existing source terms within the solute source term conceptual model requires update following the availability of additional data. Additionally, new source terms are required to be developed for solute source areas not previously included in the post-closure solute transport modelling, these include the LAAs, processing plant area and TSF.

Solute source term conceptual model update in itself does not directly address any specific Environmental Requirements, however it does form a critical part in a number of groundwater and surface water studies that do.

The objective of this study is to define all sources of contamination on the RPA for inclusion in the post-closure solute transport modelling. Detail the COPCs present, the concentration or mass of the COPCs and any geochemical processes relevant to the mobilisation of COPCs.

The output of the study will feed directly into the post-closure solute transport modelling.

Scope and approach

INTERA have been engaged to update the existing solute source term for the post closure solute transport modelling. Additional scope has been included for the assessment of any new source terms that have not been previously included in the post closure solute transport modelling. These include the LAAs, processing area (mill, power station, CCDs, hydrocarbon storage, historic landfills, etc), TSF, and wetland filters.

The project consists of a desktop analysis of existing investigations, data and studies.

The scope of the source term work as a whole (original waste rock and tailings-derived materials scope and new additional scope) involves the following:

- update the conceptual understanding of concentrations of COPCs from waste rock
- update the conceptual understanding of concentrations of COPCs associated with tailings-derived materials
- update the INTERA (2016) conceptual understanding of groundwater impacts for the areas of interest/concern not associated with waste rock or tailings-derived materials

- estimate the COPC source concentrations suitable for use as inputs in simulating/calculating solute loading to creeks for waste rock, tailings-derived materials, and areas of interest/concern not associated with waste rock or tailings-derived materials
- draft and finalise a source term concentration report that will include all COPC sources at the minesite. The report will separate the COPC sources by the various primary materials/areas associated with the COPCs. For each material/area, descriptions of the data reviewed, assessments conducted, assumptions used, and results obtained will be provided.

5.5.2.8 Literature review on contaminant mobility

This project relates to multiple KKNs:

- WS1: Characterising contaminant sources on the RPA
- WS2. Predicting transport of contaminants in groundwater
- WS3. Predicting transport of contaminants in surface water

Factors influencing contaminant mobility in the sources and several pathways are covered by different KKNs. Details relevant to each KKN are described below.

Scope and approach

Undertake a desktop literature review summarising the site specific studies of contaminant mobility in water, sediment, soils, waste rock and tailings in the context of each KKN question and identify factors controlling mobility which need to be understood.

Results and conclusion

Literature reviews are attached to KKN closeout forms for review by relevant external stakeholders. Acceptance of the literature reviews results in KKN closeout.

KKN	Compartment	Why factors controlling mobility need to be understood	Status
WS1b	Sources	Contributes to whole-of-site contaminant transport modelling to predict post-closure water quality. Inform the rehabilitation and risk management of the site.	Literature review completed and attached to KKN closeout form for stakeholder review. Any further requirements for information can be addressed within projects against contaminant transport modelling. SSB feedback was to review need for additional information once final scenarios for predicting post-closure
WS2b	Groundwater pathway	Is conservative modelling or reactive modelling required? What factors are important?	
WS3c	Surface water pathway		

KKN	Compartment	Why factors controlling mobility need to be understood	Status
			surface water quality are completed. KKN closeout pending this.
WS3g	Surface water –sediment interactions	To determine if closure criteria will protect both environmental compartments	U & S identified as sediment CoPEC (contaminant of environmental concern). U rehabilitation standard protects both sediment and water. SO_4 rehabilitation standard derived to protect ASS forming.
WS3e	Groundwater – surface water interactions	Potential to limit or increase their concentrations from groundwater to surface water. Which could affect surface water quality predictions. *Note the KKN question focuses on physical influences, not chemical aspects. Jenny Stauber suggested including chemico-physical drivers at Nov 2019 meeting.*	KKN WS1b closeout covers the behaviour of contaminants in sediments (the interface) and the influence of factors such as pH, oxidation, secondary mineralisation etc at the source. Reactive transport drivers have been summarised in KKN WS2b & WS3c closeout. Reactive transport modelling discussed wrt WS2b includes the near surface layers.
WS5b	Bioavailability and toxicity of sediments contaminants	Bioavailability mentioned in KKN title not in question. Question is about the Influence of toxicity modifying factors to enable (U) guideline value to be adjusted if sediments different from Gulungul Billabong.	Sediment is one of the sources reviewed in KKNWS1b closeout. Reports on U behaviour in sediments passed to SSB who are closing this KKN.
RAD9b	Concentration factors for bushfood	Quantify transfer from the environment (e.g. soil and water) to food items.	This is a SSB KKN.

5.5.2.9 Update groundwater solute transport modelling and conceptual model

This project relates to multiple KKNs:

- WS2. Predicting transport of contaminants in groundwater
- WS3. Predicting transport of contaminants in surface water
- RAD2. Radionuclides in aquatic ecosystems

Post-closure solute transport modelling is required to understand the mobilisation of COPCs from the RPA to the surrounding environment. This includes the mobilisation of contaminants from the storage of tailings, brines and contaminated material in the backfilled pits, from the landform waste rock, and from the LAAs located around the mine. The post closure solute transport modelling is split into multiple phases to support project execution.

The first phase of post closure solute transport modelling is to update to the Ranger conceptual groundwater model that was originally developed in 2014 (INTERA 2014a, 2014b) and then updated in 2016 (INTERA 2016).

In parallel to the update to the Ranger Conceptual Model, updates are required to specific inputs for the modelling, including 1250-01 Background COPCs, and 1250-08 Solute Source Area. The second phase is the 1250-11 post-closure solute transport modelling with uncertainty analysis.

The output of 1250-11 post-closure solute transport with uncertainty analysis is a key input to 1260-01 Surface Water Modelling.

Regular updates on the state and progress of the solute transport modelling are provided to stakeholders at MTC meetings and Ranger Closure Collaborative Forums. Further consultation is undertaken regularly with the SSB throughout the modelling process.

Solute transport modelling is required to directly address or support the Environmental Requirements (ER’s).

The objective was to update to the Ranger conceptual groundwater model that was originally developed in 2014 (INTERA 2014a, 2014b) and then updated in 2016 (INTERA 2016). The update will be reviewed by stakeholders (SSB) prior to progressing the post closure solute transport modelling. This project is complete and an updated Ranger Conceptual Model is developed (INTERA 2019a). The outputs of the update of the conceptual model and solute transport modelling are required to support the Pit 3 backfill MTC application and address KKN WS2.

Scope and approach

The scope for the update to the Ranger conceptual groundwater model consisted of:

- review all available historical models, studies and projects on groundwater modelling, groundwater flow, and hydrogeological conceptualisations
- incorporate all recently available data, including groundwater monitoring, hydrogeological drilling
- review the exploration drilling data set to further refine the weathered zone and geological structures within the conceptual model
- update the Ranger Conceptual Model and undertake transient model calibration of the numerical model
- prepare a detailed report describing all updates and calibration of the Ranger Conceptual Model

- undertake head recovery modelling to predict when groundwater levels will recover to steady state across site. Include preliminary particle tracking modelling to understand solute transport pathways

Regular engagement was undertaken with stakeholders throughout the model update process and following completion. INTERA presented outcomes of the model update and calibration at ARRTC May 2019, follow-up review discussions occurred with the SSB. The revised report was issued in October 2019.

This project is complete and the updated Ranger Conceptual Model has been developed.

5.5.2.10 Post closure solute transport modelling with uncertainty analysis

This project relates to multiple KKNs:

- WS2. Predicting transport of contaminants in groundwater
- WS3. Predicting transport of contaminants in surface water

Post-closure solute transport modelling is required to understand the mobilisation of COPCs from the RPA to the surrounding environment. This includes the mobilisation of contaminants from the storage of tailings, brines and contaminated material in the backfilled pits, from the landform waste rock, and from the LAAs located around the mine. The post closure solute transport modelling is split into multiple phases to support execution.

The first phase of post closure solute transport modelling is to update to the Ranger conceptual groundwater model that was originally developed in 2014 (INTERA, 2014) and then updated in 2016 (INTERA 2016). The update to the groundwater solute transport conceptual model was completed in October 2019 by INTERA (Section 5.5.2.9).

Following the update to the conceptual model, multiple projects have commenced to support the update to the solute source area / conceptual model update, these including the Background COPCs in groundwater study and drilling campaigns (contaminated sites, TSF, stockpiles etc).

In parallel to the solute source area / conceptual model update, a study to develop a framework to link the outputs of the groundwater modelling, to the surface water modelling is underway. The aim is for a single report that summarises historical investigations, along with a review of more recent data to form a robust relationship for linking the two modelling packages together.

The post-closure solute transport modelling with uncertainty analysis forms the final step to predicting contaminant loadings from groundwater to the environment for 10,000 years post-closure. These loadings over time will then be evaluated through surface water modelling for assessment against closure criteria.

Key objectives of this project are to:

- develop probabilistic predictions of solute loading from Ranger Mine sources to Magela, Corridor, Coonjimba, and Gulungul creeks in the 10,000 years following mine closure

- calculate solute loads to the creeks for 20 COPCs: magnesium (Mg), uranium, manganese, radium-226, total phosphate, nitrate as nitrogen, total ammonia as nitrogen, polonium-210, iron, copper, lead, cadmium, zinc, chromium, vanadium, calcium, nickel, selenium, aluminium, and sulfate

Scope and approach

INTERA has been engaged to undertake the post closure solute transport modelling. A scope of work has been prepared: *Scope of Work: Predictive Modelling of Ranger Post-Closure Solute Loading with uncertainty Analysis,* (INTERA 2019b). The scope of work (INTERA 2019b) outlines a two phase approach to the study including key deliverables and regular engagement with stakeholders.

Engagement with stakeholders is undertaken at approximately 6 weekly intervals, or when key milestones have been reached. These engagements are opportune times to discuss progress to date, clarify any concerns or comments, provide opportunity for feedback on the process and outline the tasks ahead.

Following completion by INTERA of the post-closure solute transport modelling with uncertainty analysis, the report will be provided to stakeholders for review and feedback.

5.5.2.11 Surface water modelling

This project relates to multiple KKNs:

- WS3. Predicting transport of contaminants in surface water
- RAD2. Radionuclides in aquatic ecosystems
- RAD9. Impacts of contaminants on human health

A surface water model of the RPA is required to predict concentrations of COPCs in surface waters present on the RPA and downstream of the post closure phase (Section 5.4.4).

The key objective of the study is to develop a surface water model that provides predictions of flow and COPC / sediment concentrations in Gulungul, Corridor, and Magela Creeks on the Ranger Project Area and downstream off the RPA after closure of the mine.

The output and results of the surface water model will form part of the Pit 3 backfill application.

Scope and approach

- Project start-up: collate and review all data pertaining to:
    - topographic information, landform profiles, LIDAR surveys, cross sections, billabong surveys.
    - review previously developed sitewide surface water models

- local area surface water models, including OPSIM, rainfall runoff, 2D hydraulic models.
- landform Evolution Modelling (LEM) for time series sediment loading to creeks.
- erosion and sediment control features of the sites post closure
- draft closure criteria
- all available rainfall, flow, water quality data for the waterways of concern
- rating curves for all waterways of concern
- records of historical COPCs and sediment discharge from the site
- aerial photographs
- review previous studies and reports.

- Model conceptualisation
  - develop and refine the modelling framework for the study
  - develop modelling concept for each COPC and suspended sediment
  - develop Groundwater to Surface water model integration method
  - develop tech memo summarising the available data and proposed model concept.
  - update stakeholders in regard to surface water modelling progress
- Configuration and calibration of the surface water model
  - build the model in accordance with the framework
  - develop backwater billabong relationships
  - develop climatic sequences for calibration simulations
  - create COPC and sediment load inputs files
  - calibrate model to reasonably match recorded stream flows
  - develop modelling methodology for each COPC and suspended sediment
  - calibrate model to reasonably match recorded water quality data
  - undertake model verification
  - prepare Configuration and Calibration Report

- update stakeholders in regard to surface water modelling progress
- stakeholder review of surface water configuration and calibration
- updates to address key/critical stakeholder feedback

- Application of surface water model
  - model agreed surface water model scenario cases
  - review results of model scenarios
  - prepare visualisation maps and graphics to effective communicate results
  - prepare surface water modelling results report
  - present modelling results to stakeholders
  - stakeholder review of surface modelling results
  - provide final report of surface water model results

Results and conclusion

The modelling was completed by Water Solutions in May 2020. A preliminary report has been provided which details the configuration and calibration of the model along with preliminary predictions. ERA has identified that further work is required to refine the model configuration and calibration. Additionally preliminary feedback from stakeholders is that further work is required on key elements of the model including downstream calibration and groundwater to surface water interaction. ERA is currently awaiting feedback from stakeholders on the scope of work of the final surface water model.

5.5.2.12 Surface water groundwater interaction

This project relates to KKN:

- WS3. Predicting transport of contaminants in surface water

Understanding and quantifying groundwater to surface water interaction forms a key component for the linking the groundwater solute transport model to the surface water model. The groundwater to surface water interactions relate to the timing, and location of groundwater flow and in turn potential for solute transport from groundwater into the receiving environments. Understanding this relationship and accurately representing it in the modelling is vital to accurately predicting the possible contamination concentrations in the receiving environment.

The objective of the study is to develop a report summarising the following:

- an understanding of the variations in groundwater discharge volumes into creeks over time relative to the surface water flow rates and volumes using a groundwater model that has greater refinement in spatial and temporal discretisation
- an evaluation of concentration data from groundwater bores and surface waters in conjunction with the model results to develop an improved conceptualisation of groundwater / surface water interaction and variation in surface water concentrations as surface water flows decrease when the wet season progresses into the dry season.

Scope and approach

The project requires the following:

- review historical studies into groundwater to surface water interactions, both regional and local scale
- review existing data sets including groundwater and surface water levels, and water chemistry to understand changes in hydraulic gradients adjacent creeks over the wet season
- review radon in groundwater studies to further support model conceptualisation and development
- develop updated groundwater to surface water conceptualisation utilising all available data
- test and validate updated conceptualisation within high spatial and temporal resolution numerical groundwater model
- develop a groundwater to surface water flow relationship that can be implemented in OPSIM to support the surface water modelling
- review and interpret data from the completed fieldwork
- multiple engagement sessions between groundwater and surface water modelling consultants have occurred to discuss and refine model integration linkage

5.5.2.13 Acid Sulfate Sediments management

This project relates to KKN:

- WS5. Determining the impact of contaminated sediments on aquatic biodiversity and ecosystem health
- RAD9. Impacts of contaminants on human health

Observed acidification events in Coonjimba Billabong (located on the RPA) during the early-wet seasons for the past several years indicate that on-site sediments may present a source of acidic water, metals and sulfate.

Assessment of recent water quality in Coonjimba Billabong, review of past studies of sulfate behaviour and acid sulfate soils (ASS) at Ranger and naturally in the Magela Catchment, and sediment studies were undertaken to understand the drivers and extent of the ASS issue in Coonjimba Billabong (Esslemont & Iles 2015, Esslemont, 2016). Baldwin (2017) reviewed these reports and other information, made several recommendations, and suggested a limit of 10 mg/L of sulfate in waters to protect against the development of ASS. The SSB adopted this value as the rehabilitation standard to apply at the mine lease boundaries.

Baldwin (2017) recommended a series of laboratory and modelling studies be undertaken to determine the persistence (and associated risk to the environment) of ASS at the Ranger Uranium Mining Site. This lead to the KKN question describing the need to predict sulfate budgets for the billabongs (i.e. Coonjimba, Georgetown, Gulungul) to assess the risk of acid sulfate sediment formation.

ERA contracted ERM to develop a preliminary conceptual model of ASS at Ranger (See 5.5.2.3)

ASS sediment sampling is planned for 2020 based on the conceptual model and Baldwin 2017 recommendations.

Scope and approach

Based on the results of the conceptual model and field assessments, a risk assessment of domains across the minesite will be undertaken to understand the future ASS occurrences/persistence in the billabongs. If the risk assessment indicates sulfate in water needs to be reduced or ASS sediments treated, trial mitigations and remediation options will be investigated.

5.5.2.14 Surface water pathway risk assessments (release pathways onsite)

This project relates to multiple KKNs:

- WS5. Determining the impact of contaminated sediments on aquatic biodiversity and ecosystem health
- RAD9. Impacts of contaminants on human health

There is a need to assess what, if any, decommissioning/remediation is required for on site billabongs. The aim of any such work will be to minimise disturbance within the context of impacts that are ALARA-BPT.

Numerous studies have been completed, commenced or planned to understand what contamination exists, or is expected on the RPA following closure. The outputs of these studies will be used to understand the risks posed to the primary environmental objectives and the options for management of the risks.

This risk assessment of the surface water pathway on the RPA will use the results of projects against several KKNs, particularly those predicting contaminant concentrations in surface water and sediment (WS1, 2 and 3) and the effect of those concentrations to the ecosystem (WS5 & WS7) and human health (RAD9).

5.5.2.15 Eutrophication risk study

This project relates to KKN:

- WS6. Determining the impact of nutrients in surface water on aquatic biodiversity and ecosystem health

Sources of nutrients

There are three major sources of trace metals and nutrients to the Magela Creek system: natural (rainwater and pristine catchment), the Ranger uranium mining operation, and the Jabiru township (Hart *et al* 1986b).

The sources of nutrients at Ranger to the water management system are from; waste rock, ammonia and phosphate (in lime) added to the mill process circuit, residual nitrates from blast residue in waste rock, and fertiliser application. These sources result in the following different water quality profiles for nutrients:

- ammonia is high in process water but not pond or release water
- nitrate levels are negligible, moderate and lo in process, pond and release waters respectively
- phosphate is low in all waters

The risk from nutrients has been low during the operational phase as waters are segregated and treated before directing to the release water circuit.

Load limits

Currently ERA must comply with Annual Additional Load Limits (AALL) for the discharge of NO_3-N (4.4 t/y) and PO_4-P (2.8 t/y) to Magela Creek and with NH_3-N concentration limits in Magela Creek. The load limits were set in the 1980s (Brown *et al*. 1985). No load limit was set for ammonia; only a concentration limit was set as it was considered to pose a toxicological, rather than an eutrophication risk.

Brown *et al* 1985 refers to a study of ecological risk (no report cited) as the basis for the nitrate and phosphorous AALL. However, a review of the literature indicates that the AALL allow a doubling of the natural loads recorded in the 1982-83 wet season.

Scope and approach

Desktop review:

Phase 1: Review the AALLs for relevance and suitability for deriving an ammonia AALL:

- review the literature and basis of the current AALLs
- determine if the P and N AALL remain relevant and whether ammonia data are available and suitable to derive an ammonia AALL
- derive an AALL for ammonia if available data are suitable

Phase 2: Assess post-closure eutrophication risk

- compare surface water model predictions to national default guideline values (ANZECC & ARMCANZ, 2000) and background data (1st tier assessment)
- if these are exceeded conduct a higher level risk assessment in line with national guidance (ANZG, 2018)

Results and conclusions

Reports reviewed include:

- the body of work on nutrients in the Magela system, including those describing loads and concentrations of nutrients and eutrophication status of the floodplain billabongs (eg; Hart *et al* 1986b & 1987, Hart & McGregor, 1980, 1982; Walker & Tyler 1982 & 1983.)
- reports on additions of nutrients to natural waterbodies or wetland filters at Ranger Mine (Kessel, 1983; Overall 2001, 2003)
- the basis of the AALL and past nutrient concentration limits (Brown *et al*, 1985, Hart *et al* 1986b & 1987).
- national guidelines for nutrients (ANZEEC & ARMCANZ, 2000; ANZG, 2018).
- nutrient concentrations for waste rock, brines and tailings source terms (INTERA 2016).

Although cited as being based on ecological protection (Brown *et al* 1985) the basis of the current AALL appears to be a doubling of the annual loads of phosphorous-P and nitrate-N measured during the 1982-83 wet season (Hart *et al* 1986b and Hart *et al* 1987).

Ammonia was identified as a toxicant by Brown *et al* (1985) and OSS (2002) but not as a driver of potential eutrophication. Concentration limits were therefore developed for ammonia but not load limits. The addition of nitrate to the system was noted (in OSS, 2002) as not posing a risk to eutrophication yet nitrate load limits were set.

The SSB and ERA agree that the current AALL are not suitable for closure criteria and that KKN WS6b can be closed. Final documentation is being prepared for the KKN close-out relating to Annual Additional Load Limits (AALL) to be used to inform ammonia closure criteria (WS6b).

Eutrophication risk assessment (KKN WS6c)

The ERA literature review also showed that:

- the Magela Creek system is prone to natural eutrophication and is P limited, although P additions did not necessarily induce algal growth
- algae growth could occur depending on other factors such as nitrate availability, light, pH, and plant metabolism
- annual inputs of nutrients from the creek to the floodplain is very low compared to the load contained in the floodplain vegetation, benthic sediments and rain
- the trophic status of the floodplain is not greatly affected by inputs of N and P from the catchment
- the concentrations of nutrients in the waste rock source-term, the largest source of contaminants post closure are an order of magnitude lower than national guidelines for nutrients in the tropics (ANZEEC & ARMCANZ, 2000). Only ammonia in process water and brines, which make a very minor contribution to the creek waters, are higher than the default guidelines.

ERA is working with the SSB to conduct a third tier risk review based on an expanded literature review of biological effects of nutrients and initial results of modelling predicting post closure surface water quality.

5.5.2.16 Aquatic ecosystem assessment & framework development

This project related to multiple KKNs:

- WS7. Determining the impact of contaminants in surface and groundwater on aquatic biodiversity and ecosystem health
- CT1. Assessing the cumulative risks to the success of rehabilitation on-site and to the protection of the off-site environment.

Commonwealth ERs specific to the protection of water quality and the closure of Ranger Mine specify that:

- waters leaving the RPA do not compromise the achievement of the primary environmental objectives (ER 3.1) related to protection of the people, ecosystem (biodiversity and ecological processes), and World Heritage and Ramsar values of the surrounds (ER 1 and 2).
- Impacts on the RPA are as low as reasonably achievable (ALARA) (ER 1.2e).
- The RPA must be rehabilitated to a state to allow incorporation into Kakadu National Park (NP) (ER 2.1).

The SSB has set rehabilitation standards for water quality to provide high level ecosystem protection to protect biodiversity. These are based on ecotoxicity testing of local species, mesocosm studies and field macroinvertebrate and fish studies and are designed to protect 99% of species. These standards apply at the lease boundary (Supervising Scientist 2018).

Less conservative water quality objectives are required to support the RPA goal of impacts that are ALARA. ALARA allows for some change while still ensuring the primary environmental objectives off the RPA are not compromised and the RPA can potentially be incorporated into Kakadu NP in the future. The national Water Quality Management Framework (WQMF) (ANZG, 2018) will be followed and a number of assessments conducted to identify the ALARA option and water quality objectives for aquatic biodiversity and ecosystem health on the RPA.

An ecosystem vulnerability assessment is being developed as part of this project.

Understanding ecosystem response to mine effected water

An understanding of the potential impacts of mine-related stressors on aquatic biodiversity, and the endpoints representing the primary environmental objectives values of ecosystem processes, Kakadu NP World Heritage values (including culturally sensitive species) and Ramsar values is required. Biological indicators have been identified to reflect these primary environmental objectives. These biological components (species, communities, ecosystems) vary in their sensitivity to contaminants.

Solute transport modelling is currently underway to predict the concentrations of COPCs on, and downstream of, the RPA following closure. It is important to understand what type of change might occur at different contaminant concentrations to assess the suitability of the mine closure strategy, inform BPT/ALARA assessments to apprise the need for additional mine closure activities, and support the RPA on-site water quality objectives.

Scope and approach

BMT has been working with ERA and stakeholders since 2017 in a three-phase project to:

- identify preliminary ecological and cultural endpoints for each of the primary environmental objectives (BMT WBM 2017)
- map environmental values for different water types on and off the RPA (BMT 2018)
- develop a risk-based vulnerability assessment framework (VAF) considering impact components such as duration, geographic extent and resilience, to determine how different concentrations of magnesium—potentially the most restrictive contaminant of concern—might affect these endpoints. This involved considering direct sensitivity to magnesium concentrations and indirect sensitivity via other factors affecting vulnerability, such as habitat, diet, reproduction and dispersion. (BMT, 2019).

Results and conclusions

This project is > 80 % complete. The vulnerability assessment has been conducted and a phase 3 draft report produced. New data which has since become available will be captured and considered in a re-assessment. The phase 3 report will then be updated with new biological effects information and a rescoring of vulnerability can then proceed.

Monitoring is recommended to address potential knowledge gaps identified in the aquatic ecosystem assessment & framework development. Monitoring will also provide information on the status of the aquatic ecosystem across a contaminated gradient at site to inform ALARA assessments and agreement for on-site water quality objectives/closure criteria.

The monitoring plan will be developed once the aquatic ecosystem assessment & framework development has been finalised.

5.5.3 Health impacts of radiation and contaminants

This section provides summaries of selected completed and ongoing KKN related studies linked to the theme of *health impacts of radiation and contaminants*. Some studies inform multiple KKNs and have only been included once to avoid repetition.

KKN title	Project title	Status	Section
RAD1 Radionuclides in the rehabilitated site	Radiological Impact Assessment	In Progress	5.5.3.1
RAD2: Radionuclides in aquatic ecosystems	Bushtucker Sampling Assessments	In Progress	5.5.3.2
RAD6: Radiation dose to wildlife			
RAD7: Radiation dose to the public			
	Pit 1 Radiological Monitoring	In Progress	5.5.3.3
RAD8: Impacts of contaminants on wildlife	Human Diet assessment	Planned	N/A
RAD9: Impact of contaminants on human health			

5.5.3.1 Radiological impact assessment

This project relates to multiple KKNs:

- RAD1. Radionuclides in the rehabilitated site
- RAD6. Radiation dose to wildlife
- RAD7. Radiation dose to the public

The preliminary radiological impact assessment, required to assess the radiological impact to members of public and terrestrial and aquatic wildlife is in progress and a draft report is currently under review (JRHC, 2020). The summary below provides information on the methodology followed in the assessment for members of the public and non-human biota.

Scope and approach

The following radiation exposure pathways were considered to determine the radiological impacts of the closure of the Ranger Mine on human and non-human biota:

- incremental radon concentrations
- gamma radiation levels
- radionuclide concentrations in dust
- environmental radionuclide concentrations,

All concentrations considered were above naturally occurring background levels. These incremental post closure levels were determined via source modelling as outlined below.

Atmospheric dispersion modelling of radon and particulate matter for post-closure conditions was completed in 2018 (SLR 2018a). This modelling included:

- meteorological modelling using the weather research and forecast model, and CALMET models to compile a three-dimensional meteorological dataset for the study domain
- emission estimation of radon from waste rock covered areas and the LAAs, based on radon flux rate information provided by ERA, with estimation of particulate emissions performed using published emission factors for wind erosion (DSEWPC 2012)
- dispersion modelling of the downwind dispersion of estimated emissions of particulate matter and radon using the CALPUFF dispersion model

For this study the meteorological data inputs have been compiled using the Weather Research and Forecast (WRF) and CALMET meteorological models. The meteorological dataset used in the modelling (based on the calendar year 2016) was validated by comparing key variables with the available measured data recorded at the nearest meteorological station, located at Jabiru Airport.

Radon and particulate emissions from the LAAs and waste rock area were modelled as ground level area sources based on the following emission rates:

- the radon emission rate provided by ERA for use in the modelling study was 0.5 $Bq/m^2/s$ for both the Ranger Mine footprint (waste rock areas) and the LAAs
- the total suspended particulates (TSP) emissions from the waste rock area and LAAs were modelled based on an uncontrolled emission rate of 0.4 kg/ha/hour and the following control factors to account for the reduction in dust emissions that may be

expected from increasing ground cover (trees, grasses, leaf litter etc) in the years following closure of the Ranger Mine:

- scenario 1 – immediately post-closure
- scenario 2 – 100 years post-closure.

In addition to control factors accounting for vegetation growth, the modelling also investigated the sensitivity of the modelling results to the effects of rainfall, which will act to suppress dust emissions. This was done by assuming that no emissions occurred on days with greater than 5 mm rain, based on data recorded at Jabiru Airport during 2016 (i.e. during the same meteorological year used in the modelling).

A concentration of 630 Bq/kg for radionuclides in the U-238 decay chain, contained within deposited dust was used in the terrestrial assessment. This concentration was not expected to change significantly over time.

Recent preliminary surface water modelling results (Water Solutions 2020) provided the predicted concentrations of uranium, Ra-226 and Po-210 at a number of locations along the surface water pathways of the RPA for the years 1, 20, 270 and 10,000 post-closure. The likely concentrations of U-238, U-234, Th-230, Pb-210 and U-235, necessary for the dose assessment, were extrapolated from these predictions using equilibrium assumptions and the ratio of radionuclides reported in Murray (1992).

The potential concentrations of radionuclides above natural background levels were then calculated for Mudginberri, Coonjimba, Georgetown and Gulungal billabongs for the timeframes 1, 20, 270 and 10,000 years.

The outcomes from the atmospheric dispersion and surface water models were used as inputs into the radiation dose assessment. The assessment considered potential radiological impacts to members of the public, as well as terrestrial and aquatic biota.

Members of the public

The dose assessment for members of the public post-closure considered the following radiation exposure pathways:

- inhalation of long-lived alpha activity (e.g. radioactive dust)
- inhalation of radon decay products
- ingestion of radioactive material in (or with) food or water
- external irradiation from gamma radiation.

Further information on post-closure landuse required for dose assessment is provided in Section 8.

Terrestrial and aquatic biota

The impact to specific terrestrial and aquatic species is based on changes in radionuclide concentrations of the media within which the species resides. For example; the media for fish is water. Therefore, determining the incremental changes in water radionuclide concentrations post closure is the basis for determining impact to fish. The method for determining the change in media concentration is via modelled dust deposition results and surface water solute transfer. The impacts to biota were then assessed using these incremental concentration changes and the ERICA assessment software tool (http://www.erica-tool.com/).

Post-closure guidance values have been developed to provide radiological protection to terrestrial and freshwater aquatic species (Doering & Bollhöfer 2016, Doering *et al*. 2019). The guidance values were compared to the predicted changes in media concentrations for above background concentrations of Ra-226. Guidance values for Ra-226 concentrations in water and soils were not exceeded.

As the guidance values were not exceeded, a limited number of more targeted ERICA assessments were conducted:

- terrestrial species on the final landform at Closure
- freshwater aquatic species in the Gulungul Billabong at years 1, 20, 270 and 10,000
- freshwater aquatic species in the Coonjimba Billabong at year 270
- freshwater aquatic species in the Mudginberri Billabong at year 270

A number of representative organisms were considered in the ERICA assessment:

- freshwater fish (including benthic and pelagic species)
- molluscs (including bivalve and gastropod species)
- freshwater reptile
- freshwater vascular plants
- amphibian
- arthropod
- bird
- grasses & herbs
- mammal - large
- mammal - small-burrowing
- reptile

- tree

The outputs of the ERICA assessment, as dose rates to representative organisms, will be reviewed against closure criteria dose rates of 100 uGy / hr for the most exposed terrestrial species and 400 uGy / hr for the most exposed aquatic species.

Results of the radiological impact assessment will be preliminary as the water quality data used in this report are being updated in the final Surface Water Model (Section 5.5.2.11). The radiological impact assessment will be updated after water quality data is finalised. The complete dose assessment results will be included in the 2021 MCP.

5.5.3.2 Bushtucker sampling

An Independent Surface Water Working Group (ISWWG) conducted a review of the surface water management and monitoring associated with Ranger Mine in 2013. The ISWWG (Hart & Taylor, 2013a) recommended the re-introduction of the bush tucker monitoring program:

> *Recommendation 6: A routine 'metals (including radionuclides) in bush tucker' monitoring program be re-introduced, with ERA and GAC to provide details on the scope and objectives for such a program, and SSD to review existing 'metals in bush tucker' data base and provide advice on program design.*

Hart and Taylor (2013b) detailed the information and rationale that led to these recommendations.

The above recommendation was aimed at addressing concerns of the Mirarr Traditional Owners regarding the contaminant levels in bush tucker from Mudginberri Billabong by reintroducing a monitoring program for heavy metals and radionuclides in fishes and other freshwater biota.

The targeted species for the sampling program have been discussed in the Bush Food Diet section in the document *Post Closure Land Use* (Paulka 2016).

This study is undertaken in two phases. The first phase of this study is complete, and focussed on terrestrial fruit and vegetables collected from the Trial landform and other areas on the RPA. The second phase of this study will look at collecting and analysing a variety of terrestrial and aquatic fauna, to be undertaken in the second half of 2020.

Scope and approach

The aim of this project is to determine the bioaccumulation of heavy metals and radionuclides in traditional Bininj food and to interpret and communicate the results.

The first phase of the project assessed selected flora species. Flora, except for yams, have been sourced from the Trial Landform (TLF). Yams have been sourced from elsewhere on the RPA as they are not present on the TLF.

Fauna sampling and assessment will be completed as phase two. The fauna species selected will include a variety of introduced and native species found on the Ranger Project Area and surrounding Kakadu NP. Fish species will be sourced from Mudginberri and Georgetown billabongs. The locations for fauna sampling are shown in Figure 5-105.

All approvals will be sought prior to the commencement of works, including Charles Darwin University animal ethics approval, Parks Australia Approval, Fisheries Approval. The work will involve Traditional Owners where possible.

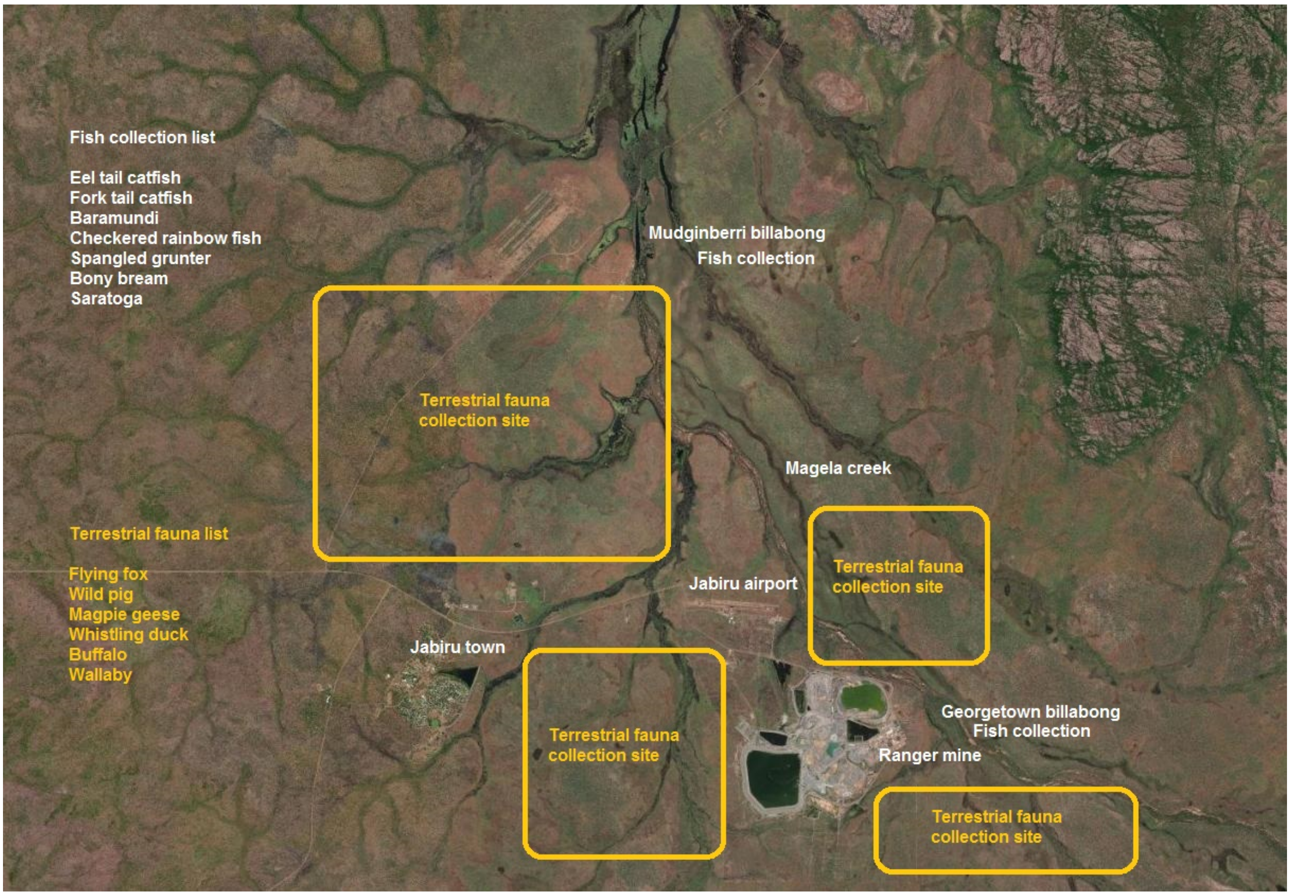


Figure 5-105 Fauna (bushfood) sampling locations within Kakadu National Park and the RPA.

5.5.3.3 Pit 1 radiological monitoring

ERA is currently finalising the scope of works to undertake radiological monitoring on the completed Pit 1 landform.

Scope and approach

A radiation survey and sampling program is to be undertaken and will consist of four components:

- Surface gamma survey
- Radon-222 exhalation flux density measurement

- Radium-226 waste rock substrate sampling
- Radon-222 in air measurement (passive)

The survey and sampling will be based on a systematic random sampling approach as shown in Figure 5-106 below (IAEA 2019). The systematic random sampling approach will allow radiological monitoring to be deployed without interference with other Pit 1 works (contouring, irrigation, revegetation, etc).

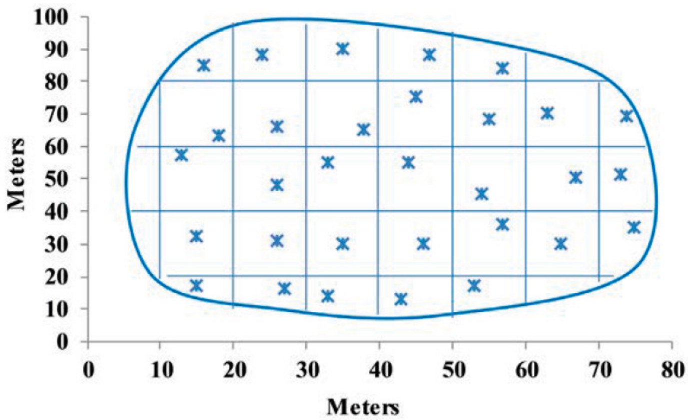


Figure 5-106 Systematic random sampling approach (IAEA 2019)

Gamma Survey

A gamma survey will be performed by competent trained personnel using a gamma detector in a regular grid pattern over the Pit 1 area. Absorbed gamma dose rates are to be measured at a height of 1m above the ground level and integrated over a 60 second time interval.

Radon-222 exhalation flux measurement

Brass canisters containing activated charcoal will be used to collect the exhaled Radon-222 from the surface waste rock. The canisters will be standard brass cylinders with an internal diameter of 0.007 m, depth 0.058 m and a wall thickness of 0.004 m, or other appropriate design proved suited for the purpose of the sampling program. The canisters will be prepared by heating (over 110 °C) over 48 hours (or other suitable method) to eliminate adsorbed substances prior to the measurement.

The mouth (face) of the canister will be put against the ground surface and sealed when necessary. Putty seal will be used to seal canisters on Pit 1 as it will be a waste rock surface. Areas of water inundation will be avoided. The canisters will be left for 3 days (72 hours) to secure the total adsorption of Radon-222 and the shorter-lived progeny radionuclides to the charcoal contained.

A number of sealed 'blank' canisters will be deployed in the field for background reference data and sent to the lab with the other samples for analysis. The decay of Radon-222 progeny will be measured with a NaI(TI) gamma detector calibrated for the respective cup geometry. The

ERA

Radon-222 exhalation flux density over the period of exposure of the charcoal canister on the landform will be estimated using published methodologies with Spehr and Johnston (1983) and Bollhöfer *et al*. (2005) as examples.

To assess seasonal variability, ERA will aim to undertake Radon-222 exhalation flux measurements at the end of dry-season in 2020 and at the end of wet season in 2021.

Radium-226 waste rock substrate sampling

Surface substrate samples of 10cm depth will be collected from directly underneath all the locations where Radon-222 exhalation flux measurements occur. Sufficient volume of substrate to enable analysis is to be collected from each location.

The collected substrate samples are to be homogenised in preparation for radionuclide analysis by gamma spectrometry. Samples will be sent for analysis with an additional storage period of a minimum 24 days after pressing to allow for the ingrowth Radon-222 progeny radionuclides. Radon-222 is used as a proxy measurement of Radon-226 in the sample.

Radon-222 in air measurement

Passive radon monitors (PRM) will be used for the measurement of radon in air. The monitors will be placed 1 m to 2 m above the ground level for 3 months and then collected to be sent to certified laboratory for Radon-222 analysis. Sampling locations will follow the same grid pattern as Radon-222 exhalation and Radium-226 sampling. The PRM will then be sent to an accredited laboratory for radon gas decay counts.

5.5.4 Ecosystem rehabilitation

This section provides summaries of the completed studies relating to the theme Ecosystem Rehabilitation as well as selected completed and ongoing KKN related studies. Some studies inform multiple KKNs and have only been included once to avoid repetition.

KKN title	Project title	Status	Section
ESR1. Determining the requirements and characteristics of terrestrial vegetation in natural ecosystems adjacent to the minesite, including Kakadu National Park.	Conceptual Model of Final Revegetation Reference Ecosystem/s	In Progress	5.5.4.1
ESR2. Determining the requirements and characteristics of a terrestrial faunal community similar to natural ecosystems adjacent to the minesite, including Kakadu National Park	Terrestrial fauna objectives & recolonisation strategy	In Progress	□
	Trial Habitat Creation on Trial Landform	In Progress	0

KKN title	Project title	Status	Section
ESR3. Understanding how to establish native terrestrial vegetation, including understory species.	Understorey nursery and TLF trials	In Progress	5.5.4.4
	Pit 1 Revegetation Studies	In Progress	9.3.1.3
ESR5. Develop a restoration trajectory for Ranger Mine	Evaluation of Key Attributes of Nutrient Cycling in Revegetated Waste Rock Landform of Ranger Uranium Mine	Complete	Appendix 5.1
ESR7. Understanding the effect of waste rock properties on ecosystem establishment and sustainability	Study of Root depth on TLF	Complete	Appendix 5.1
	Soil formation (PSD monitoring) on TLF at Year 10	Planned	0

5.5.4.1 Conceptual model of final revegetation reference ecosystem/s

This project relates to multiple KKNs:

- ESR1C. What values should be prescribed to each indicator of similarity to demonstrate revegetation success?
- ESR5B. What are possible/agreed restoration trajectories (flora and fauna) across the Ranger Minesite; and which would ensure they will move to a sustainable ecosystem similar to those adjacent to the minesite, including Kakadu National Park?
- ESR 8A. What is the most appropriate fire management regime to ensure a fire resilient ecosystem on the rehabilitated site?

This project aims to review and compare industry best practice, ERA and the SSB approaches to reference site selection and flora and fauna closure criteria development. From this, ERA will develop the best approach for application at Ranger, including suitable reference ecosystems, justified closure criteria and complementary revegetation methods.

Scope and approach

The project is focussed on defining conceptual reference ecosystems and closure criteria for the post-mining Ranger landscape. Dr Libby Mattiske from Mattiske Consulting has been engaged to undertake this work, which requires the following:

- collate and analyse available baseline and rehabilitation datasets relevant to the Ranger Mine to develop a series of site specific reference ecosystems

- complete an assessment to identify the most suitable conceptual reference ecosystem for each domain, given specific constraints of the post-mining domains
- undertake an extensive benchmarking study to review and compare industry best practice for setting practical closure criteria
- develop closure criteria for each revegetation domain, including flora, fauna and other attributes that cover community composition, structure, and function (including resilience and sustainability

Progress

During 2019/2020 ERA and Dr Mattiske achieved a number of key steps, in consultation with key stakeholders, most of which is integrated within Appendix 5.1:

- completed the collation and analysis of available baseline data and proposed a series of potential conceptual reference ecosystems (Appendix 5.1)
- developed a package of technical information to inform future revegetation domain definition
- agreed on descriptive closure criteria with stakeholders (Section 8)

In 2020/2021, ERA shall continue working towards quantitative closure criteria through the following steps:

- review all available rehabilitation monitoring data from ERA including trial landform data, previous revegetation trials, and early results from Stage 13 and Pit 1 revegetation activities
- access relevant rehabilitation data from other sites, such as the *Eucalyptus tetrodonta* dominated revegetation at Gove and Weipa bauxite mines (over 40 years of knowledge)
- utilise the State-and-Transition model that has recently been developed (Richards et al. 2020 - in draft) to refine the trajectories for key parameters of the revegetation, to identify milestones and thresholds to inform the ERA Adaptive Management Plan
- review other trajectory study options as recently developed by Steedman et al. (2019) utilising species richness and density datasets to evaluate progress on rehabilitation areas
- propose quantitative closure criteria for the target 'close-out' timeframe expressed relative to the appropriate conceptual reference ecosystem
- undertake a statistical review and benchmarking exercise on how quantitative closure criteria should be monitored and assessed at Ranger Mine

5.5.4.2 Terrestrial fauna objectives & recolonisation strategy

This project relates to multiple KKNs:

- ESR2A. What faunal community structure (composition, relative abundance, functional groups) is present in natural ecosystems adjacent to the minesite, and what factors influence variation in these community parameters?
- ESR2C. What is the risk of feral animals (e.g. cats and dogs) to faunal colonisation and long-term sustainability?
- ESR5B. What are possible/agreed restoration trajectories (flora and fauna) across the Ranger Minesite; and which would ensure they will move to a sustainable ecosystem similar to those adjacent to the minesite, including Kakadu National Park?

Scope and approach

This project will identify the parameters required to identify the attributes of the terrestrial ecosystem that will enable recolonisation of the final landform with a diverse fauna community. This diversity includes the presence of invertebrate and vertebrate fauna (including consideration of richness, diversity, composition, occupancy and functional diversity), taxa of specific interest for their environmental and cultural significance and the management of exotic fauna.

This work will support the development and finalisation of fauna-related closure criteria, both direct and indirect measures. SLR has been engaged by ERA to undertake this project.

This project comprises two stages:

1. in consultation with key stakeholders, draft a report on the Ranger fauna closure criteria.
2. in consultation with key stakeholders, develop a recolonisation plan and monitoring program to facilitate fauna return.

Note: The project "Trial habitat creation on the TLF" is related to, and will inform, this larger project.

Progress

In 2020, SLR developed an updated suite of terrestrial fauna closure criteria based on scientific publications and informal consultation with key stakeholders. The closure criteria comprised a combination of metrics that assess attributes of the ecosystem to facilitate the recolonisation of a diverse fauna community, the presence of fauna, taxa of specific interest for their environmental and cultural significance and the management of exotic fauna. A draft report is currently under review and will be made available to stakeholders for consultation once it is finalised. A set of proposed draft closure criteria have been provided in Section 8.

5.5.4.3 Trial artificial habitat creation on trial landform

This project relates to multiple KKNs:

- ESR2B. What habitat, including enhancements, should be provided on the rehabilitated site to ensure or expedite the colonisation of fauna, including threatened species?
- ESR5. Develop a restoration trajectory for Ranger Mine.

This project will identify the types of artificial habitat options available, and test their effectiveness in facilitating the utilisation by native species on the trial landform, and also suitable bushland sites.

Scope and approach

The presence of suitable habitat is an essential precursor to fauna recolonization. The development of mature vegetation communities correlates with increased faunal diversity. The presence of vegetation communities has frequently been used as an indicator of fauna recolonisation in mine closure (see reviews by Cross *et al*., 2019, Cristescu *et al*., 2012). Vegetation features that have been considered as indicators of the development of suitable fauna habitat for a diverse range of fauna include:

- tree hollows
- edible fruit-bearing trees and shrubs
- leaf litter and woody debris

Tree hollows provide important habitat for amphibian, bird, mammal and reptile species, including many species which are hollow-dependent (Taylor *et al.* 2003, Goldingay 2009, Goldingay 2011, Lindenmayer *et al.* 2014). Individuals of hollow-using and dependent species generally use multiple hollows selected on a number of characteristics, which potentially include tree size, height of hollow, entrance size, hollow form and position, hollow aspect and/or hollow depth (Goldingay 2009, 2011). Hollows (particularly uncommon large hollows) occur most frequently in large, old trees and Goldingay (2011) estimated that most trees used as mammal dens (including those in the NT) were >100 years of age.

Leaf litter and coarse woody debris (generally fallen timber >10 cm diameter) provide habitat for fauna species, including some specialists, in tropical savanna ecosystems such as at Ranger Mine. However, ground cover is highly variable depending on fire regimes and detritivore activity. One opportunity for increasing the diversity of species able to colonise the waste rock final landform would be the establishment of fresh litter islands.

This project will trial the use of artificial nest boxes on the trial landform to expedite the colonisation of the landform by fauna. ERA will deploy a variety of nesting boxes on the TLF, and also suitable bushland sites, to determine their effectiveness. Nesting boxes for arboreal mammals, bats and birds have been procured and safety and logistic arrangements for installation are currently underway. Other habitat methods relating to the provision of artificial ground cover (e.g. pipes, boards, rock piles) will also be trialled. Further details on litter islands has been provided in Section 3.3.3.3 of Appendix 5.1.

Evaluation and reporting outcomes

- regular updates to stakeholders as the trials progress.
- outcomes will be included in the Ranger Mine Closure Plan.

5.5.4.4 Understorey nursery and TLF trials

This project relates to multiple KKNs:

- ESR3A. How do we successfully establish terrestrial vegetation, including understory (e.g. seed supply, seed treatment and timing of planting)?
- ESR7D. Are there any other properties of the rehabilitated site that could be attributed to any observed impairment of ecosystem establishment and sustainability, including vegetation and key functional groups of soil fauna?

Scope and approach

ERA need to demonstrate the ability to establish the full range (or an appropriate complement) of native vegetation species from the reference ecosystem. While this has been shown in initial trials for key overstorey species, there is far less available evidence for the successful establishment of a diverse suite of understorey species. This study will test a large suite of understorey species under trees in the TLF, site 1A and also on bare waste rock landform.

This project includes a number of trials including:

- 2020-21 TLF ‘secondary’ introductions trial
- 2020-21 TLF ‘secondary’ introductions: Understorey direct seeding trial
- ongoing monitoring of 2018 understorey trial
- nursery trials

These trials are discussed in Appendix 5.1

5.5.4.5 Soil formation (PSD monitoring) on the TLF at Year 10

This project relates to KKN ESR7C: Will ecological processes required for vegetation sustainability (e.g. soil formation) occur on the rehabilitated landform and if not, what are the mitigation responses?

Scope and approach

The TLF was constructed in 2009 from waste rock materials (100%) in TLF, sections 1A and 1B. The waste rock material was from Pit 3 run-of-mine which has a low content of fines (particle size < 2 mm). The fines content in TLF 1A ranges from 39 % to 27 % with an average of about 33 %. Plant available water can only be held in fines of the waste rock substrate. Low fines content results in a low plant available water capacity of the landform. The WAVES model demonstrated that a waste rock landform of 4-6 m will be able to support a woodland vegetation that is similar to that at the Georgetown creek reference sites 30 and 21 (Lu *et al* 2019). It is anticipated that as the waste rock weathers through physical/chemical and biological weathering processes the fines content in the substrate shall increase, thereby increasing the plant available water in the landform.

This project involves the collection and analysis of PSD in the top 10 cm of section 1A and 1B of the TLF. Results will be compared to previous results to demonstrate the degree of increase in fines content in the substrate after 10 years since construction of the landform.

Samples have been collected and analysis is underway.

5.5.5 Cross theme

KKN title	Project title	Status	Section
CT1. Assessing the cumulative risks to the success of rehabilitation on-site and to the protection of the off-site environment	Climate change and mine closure	Completed	5.5.5.1

5.5.5.1 Climate change and mine closure

A staged approach is recommended in which a first pass risk screening is first undertaken to understand how direct and indirect pressures from climate change may affect all aspects of the closure plan. The first pass assessment includes undertaking a review of any previous climate change considerations in the Kakadu NP and in relation to the mine (including by the SSB).

The approach will enable the climate change for the closure of Ranger Mine to be framed appropriately, something which is critical for the longer-term success of adaptation planning. The risk screening enables a more detailed, targeted approach towards understanding and managing climate risk, to be scoped. This will also identify any further studies or analyses that may be required and assist in the minimisation of unnecessary expenditure whilst ensuring that any studies and analysis will fill gaps and add value to mine closure adaptation.

The approach aligns with the ISO31000 Risk Standard and with leading practice in climate adaptation risk assessment framing, including guidance from ICMM on considering climate change for closure.

Scope and approach

This project was initiated to identify how climate change is likely to affect the plan for the closure of Ranger Mine and to determine any additional investigations or actions that are required to help address any challenges. The project aims to:

- understand how direct and indirect pressures from climate change may affect all aspects of the closure plan and the risks it may create
- understand climate change predictions of rainfall, temperature, cyclones, sea level change etc. for the region surrounding Ranger Mine during the decommissioning at post closure periods
- understand how the climate change predictions are likely to affect rehabilitation of the Ranger Mine and the surrounding environment and ecosystem
- screen risks to identify high risk issues
- identify if additional studies or processes are needed to underpin further risk assessment or management
- identify scenarios for modelling of high risk issues

Results and conclusions

Climate change descriptions for Kakadu NP have been completed and a stakeholder inception meeting for context setting, information availability, method and project planninga was undertaken in 2019.

A stakeholder workshop was held in ERA's Darwin office conference room on March 2020 to undertake a first pass assessment of climate change risk to the closure of the mine. The assessment was undertaken by subject matter experts from within and outside of ERA. A further on-line workshop was conducted with bushfire experts to gather additional expert input into this critical aspect.

The process included delivery of a briefing on climate projections for the target area, based on available information obtained from reliable resources including the Commonwealth Scientific and Industrial Research Organisation (CSIRO), the Bureau of Meteorology (BoM) and the National Climate Change Adaptation Research Facility (NCCARF). Additional information was drawn from published peer reviewed literature.

An overview of the risk assessment process was presented and included discussion on the likelihood and consequence tables to underpin the risk analysis and ensure that all participants were comfortable with the approach. Stakeholders reaffirmed the outcomes for the project from the Inception Workshop, including the areas that should covered by the assessment and the projected timeframes that should be covered in the assessment. It was agreed that Jabiru and the airport were not to be included in the assessment and that the main timeframes to be considered were 2030 (initial post-closure planting and maintenance period), 2050 (planned post-closure monitoring end date), and 2100 (best available long-term projections). A mid-range climate change scenario of RCP4.5 was selected and a business as usual climate change scenario of RCP8.5. Using these two possible futures would help to determine when any major risks were likely to occur. There is little difference between the climate change projections of the two scenarios until after 2050.

In assessing risk, the current management plans and activities relating to the mine closure were discussed. Their role in addressing relevant climate change risks was assessed to enable any residual risk to be identified.

Discussion took place regarding the assessment of climate related risks for longer time periods associated with mine closure including the 10,000 year time period to be consistent with regulatory conditions. There are few climate change data available for those periods and the uncertainties associated with them is extreme. Accordingly, it was agreed that there was little merit in including these risks in the risk assessment activity.

The approach was then used to work through risks associated with: heat, sea-level rise and salinity, rainfall and drought, cyclones, and bushfire.

Thirty-seven potential risks were discussed and assessed. Risks were classified into four key areas

(1) onsite activities (management and monitoring)

(2) vegetation

(3) onsite and receiving water quantity, quality and ecology

(4) erosion and sediment.

In general, the relatively short period (compared to climate change timeframes) of active onsite management and monitoring, expected before the site stabilises and meets close-out conditions, meant that the risk profile for the mine closure was fairly low.

A full report of the risks assessed and recommendations has been finalised and shared with stakeholders.

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APPENDIX 5.1: REVEGETATION KNOWLEDGE BASE

REVEGETATION KNOWLEDGE BASE

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1 REVEGETATION OBJECTIVES, GUIDELINES AND STANDARDS

1.1 Objectives

The revegetation objective for ERA Ranger Mine is stated in the *Environmental Requirements of the Commonwealth of Australia for the Operation of Ranger Uranium Mine* (1999), which sets out the overarching environmental management at Ranger (referred to as the 'Environmental Requirements' or 'ERs'). Of direct relevance to this revegetation strategy are the following clauses:

> *2.1 … the company must rehabilitate the Ranger Project Area to establish an environment similar to the adjacent areas of Kakadu National Park, such that, in the opinion of the Minister with the advice of the Supervising Scientist, the rehabilitated area could be incorporated into the Kakadu National Park.*
>
> *2.2(a) Revegetation of the disturbed sites of the Ranger Project Area using local native plant species similar in density and abundance to those existing in adjacent areas of Kakadu National Park, to form an ecosystem the long term viability of which would not require a maintenance regime significantly different from that appropriate to adjacent areas of the park.*

Relatively high-level rehabilitation objectives, including the required post-mining land use, must be further developed and translated, through consideration of physical, chemical and other constraints of the altered conditions, into clear qualitative and/or quantitative targets (criteria) (e.g. Young *et al*. 2019c). This is necessary for rehabilitation planning and execution, subsequent monitoring and management of the developing ecosystem, and final assessment and sign-off (relinquishment) of the mature rehabilitated ecosystem. The following diagram represents the approach taken at ERA and indicates the process of refinement of objectives considering post-mining conditions, conceptual reference ecosystems, closure criteria, monitoring and management.

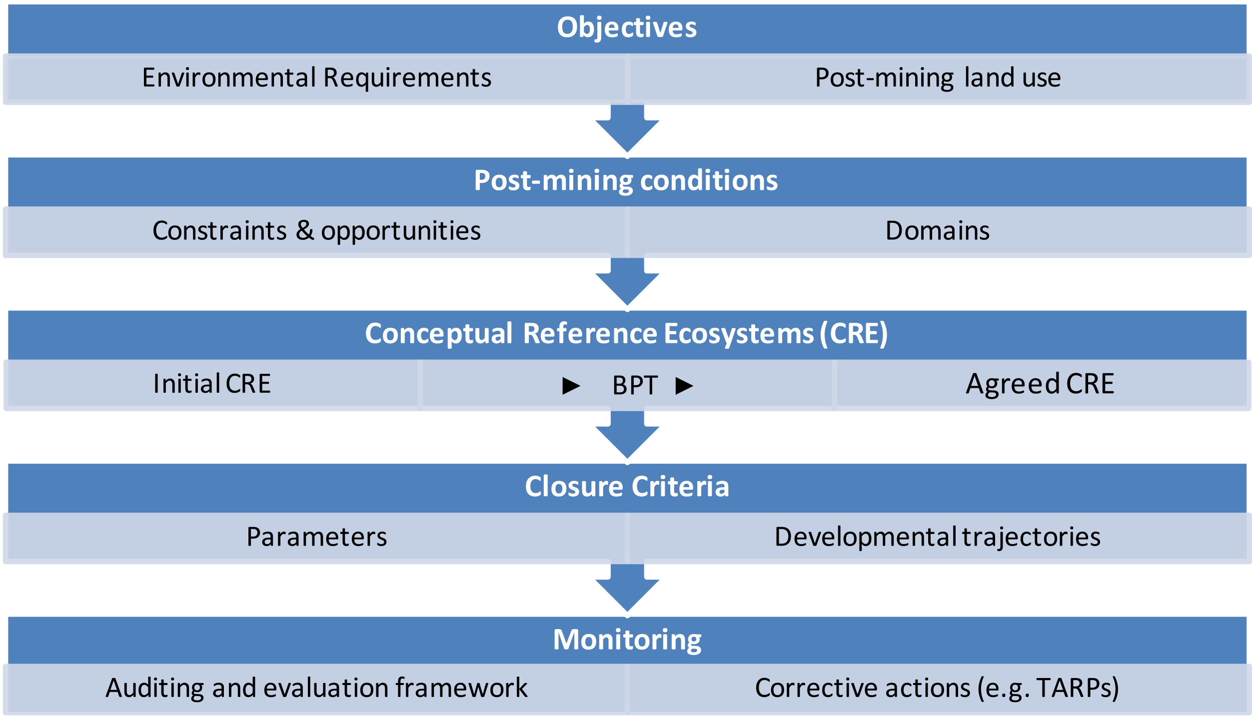


Figure 1-1: Process flow diagram (consistent with Young *et al*. 2019c).

Such an approach reflects the ongoing maturation of industry standards, which now recognise the need for all operators and stakeholders alike to have a clear understanding of the legal commitments and target end land uses, including specifics on the anticipated closure and rehabilitation challenges and opportunities with an emphasis and commitment to the process of continual review and improvement.

Of particular importance is the need to clearly differentiate between the ideal of 'restoration of the ecosystem to its pre-existing state' and the practical and feasible, given the often significantly altered post-mining site conditions. Different mines have very variable site conditions that need to be rehabilitated after mining and associated activities. For example, revegetation practices at large, progressive shallow mining operations such as bauxite mining or sand mining (where it may possible to replace topsoil and overburden directly during operations) are not feasible at open pit operations where minimising the disturbance footprint has been a priority, such as Ranger.

The Australian federal government's '*Leading Practice Sustainable Development Program for the Mining Industry'* series (Australian Government 2016b) uses the following distinction:

> ***Rehabilitation*** - *The return of disturbed land to a stable, productive and self-sustaining condition, after taking into account beneficial uses of the site and surrounding land. Reinstatement of degrees of ecosystem structure and function where restoration is not the aspiration.*
>
> ***Restoration*** - *Re-establishment of ecosystem structure and function to an image of its prior near-natural state or replication to a desired reference ecosystem.*

The National standards for the practice of ecological restoration in Australia (SRG SERA 2017) uses a similar, but slight different definition, namely:

> ***Rehabilitation*** *- reinstating some form of ecosystem functionality without seeking to also recover a substantial proportion of the native biota found in an appropriate native reference ecosystem. (Note: Such rehabilitation is especially encouraged and valued where it: (i) improves ecological condition or function and (ii) is the highest standard that can be applied.)*
>
> ***Ecological restoration*** *- the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. (Note: Single species restoration can be considered complementary and an important component of ecological restoration.)*

This differentiation between restoration and rehabilitation is important as it directly relates to the setting of realistic rehabilitation objectives and targets as well as the development and execution of appropriate, successful rehabilitation programs. Due to the significantly altered post-mining landscape at Ranger Mine, ERA uses the term rehabilitation to describe the overall closure program, including revegetation and subsequent ecosystem development.

To identify and accommodate this situation, an understanding of the ecosystem functions on both the native environments and their relevance to the post mining conditions is required, in particular a clear and definitive understanding of the constraints imposed on rehabilitation processes by the post-mining engineered landforms and site conditions.

In recent times the approach of designing ecosystems that have values in common with the native ecosystems in the local or regional context has become widespread (Mattiske & Meek 2020). At Ranger this is particularly evident in the language of the Environmental Requirements and also industry, community and stakeholder expectations. Whilst similarity and alignment of rehabilitation areas with the local native ecosystems is a reasonable goal, the influence of constraints and threats (e.g. seasonal establishment and growth conditions, site physical and chemical constraints, water availability, fires, weeds and exotic fauna) must be factored in. This may restrict the ability of a site to achieve a high degree of similarly to the reference ecosystem, at least within the timeframes normally available for revegetation establishment, management and site relinquishment (e.g. decades compared to centuries or more for achieving a mature, reference ecosystem). This does not necessarily mean that current rehabilitation and relinquishment timeframes are inappropriate (in fact, the timely return of land to post-mining landholders is often another rehabilitation driver), but that measures of success must take this into consideration and effort must be put into providing a sufficient level of confidence for the ongoing development of the relinquished rehabilitation towards the final, mature end state over time.

These concepts and how they are applied at ERA Ranger Mine are covered in more detail in Section 2 below.

1.2 Guidelines and standards

There are numerous Australian and international sources of guidance on the process and management of rehabilitation and closure in the mining industry including:

- Mine closure – leading practice sustainable development program for the mining industry (Australian Government 2016a)
- Mine rehabilitation – leading practice sustainable development program for the mining industry (Australian Government 2016b)
- Integrated mine closure – good practice guide. Second edition. (ICMM 2018)
- National standards for the practice of ecological restoration in Australia. Second Edition. (SRG SERA 2017)
- Guidance for the assessment of environmental factors – rehabilitation of terrestrial ecosystems. No. 6. (WA EPA 2006)
- Statutory Guidelines for Mine Closure Plans (WA Department of Mines, Industry Regulation and Safety 2020)
- Completion criteria framework: an overview. (Young *et al.* 2019a)
- Completion criteria framework: endorsed by the Department of Mines, Industry Regulation and Safety. (Young *et al.* 2019b), and
- Project report: a framework for developing mine-site completion criteria in Western Australia. (Young *et al*. 2019c).

The current standards associated with baseline studies and rehabilitation studies has progressed significantly in the last few decades in line with increasing community expectations and also increasing industry standards associated with ecological rehabilitation and restoration programs (SRG SERA 2017; ICMM 2018; WA EPA 2016a, 2016b; Australian Government 2016b; Kragt *et* al. 2019).

In 2018, the SSB drafted an "*Ecosystem Restoration – Rehabilitation Standard for the Ranger uranium mine*" that aims to describe the requirements for restoring the terrestrial ecosystem of the Ranger Project Area (including riparian areas) in the Alligator Rivers Region of the Northern Territory (Supervising Scientist, 2018). This standard is considered by ERA, along with the overarching Environmental Requirements and corporate standards, to determine the desired outcomes for environmental protection at Ranger Mine.

2 REFERENCE ECOSYSTEMS AND CLOSURE CRITERIA

As prescribed in the ERs, ERA must establish an environment using local native plant species similar in density and abundance to those existing in adjacent areas of Kakadu National Park.

At Ranger Uranium Mine, the waste rock final landform is dramatically different to pre-mining conditions and, although ERA has shown that this material can support development of a native woodland ecosystem (on the Trial Landform and other trials – see Section 3), there will likely be a degree of difference in these revegetated ecosystems to those that were there previously. The specific physical and chemical constraints (if any) of the rehabilitated landform must be considered (in the form of 'revegetation domains') and appropriate reference sites chosen representing native ecosystems likely to be suited to the post-mining conditions (SRG SERA 2017).

In the absence of a natural reference ecosystem with a similar topography and substrate as the final landform, a nearby natural reference ecosystem can be adopted but "adjusted to accommodate changed or predicted environmental conditions" (SRG SERA 2017). The reference ecosystem in the case of Ranger Mine will be a conceptual model synthesised from appropriate reference sites chosen considering, and/or adjusted for, the permanent and irreversible changes to the site based on research, trials, experience, benchmarking, and historical and predictive records.

Closure criteria are the qualitative or quantitative standards of performance used to measure the achievement of the rehabilitation closure objectives for the closure of the site and needed for the relinquishment of the mining lease (WA EPA 2006). They are usually expressed relative to a reference ecosystem (Young *et al.* 2019b) and a key principle of completion criteria development is that the change in the nature of the site as a result of mining is acknowledged (Young *et al.* 2019c).

ERA has developed a set of descriptive closure criteria, agreed with key stakeholders (SSB and NLC) in 2020. This is seen as a positive and important step on the journey towards developing quantitative criteria against the full suite of conceptual reference ecosystems suited to the revegetation domains of the rehabilitated mine site.

Figure 2 indicates the relationship between reference ecosystems (on the longer time frame), closure criteria (eg. after 25 years) and the revegetation domains associated with the post-mining site conditions in the rehabilitation areas.

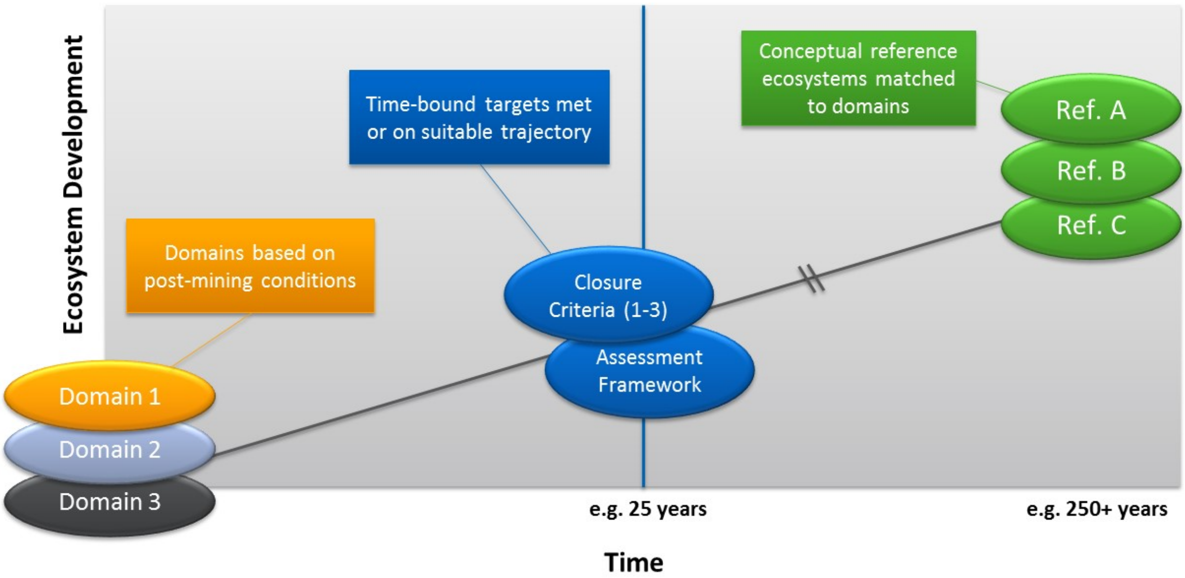


Figure 2-1: Relationship between domains, trajectories and (conceptual) reference ecosystems.

2.1 Reference site selection at Ranger Mine

A key element of the Ranger Revegetation Strategy (Reddell & Meek 2004; Section 1) has been to identify and describe vegetation types that are ecologically, culturally and technically realistic target endpoints, for different facets of the final landform, based on the likely physical and chemical environments that will be created. The identification of suitable reference vegetation types has mainly been based on ERA surveys in the surrounding natural landscapes that are potential geomorphic analogues of those formed on the final landform (based on the reasonable assumption that many of the environmental determinants of vegetation distribution will be similar in these settings). The intention is to revegetate the majority of the landform post mining with open eucalypt-dominated woodlands similar to the native vegetation typical of the surrounding areas near Ranger and within Kakadu National Park.

As work on this has progressed, including collaboration with key stakeholders, a clearer pathway towards development of an agreed conceptual reference ecosystem model for Ranger Mine revegetation has appeared, as outlined below:

- Ensure a shared understanding of clear and specific objectives (Section 1).
- Understand the ideal environmental conditions for the target post-mine land use and, as far as practicable, consider these in the design and execution of the rehabilitated landform.
- Understand any constraints (and opportunities) to revegetation establishment imposed by the post-mining conditions (Section 2.1.1).

- Identify and characterise a series of natural reference sites including common vegetation found in nearby areas, and/or vegetation likely to be suited to the different conditions in each revegetation domain (Section 2.1.2).
- Review and modify the natural reference sites based on research, trials, experience, benchmarking, and historical and predictive records (Section 2.2).

The following sections cover the final three stages of this approach.

2.1.1 Influence of post-mining conditions

A primary objective for rehabilitation of the Ranger Project Area is to return a native ecosystem similar to those in nearby Kakadu National Park. To ensure that the specific goals underpinning this objective are realistic and achievable, it is important to take into consideration all elements that may constrain or favour the various options (Young *et al.* 2019a-c; McCullough 2016).

At Ranger Mine, there are a range of physical constraints that may affect our ability to achieve the objective at a species level, community level, structurally, and also with regard to spatial distributions across the landscape (the final landform is an engineered landform and the locations or extents of the various constraints will not necessarily occur in a 'natural' distribution).

A preliminary approach to assessing the potential of post-mining landscapes is to undertake a landscape capability assessment (Young *et al*., 2019c). In 2020, ERA commissioned 2rog Consulting to assess and describe the land capability of the proposed final landform (2rog Consulting, 2020).

Also in 2020, ERA produced a technical brief of potential physical and chemical constraints that may influence vegetation suitability (as evidenced by their ability to establish and develop into a sustainable ecosystem), particularly on the waste rock final landform. This brief was reviewed with key stakeholders (May 2020 Ecosystem Restoration Working Group, comprising ERA, SSB, NLC and select ARRTC representatives) and it was agreed that most constraints warranted further consideration as ERA continues to refine the agreed reference ecosystems and related criteria. These constraints are discussed below, including:

- material type and relationships to plant water availability, rooting depth and so on
- surface hydrology and subsurface hydrogeology, including seasonal variations
- substrate chemical status, including nutrients and contaminants of potential concern
- slopes and aspect

The extent and influence of these constraints was used in the following sections to develop a series of revegetation domains across the post-mining land form and then on the basis of these match each domain to a suitable reference ecosystem considering relevant environmental conditions (Section 2.1.2).

2.1.1.1 Land capability assessment

In 2020, ERA commissioned 2rog Consulting to assess and describe the land capability of the proposed final landform (2rog Consulting, 2020). The project was to consider the land capability of the final landform and place this landform capability within context of a broader regional area. Spatial data from existing sources including site-based mapping and modelling and regionally available data were to be used in conjunction with the NT guidelines for land capability assessment. A summary of the assessment outcome is provided below.

Land capability assessment in the Northern Territory is included within the land clearing and native vegetation management guidelines (DoENR, 2019). Land capability and land suitability assessments are used to determine if a soil and land resource is appropriate for the intended post-clearing land use. Land capability assessments evaluate the key soil and land resource parameters recorded in a land type map against a defined set of criteria to determine an overall land capability class. There are four land capability classes, Class 1 is the most versatile resource with Class 4 the most constrained.

Resulting from the 2rog assessment, almost 90% of the final landform (including some of the natural surrounds) was found to be classed as 'marginal' (land with severe constraints and requires considerable management practices) or 'not recommended' (land with extreme constraints too severe to develop. Can only be overcome with major management and/or engineered solutions).

Table 2-1 Classes resulting from the land capability assessment (2rog 2020).

Capability Class	Regional		RPA		Final landform & surrounds	
	ha	%	ha	%	ha	%
1 - High		0%		0%	53	3%
2- Moderate	18,444	2%	453	6%	184	11%
3 - Marginal	136,277	18%	2,260	29%	369	22%
4 - Not recommended	597,406	79%	5,196	66%	1,112	67%
TOTAL	752,127		7,908		1,665	

2.1.1.2 Material Type

The characteristics of the waste rock being used to construct the final landform have been documented in MCP Section 5.5.1.2 The key aspects of waste rock impacting vegetation establishment relate to plant water availability (PAW) and rooting depth.

Waste rock PAW depends on the proportion of fines (<2 mm) in the material as well as the total depth available for plant root establishment. For example, Section 1a of the Trial Landform (TLF) was constructed of material with an average of 33% fines and has been able to successfully establish a native woodland ecosystem; although some specific species have struggled (e.g. *Eucalyptus miniata* and *Acacia mimula*) and adjustments in species mix may be required to ensure the functionality of the target ecosystem is achieved (e.g. using *E. phoenicea*, *E. tintinnans* and *Acacia latescens*).

Monitoring of the TLF and WAVES modelling has indicated that a minimum of 15% fines is sufficient to sustain a native woodland ecosystem (Lu *et al*. 2019). It is understood that material with higher fines will have a greater PAW, act more like a natural 'soil' and be able to support the local, natural woodland ecosystems with fewer adjustments.

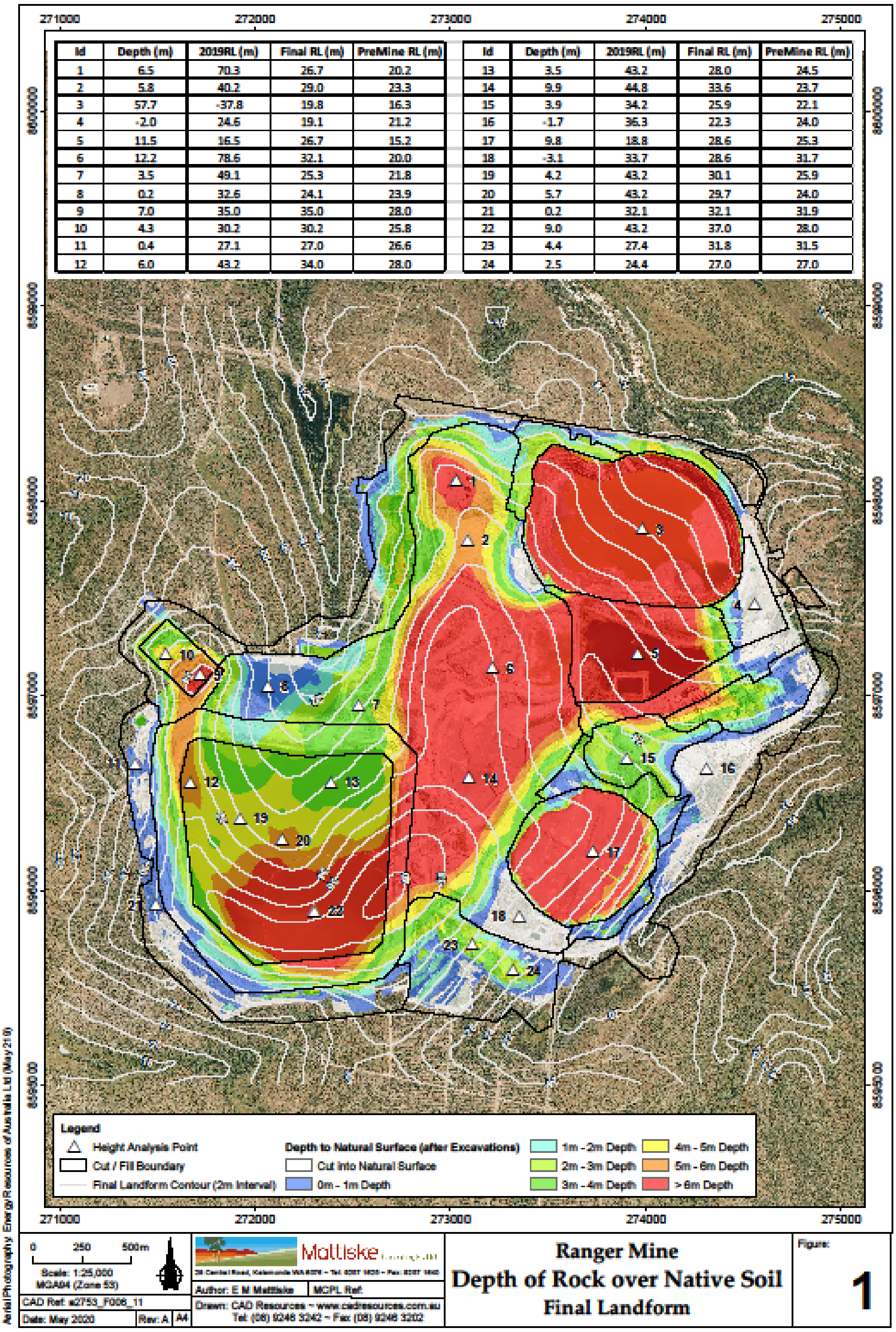
Particle size distribution (PSD) analysis of waste rock in stockpiles indicates that the waste rock ranges between 10%-60% fines (Section 4.1). Mine planning and bulk earthworks processes have been developed to ensure that the material to be placed in the surface growth layers (e.g. up to 6 m depth) of the final landform (FLF) is not below 15% fines and, wherever possible has more fines to optimise PAW.

Whilst it is not possible to exactly predict the PSD of all construction materials and therefore the occurrence of the different PAW 'zones' across the final landform surface, ERA has implemented an execution methodology that will ensure that the nature of the material in the 6 m growth layer is understood prior to final revegetation planning and execution. Once construction and land-forming is completed, and inspection of the planting area will enable the final revegetation plan to identify the most suitable target native ecosystem and propagation and planting execution can proceed.

Except for the backfilled pits and the upper reaches of the final landform, 62% of the final landform has less than 6 m of waste rock overlying natural soils (Figure 2-2 and Figure 2-2). This means that plants in these areas, particularly larger plants with greater rooting depths, will likely be able to access any PAW in these soil and have improved plant-water relations in the late dry season when seasonal stresses are greatest. Plants on the other 38% of the FLF will have at least 6m of waste rock rooting depth available which has been modelled as sufficient to sustain a native woodland ecosystem dependent on the fines proportion (eg. minimum 15% fines) (Lu *et al*. 2019).

Table 2-2: Depth of waste rock over natural soils.

Depth	Area (ha)
Cut into Natural Surface	65
0m - 1m	73
1m - 2m	52
2m - 3m	59
3m - 4m	86
4m - 5m	72
5m - 6m	57
> 6m	283
Total	747



Id	Depth (m)	2019RL (m)	Final RL (m)	PreMine RL (m)
1	6.5	70.3	26.7	20.2
2	5.8	40.2	29.0	23.3
3	57.7	-37.8	19.8	16.3
4	-2.0	24.6	19.1	21.2
5	11.5	16.5	26.7	15.2
6	12.2	78.6	32.1	20.0
7	3.5	49.1	25.3	21.8
8	0.2	32.6	24.1	23.9
9	7.0	35.0	35.0	28.0
10	4.3	30.2	30.2	25.8
11	0.4	27.1	27.0	26.6
12	6.0	43.2	34.0	28.0
13	3.5	43.2	28.0	24.5
14	9.9	44.8	33.6	23.7
15	3.9	34.2	25.9	22.1
16	-1.7	36.3	22.3	24.0
17	9.8	18.8	28.6	25.3
18	-3.1	33.7	28.6	31.7
19	4.2	43.2	30.1	25.9
20	5.7	43.2	29.7	24.0
21	0.2	32.1	32.1	31.9
22	9.0	43.2	37.0	28.0
23	4.4	27.4	31.8	31.5
24	2.5	24.4	27.0	27.0

Figure 2-2: Depth of rock over natural soil.

2.1.1.3 Surface hydrology and subsurface hydrogeology

The main impact of surface hydrology is in the distribution of basins and drainage features across the integrated final landform (Figure 2-3). A range of suitable vegetation will be required to colonise and stabilise these features, from the drier upper reaches down towards where drainage lines develop into riparian creeks. Suitable reference ecosystems will be further investigated and a suitable revegetation plan developed.

Due to differences in hydraulic conductivity of the waste rock of the final landform and the underlying natural soils, modelling indicates that areas around the FLF perimeter may experience extended periods of saturated soils. Although relatively small in areal extent, this scenario would largely preclude the establishment of vegetation of the common regional woodlands which are used to a prolonged dry season each year. It is likely that alternative reference ecosystems will required for these areas, however that is outside this current scope of work.

Similarly, the nature of the subsurface hydrogeology in the area of the Tailings Storage Facility (TSF) will likely be an influence on what vegetation can establish. As agreed through stakeholder consultation, further investigations into these constraints, and identification and collection of suitable reference baseline data, will be conducted.

Emergent vegetative features in constructed waterbodies

The RPA has two wetland filters: the CCWLF (currently in operation) and the RP1 wetland filter (currently removed from operational use).

Valdron Clark (2011) describes the dominant vegetation species in the RP1 wetland filter, describing previous studies of the species on and off the RPA, the historical distribution and abundance of the species in the wetland filters, propagation methods, and their tolerances to environmental factors including water quality and hydrological regimes.

A series of four reports were prepared between July 2013 and November 2014 to chronicle the emergent vegetative features in the two artificially constructed waterbodies and water management sumps on the RPA (Valdron Clark 2013a, 2013b, 2014a, 2014b).

Water quality within the RP1 wetland filter is of pond water quality and water levels closely resemble seasonal cycles (Valdron Clark 2014b). The CCWLF has received inputs of varying water quality since its construction, including rainfall and surface water intercepts from the Southern Stockpile and Corridor Road, pond water permeate, and minor inputs from the TSF and Brine Concentrator (BC) distillate. The influx of distillate into the wetland in October 2013 resulted in recorded temperatures of between 45 and 55 ºC contributing to the dieback of aquatic plants throughout two of the wetland cells. However, aquatic flora species recovered, particularly *Eleocharis* species which demonstrated recruitment of new culms protruding from dead *Eleocharis* beds. For the most part, the wetland has continued to demonstrate resilience, in terms of vegetation establishment, as a response to wet/dry hydrological cycles (Valdron Clark 2014b).

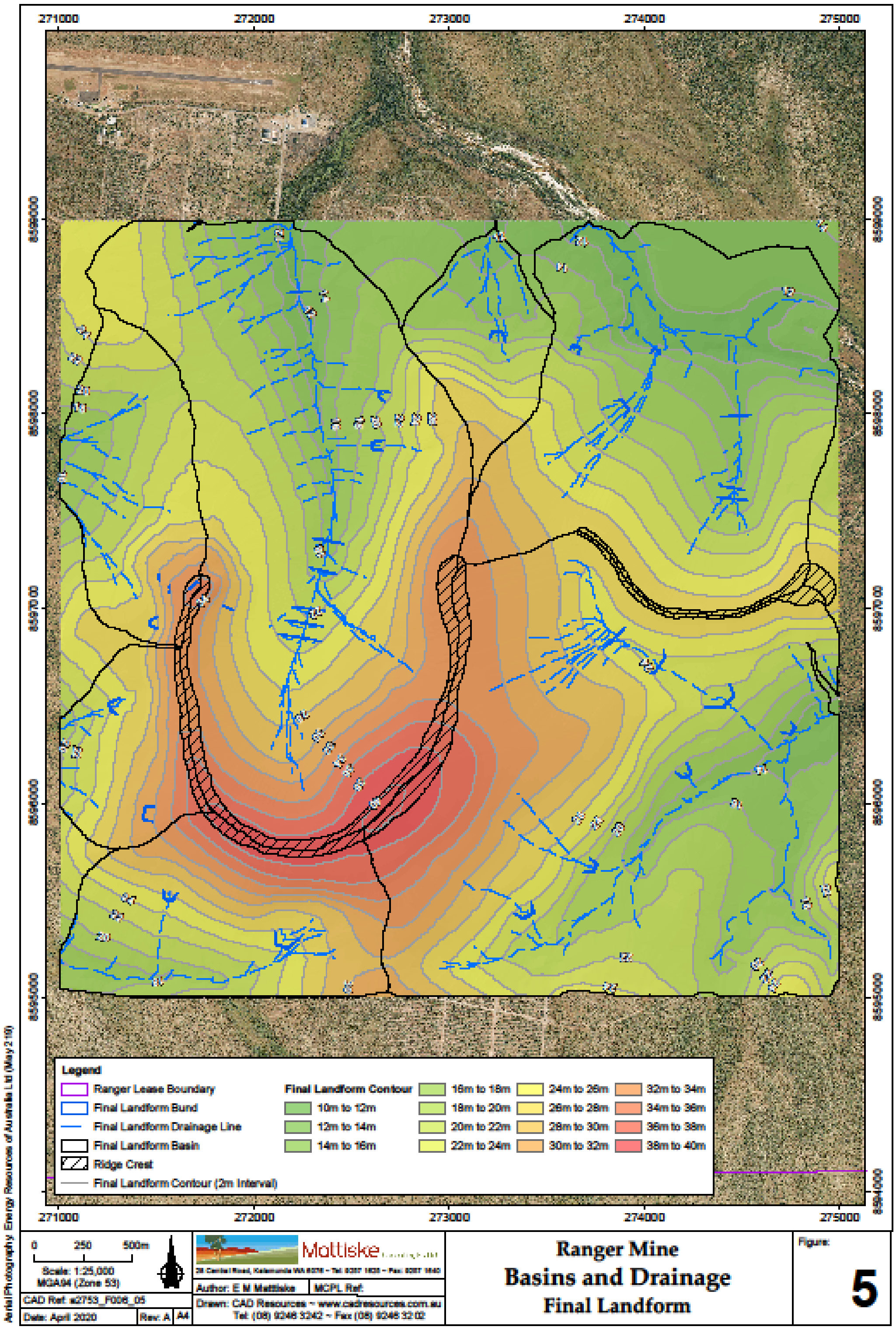


Figure 2-3: Basins and drainage features of the final landform.

Acidic conditions have been recorded in both the RP1 and CCWLF; however, recruitment of new plants and viability appear to have been unaffected by acidic conditions (Valdron Clark 2013b). In addition, frequent sightings of water monitors in the vicinity of both wetland filters suggest these artificial wetland ecosystems in the RPA are functional to some degree (Valdron Clark 2013b).

These reports provide evidence of the natural colonisation and successful establishment of aquatic vegetation habitats within constructed waterbodies on the RPA and an understanding of environmental conditions to support sustainability of these habitats.

2.1.1.4 Substrate chemical status, including nutrients and contaminants of potential concern

As discussed in the 2018 *Cumulative ecological risk assessment for the rehabilitation and closure of Ranger uranium mine* (Bayliss 2018), chemicals in substrates can play a critical role in revegetation success, including: a limiting nutrient; a toxicant above a threshold effects level; a modifier or facilitator of other chemical processes/interactions; or a combination. Overall, the waste rock material at Ranger Mine differs from natural soils by having higher pH, EC, CEC, Mg, total P and SO4 concentrations, and having lower levels of organic carbon. The ecological risk assessment found that risks to revegetation from mine-derived chemicals is assumed zero (Bayliss 2018).

The TLF showed successful vegetation establishment and development with a methodology including application of fertiliser. The current ERA revegetation method also includes provision of a suitable fertiliser upon tubestock planting with a follow-up application in the subsequent wet season.

As part of the technical constraints review, it was identified that areas of potential acid sulfate soils may be present, particularly in areas requiring future 'riparian' revegetation. Studies into this are ongoing and a specific revegetation strategy, including suitable reference ecosystems, shall be developed.

2.1.1.5 Slope and aspect

Whilst slopes and aspects can be significant influences in some mine rehabilitation scenarios, at Ranger Mine almost all slopes are less than 5° and do not require any particularly drastic revegetation treatment. The Ranger Mine rehabilitation plan allows for surface ripping of areas with steeper slopes, which should mitigate against any potential erosion risks.

2.1.2 Identify suitable natural reference sites

2.1.2.1 Targeted surveys of natural ecosystems

A description of the natural vegetation communities and flora and fauna of the region is provided in MCP Section 5.3.3. This section below shall cover surveys undertaken specifically to support development of a conceptual reference ecosystem for Ranger Mine rehabilitation.

The final landform at Ranger Mine is being designed to resemble, and behave in a manner similar to, landforms of the surrounding area, while still providing for the long-term protection of the environment. Based on the likely low-rocky rise features of this landform, most research to date has focussed on identifying and characterising natural ecosystems occurring in comparable landscape locations, for use as appropriate reference ecosystems. There is a range of vegetation community types in areas outside the mine footprint that represent the spectrum of environments likely to be found across the rehabilitated Ranger Mine final landform and Project Area. By understanding the environmental features that are associated with the normal range of native vegetation community types, the conditions required to support these communities and/or the community types that best suit particular environmental conditions of the Ranger Mine final landform, can be identified (Humphrey *et al*. 2009). Understanding environmental features that are associated with the normal range of native vegetation community types (including PAW) informs the design and construction of the Ranger Mine final landform.

Early work by the Supervising Scientist (Needham *et al*. 1973) and NT Land Conservation Unit (Uren 1992) identified a number of locations in the Alligator Rivers Region as being weathered hills composed of Cahill formation schists – likely to be natural sites where both topography and rock type were similar to that expect on the Ranger waste rock final landform.

A Supervising Scientist study by Brennan (1995) compared vegetation found at areas adjacent to the Ranger site and those further afield (but within KNP) with a substrate likely to be more similar to the Ranger waste rock final landform. A key finding was that floristic heterogeneity (among the hill sites) was due to the dissimilarity of their substrates or parent-rock types. As Brennan (1995) states:

> *The concept of site revegetation based on the characteristics of adjacent or pre-existing plant communities has much popular appeal a clear statement of intent to restore disturbed sites to their previous undisturbed state. However, there is a potential problem in applying this concept to guide revegetation on the Ranger Waste Rock Dump (WRD) … The basis of the problem is that the landform and substrate of the WRD are not related to the pre-existing landforms, or to substrates adjacent to it. The WRD … is composed of metamorphic, Cahill-formation schists whereas adjacent substrates belong to a geologically unrelated entity known as the Koolpinyah-surface (Needham et al. 1973, Wells 1979). Given these striking geotopographic differences it seemed reasonable to suggest that native vegetation communities immediately adjacent to the WRD might not contain the most appropriate species for revegetating this area.*

The RPA and surrounding areas of Kakadu NP have been studied extensively over the last sixteen years by ERA and ERISS to obtain information from appropriate reference sites to inform revegetation planning, management and performance objectives and assessment methods (in terms of closure criteria) (eg. Hollingsworth and Meek 2003, Brennan 2005, Hollingsworth *et al.* 2007b, Humphrey 2013, Humphrey & Fox 2010, Humphrey *et al.* 2009, Humphrey *et al.* 2011, Humphrey *et al.* 2008, Humphrey *et al.* 2012; Table 2-3).

Table 2-3: Vegetation Survey Data collected in the Alligator Rivers Region (adapted from Erskine *et al*. 2019).

Reference	n	Date Surveyed	Design	Plot Size and Methods	Plots within 10km radius of Ranger Mine
Conservation Commission (White *et al*. 1985)	77	1979-1981	Unknown	Vegetation present within 50m radius of soil sampling sites. Understorey not collected	36%
Brennan (2005)	20	1991-1993	Stratified Random	Two assessments based on height >1.5m = Ten 20m x 20m randomly placed in 1ha (4000m^2); <1.5m = 20 x 5m x 5m quadrats (400m^2) 25 understorey (0.71m x 0.71m (12.5m^2)	35%
EWLS (Hollingsworth & Meek (2003)	20	2002	Stratified Systematic	For trees and shrubs >2m; 320m x 20m plots (total of 1200m^2) at each site stratified by ecosystem types. 10 understorey x 1m x 1m (10m^2)	100%
Cyclone Monica (Saynor *et al*. 2009)	31	2006	Stratified Random	For trees & shrubs >2m 30m×30m plots (900m2). Understorey not collected.	67%
Hollingsworth *et al*. (2007a)	38	2007	Stratified & mixture of random and systematic	Data from Hollingsworth and Meek (2003) and Brennan (2005)	100%
2010 Survey (Humphrey *et al*. 2012)	54	2010	Stratified Random	For trees & shrubs >2m 20m×20m plots (400m2) plots except site A53 (25m x 20m). Understorey not collected.	100%
2019-2020 (SSB 2019a)	12	2019-2020	Stratified and Random	For Trees and Shrubs: >1.5m , <1.5m on Transects in 1ha. Density of Stems and % Cover Understorey presence absence and cover. SSB S1 to SSB S10 from within 10km radius of the Ranger mine and SSB G1 and SSB G2 from part of the Georgetown area south-east of RPA.	100%

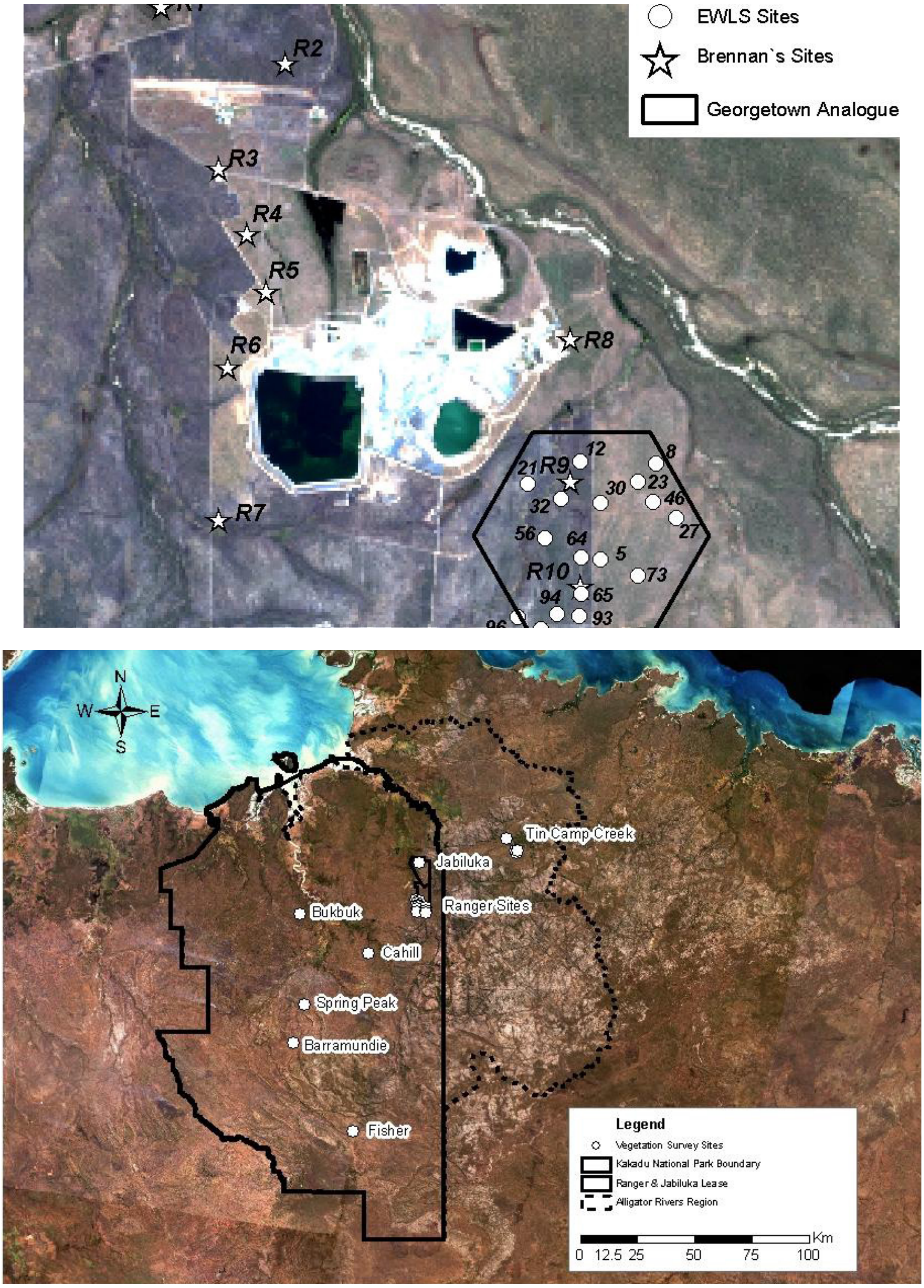


Figure 2-4: Maps of plant analogue sites surveyed by Brennan (2005) (top and bottom) and (Hollingsworth *et al.* 2003a) (bottom) (source Humphrey *et al.* 2006).

An area of focus has been the ‘The Georgetown Creek Reference Area’ (the hexagon in Figure 2-4), chosen because it is representative of nearby Kakadu NP habitats that are considered appropriate for a rocky final landform (Hollingsworth *et al*. 2003a). Early work focussed on describing the detailed geomorphic and pedological characteristics of different units that were present and on relating these to compositional and structural features of their vegetation cover (Hollingsworth *et al.* 2003, Hollingsworth & Meek 2003).

Extensive surveys of Georgetown Creek Reference Area have been completed, including a 400 ha grid survey (at 200 m spacing) that has shown graphically the natural variability of the vegetation types across the analogue area (Hollingsworth & Meek, 2003; Figure 2-5). Monitoring plots in Figure 2-5 are coloured according to vegetation type:

- Pink: Tall *Eucalyptus tetrodonta* open forest
- Yellow: Tall *E. bleeseri* and *E. tetrodonta* mixed open woodland
- Blue: Mid-high *Melaleuca viridiflora* open woodland
- Green: Tall *E. tetrodonta*, *E. miniata* and *E. tectifica* open woodland
- White: Tall *E. tetrodonta*, *E. miniata*, *E. setosa*, and *E. porrecta* open forest
- Brown: Tall *E. foelscheana, E. tetrodonta* and *E. confertiflora* mixed open woodland
- Red: Mid-high *E. confertiflora, E. tectifica* and *E. foelscheana* open woodland

The soils in the Georgetown Creek Reference Area vary in their drainage status and are typically gravelly and less than one metre deep to parent rock. The variation in the plant communities is typical of the lowland regional surface (Russell-Smith 1995) and there is a strong response to drainage and water supply (Williams *et al*. 1996). The structure and composition of the Georgetown Creek Reference Area vegetation is likely to be governed principally by water availability and plant available nutrients, typical of northern Australian savanna (Williams *et al*. 1996). Key geomorphic features (including parent material, slope, effective soil depth etc.) are also important. However, more subtle variations in the vegetation composition and structure are likely to be the result of interplay between historic factors, proximity and context (i.e. the surrounding vegetation types) and discrete, and often localised, disturbance events.

Given the variation in PSD of the TLF (as discussed in Section 4.1) some degree of variation in PSD is expected in the source rock for the Pit 1 final landform cover and therefore the surface layer. The environmental characteristics that influence variation in plant communities, as discussed above, are likely to also vary across the Pit 1 final landform cover and result in the heterogeneous combination of vegetation communities observed in Ranger reference sites.

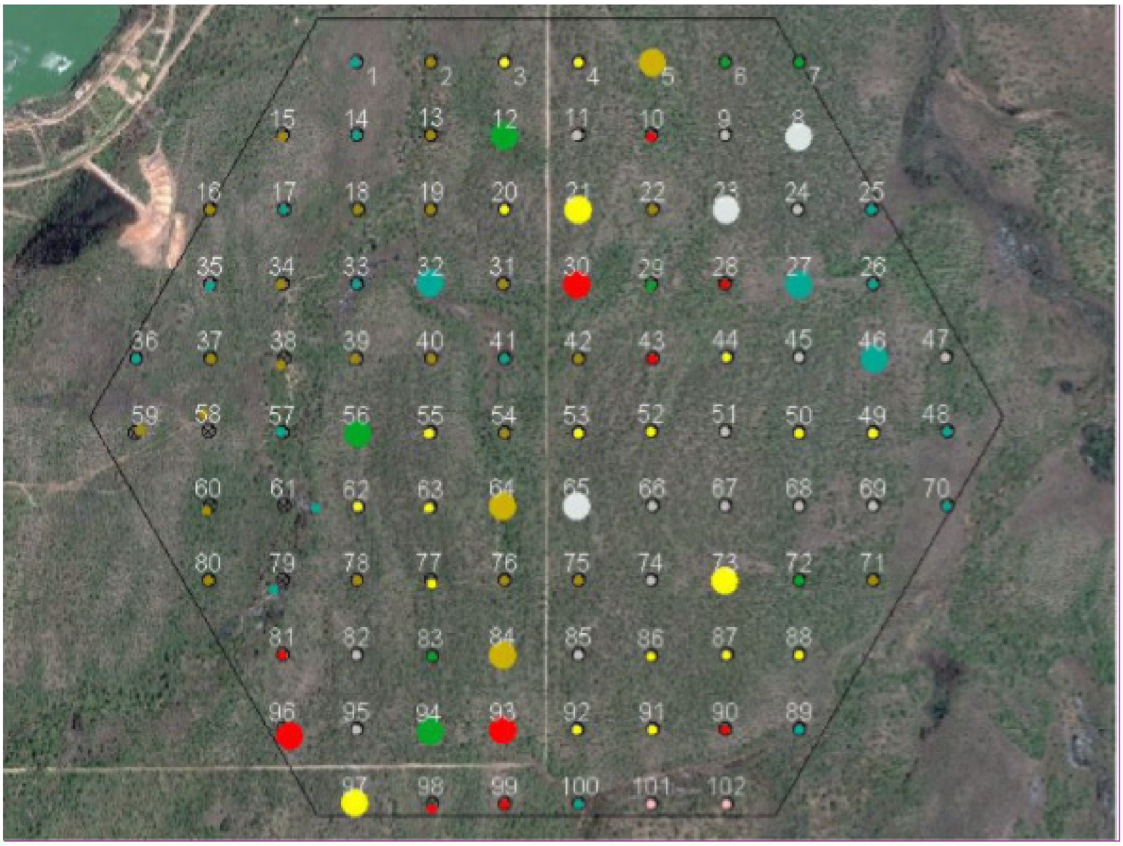


Figure 2-5: Georgetown Creek Reference Area vegetation type variation across monitoring sites

Multivariate classification of the vegetation communities surveyed in Georgetown (Hollingsworth & Meek 2003) and by Brennan (2005) resulted in four broad vegetation types based (Humphrey *et al.* 2012) (Figure 2-6).

Gardener *et al*. (2007) has described ecological attributes of each of the three community groups using species phenology, including growth form, life history, time to maturity, response to fire, type of re-sprouting and deciduousness. In general, all three communities have similar attributes, i.e. an even mix of tree and shrub species, comprising mostly long lived perennials and able to re-sprout after fire. The only attribute that differed was the relative contribution of deciduous species, with the drier community having a greater proportion of deciduous species.

This finding agrees with other studies in KNP; for example as part of the long-term, Kapalga experiment, Cook (unpublished data) demonstrated that soil depth, most likely through the mechanism of water availability during the dry season, is a major driver of tree stand structure (Cook 2020 *in draft*). The data show that evergreen trees increased in basal area as soil depth increased, but deciduous trees showed no significant variation with soil depth (Figure 2-7).

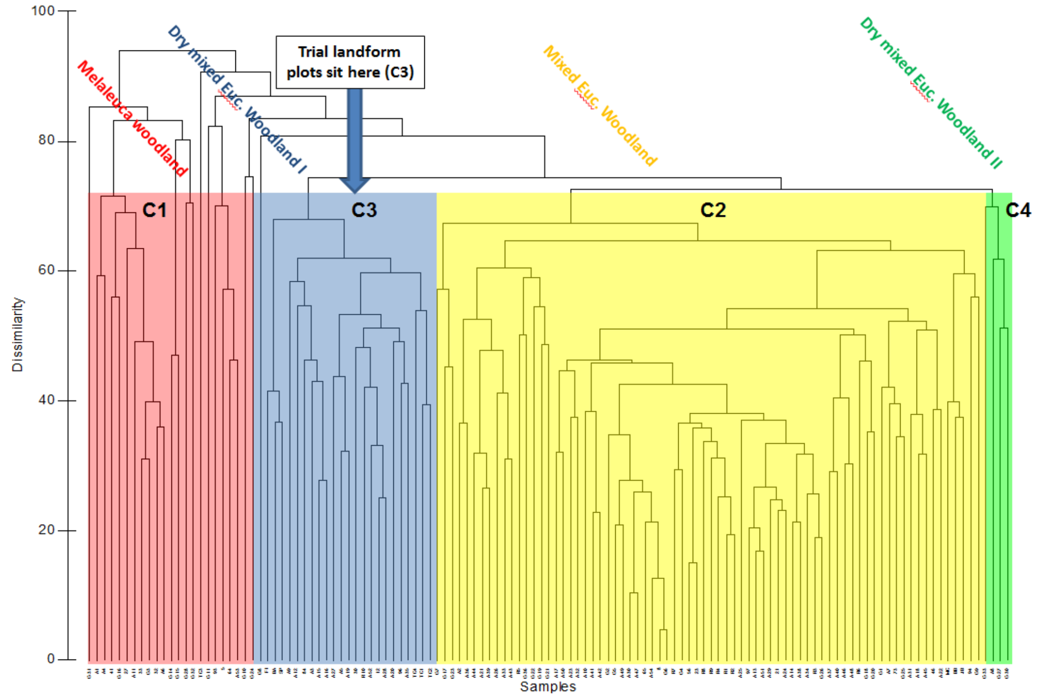


Figure 2-6: Cluster analysis (group average linkage) of trees and shrubs data for Alligator Rivers Region vegetation analogue sites. [Vegetation data log transformed density/ha units (Humphrey *et al.*, 2012).]

Table 2-4: Descriptions of the Ranger Mine analogue communities

Broad vegetation community	**Dominant and/or distinguishing tree or shrub species**	**Classification unit[1]**
Melaleuca woodland	*Melaleuca viridiflora, Pandanus spiralis, Planchonia careya*	C1
Mixed eucalypt woodland	*Acacia mimula, Eucalyptus tetrodonta, Corymbia porrecta, E. miniata, Xanthostemon paradoxus, Terminalia ferdinandiana*	C2
Dry mixed eucalypt woodland: Type 1	*Corymbia foelscheana/latifolia, X. paradoxus, T. ferdinandiana, P. careya, Cochlospermum fraseri*	C3
Dry mixed eucalypt woodland: Type 2	*Terminalia pterocarya, Acacia mimula, X. paradoxus, C. disjuncta, E. tectifica*	C4

[1] Humphrey *et al.* (2012); Figure 2A

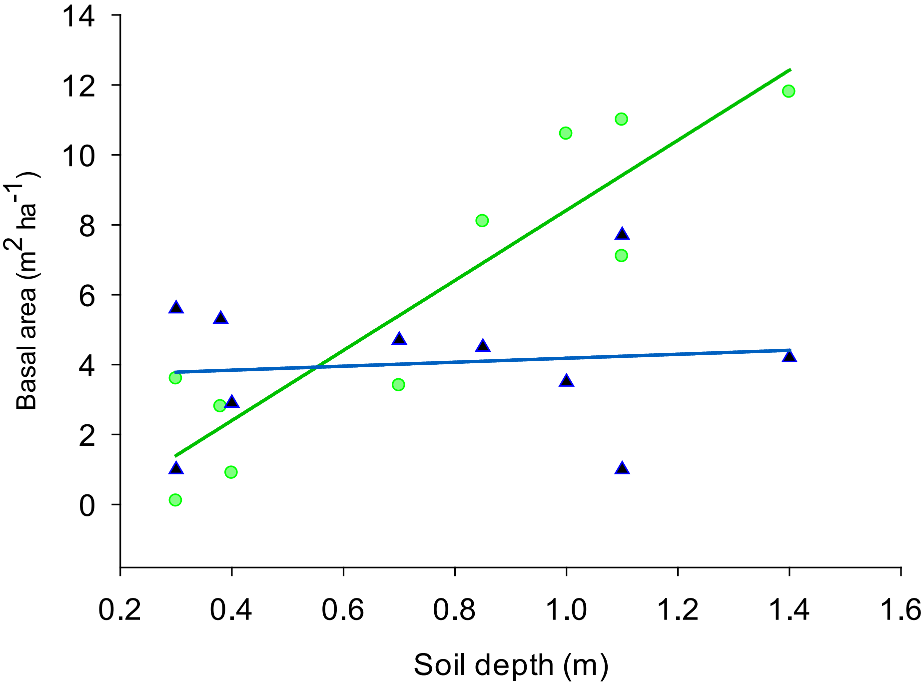


Figure 2-7: Variation in the basal area of evergreen trees (●) and deciduous trees (▲) in relation to soil depth along downslope catenary sequences at Kapalga in Kakadu National Park (Cook 2020).

In 2018/19, SSB surveyed 12 new one-hectare vegetation reference plots (including 2 sites within the Georgetown Creek Reference Area) from within a 10 km radius of the mine site. Multivariate ordination of overstorey cover data showed that nine of the sites (and ten sites if using overstorey stem densities) classified (in cluster analysis) with the dominant savanna woodland type for the local lowlands (SSB 2019a), termed 'mixed eucalypt woodland' *sensu* Humphrey *et al*. (2012). Data from these sites are interim, pending acquisition of data from larger scales, and will be useful to informing ongoing refinement of closure criteria and assessment by ERA.

2.1.2.2 Fauna baseline monitoring

A variety of flora and fauna studies in the RPA and surrounds have been conducted for purposes not specifically related to mine closure. Flora and fauna surveys conducted prior to 2012 were reviewed by ENV Australia Pty Ltd (Firth 2012) during the PFS for the Ranger 3 Deep mine development. Firth (2012) reviewed 18 flora survey reports, 26 fauna survey reports, three aquatic flora and fauna survey reports and seven reviews of previous terrestrial and aquatic flora and fauna work.

The establishment of habitats on the final landform that support fauna assemblages similar to Kakadu NP and contain culturally important fauna species is predominantly dependent on the success and final composition of the revegetation. Monitoring of the final landform and reference sites will provide data to determine trends in the composition and abundance of fauna.

Colonisation of revegetated areas by fauna of all trophic levels is critical for the healthy functioning of the ecosystem and its long-term self-sustainability (Corbett, L 1999). Successful fauna recolonisation primarily depends on the proximity to the source of the fauna and availability of suitable habitats within revegetated areas. The final landform will be surrounded by relatively healthy woodland and is therefore close to the source of native fauna. The vegetation will be established to a standard similar to the surrounding natural woodland, therefore the habitats are expected to not prohibit the natural colonisation of fauna.

Extensive fauna studies on historical revegetation trial areas on waste rock dumps in the RPA (Corbett, L 1999) demonstrated that the array of vertebrate fauna living on the revegetated waste rock dumps was typical to that found in similar habitats of Kakadu NP and that the density of frogs, native mammals and invertebrate groups was generally higher on the waste rock dumps than in similar habitats in Kakadu NP. One exception was the absence of possums and other arboreal groups on the waste rock dumps, which was probably due to the absence of extensive stands of mature trees with hollows. Such habitats will develop with further time. L. Corbett (1999) concluded that the prognosis for the Ranger Mine is that rehabilitated landforms are likely to be recolonised with representative populations of vertebrates and many invertebrates within five years of decommissioning. One of the major reasons for the relatively high fauna density on the waste rock dump was *"… good feral animal control to minimise predator impacts on founder populations."*

In 2011 ERA initiated and implemented a long-term fauna and flora monitoring program (Zimmermann 2013a) on the RPA and, in agreement with Mirrar Traditional Owners and Kakadu NP Management, in adjacent areas of Kakadu NP. The primary objective of the program was to provide crucial information about the natural woodland ecosystem (potential revegetation target habitats) for the development of realistic closure criteria. The fauna and flora monitoring program aims to establish baselines of the long-term dynamics, seasonal fluctuations and responses to natural disturbances such as fire or cyclone. This will provide the closure criteria with the spatial and temporal variations that can be expected in the natural woodland ecosystems. It also provides valuable information about ecosystem resilience, natural recruitment, self-sustainability, the relationship between habitat complexity and fauna, impact of weed incursion and many other factors crucial for the assessment of revegetation success.

Future monitoring was committed to be undertaken in close collaboration with SSB/ERISS, just as monitoring site selection had been. The site selection process, criteria of the monitoring program and initial site survey were detailed in Zimmermann (2013a).

Site selection criteria were developed to ensure that monitoring objectives are met, and data are comparable and meaningful. The criteria for site selection included: vegetation community (similar to those to be established on the final landforms), fire regime (captures variability of vegetation communities under different fire regimes), surface geology/soils (similar to those identified in the final landform vegetation communities), position in the landscape (captures the variability in crest, upper/mid/lower slope vegetation communities), cultural heritage (no impact on cultural heritage), access (easy access during all seasons and in the long term) and weed status (weed free at time of establishment). The criteria were consulted with relevant stakeholders and experts.

Based on the above criteria, suitable areas were identified on the RPA and in adjacent Kakadu NP. Approval (Kakadu NP permit) was granted from ERA for the RPA and from Traditional Owners and Kakadu NP Management for Kakadu NP to inspect these pre-selected areas and select suitable monitoring sites within them. The monitoring program and the pre-selected areas in Kakadu NP were presented to the Gunjeihmi Aboriginal Corporation (GAC) as part of the approval and consultation process.

A total of 17 monitoring sites were selected for the Ranger Mine long-term fauna and flora monitoring program, with 11 sites located on the RPA and six in the surrounding Kakadu NP (Figure 2-8). The sites fulfil all selection criteria.

The monitoring sites provide a good representation of the fire frequencies of the region. On the RPA two sites have experienced a high, three a low and six a low to medium fire frequency in the last 10-12 years. In the surrounding Kakadu NP four sites have had high and two sites low fire frequencies. Of the two broad vegetation communities identified as target habitats for the Ranger Mine revegetation, 14 monitoring sites were Mixed Eucalypt Woodland and three were Dry Mixed Woodland. The latter was not found outside of the RPA Georgetown area.

The sites are positioned on the crest, mid and lower slope representing the variation in vegetation communities derived from position in the landscape. All selected sites are weed and disturbance free, accessible and do not impact on cultural heritage and Kakadu NP values.

In 2016, Eco Logical Australia (ELA) was engaged by ERA to undertake the first full flora and fauna survey of the above monitoring sites (S. Smith 2016)

The 2016 data provided an indicative assessment of the condition of native woodland in the areas surrounding the mine footprint. The results indicated natural variability in undisturbed sites resulting from seasonal changes and in some cases fire. No other disturbances (e.g. cyclones) impacted sites between surveys.

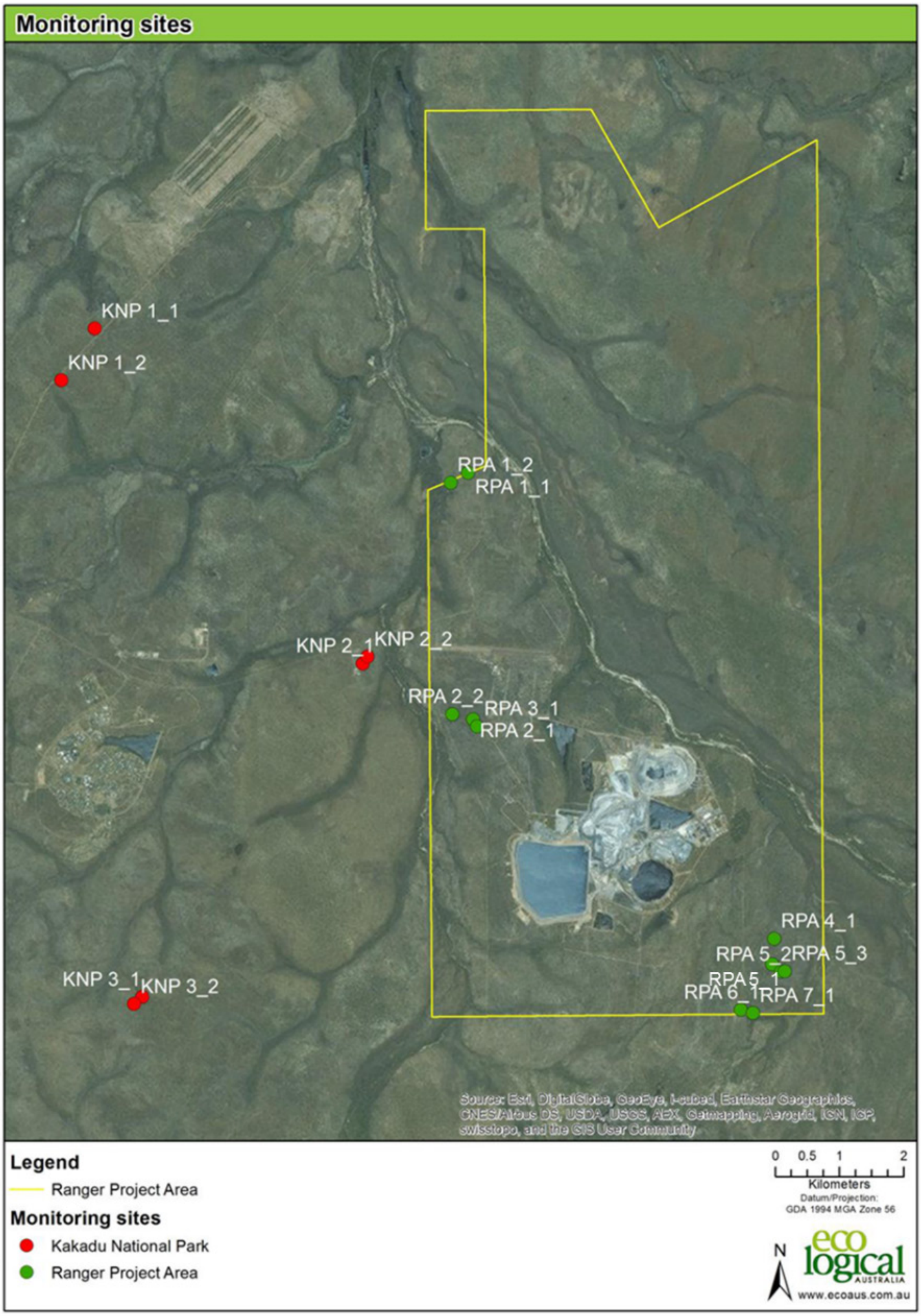


Figure 2-8: Survey sites

2.1.3 Proposed conceptual reference ecosystems for ERA Ranger Mine

ERA is collaborating with key stakeholders to define appropriate conceptual reference ecosystems and develop agreed closure criteria for the rehabilitation of Ranger Mine. Ten of the SSB 2018/19 surveyed woodland sites have been nominated as representing the 'Initial Conceptual Reference Ecosystem' (ICRE), based on which ERA may further develop additional (or alternative) 'Agreed Reference Ecosystems' (ACREs) that take into account the various constraints of the final landform.

In late 2019, ERA commissioned Dr. Libby Mattiske, a renowned expert in the field of mine site rehabilitation, monitoring and assessment, to review the available data for Ranger Mine, compare these to benchmarked approaches from other operations and jurisdictions, and recommend an updated approach to developing conceptual reference ecosystem/s and resultant closure criteria for ERA. The resultant report (Mattiske & Meek 2020 *in draft*) is summarised in the following sections and covers the integration of available datasets, the results of analyses undertaken, and presents the proposed descriptive closure criteria, supported by a benchmarking exercise and other information. This work builds on many years of research efforts with an emphasis on the current local and regional values that may influence the selection of appropriate species and communities for the rehabilitation areas predicted on the Ranger site. It also places such information into the context of the constraints to the values on the post-mining site conditions with regard for current industry practices for rehabilitation management and objective setting.

The data sets from the various studies to date were integrated and a series of analyses undertaken on the representative subsets of data to clarify a potential way forward to maximise the use of the datasets but also to refine the suitable species and community structural and floristic combinations that might assist in the revegetation assessments and adaptive management programs.

The survey data was integrated with a reliance particularly on stem numbers of the overstorey and midstorey species in line with the initial emphasis on the key framework species of the ecosystems in the Ranger area. Although some analyses were initially undertaken on the presence/absence datasets, this report concentrates on the key overstorey and midstorey species due to the greater consistency between researchers and the need to concentrate on these species for the initial revegetation works on the Ranger Mine. This initial focus also avoids the constraints of variations in seasonal conditions at the time of samplings and the complexity of different lifeforms as evident in the summary of the flora (Mattiske & Meek 2020 *in draft*).

From an initial review of dominant tree species for the SSB sites 1 to 10, it was apparent that there was significant variation in the number of stems for the respective overstorey species, Figure 2-9. This supports the degree of local variation in the sites and communities near the Ranger operations that have been apparent in previous studies. Whilst to date there has been an effort to concentrate on the dominant *Eucalyptus tetrodonta* and *Eucalyptus miniata* tree species, the results as illustrated in Figure 2-9 reflect variations in these species alone, let alone some of the other overstorey and midstorey species.

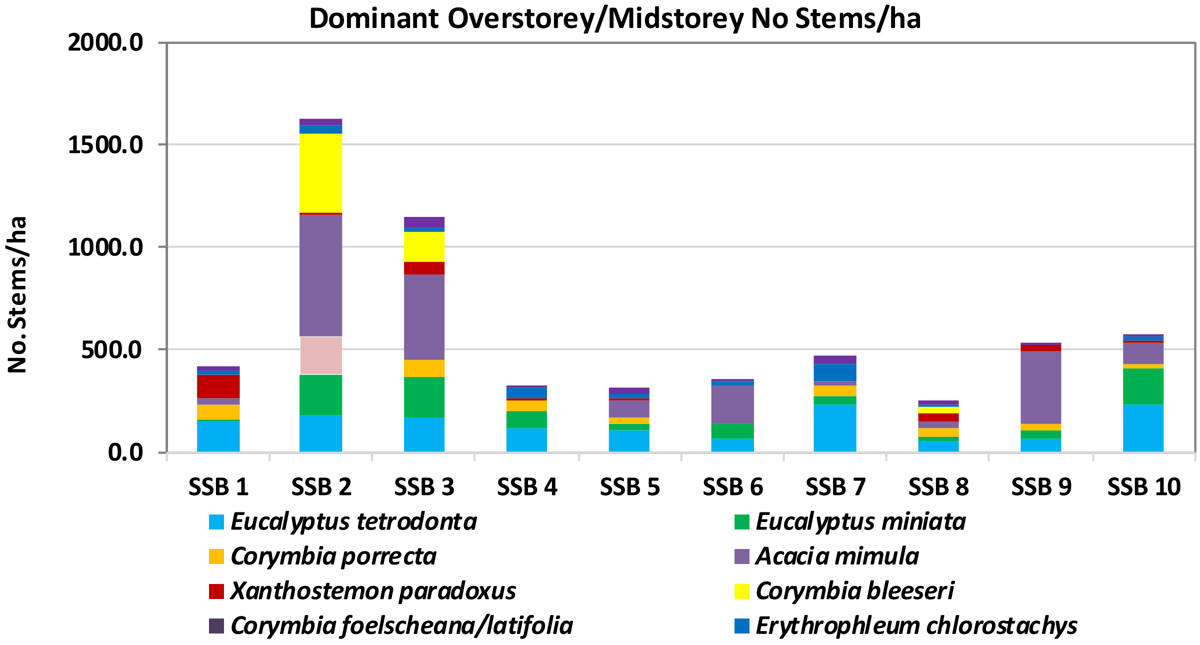


Figure 2-9: Review of Total Stem Numbers / ha on the SSB sites near Ranger (2019/2020 data).

The following dendrogram summarises the results from the analysis using Clarke and Gorley (2015) Primer version 7.0.13 with a Bray Curtis similarity of all overstorey and midstorey stems for the ten SSB sites which concentrated on the *Eucalyptus tetrodonta-Eucalyptus miniata* woodland communities which is proposed by SSB as the ICRE (Initial Conceptual Reference Ecosystem), Figure 2-10. This approach supports the trends in the dominant overstorey/midstorey species as summarised in Figure 2-9 and reflects the subgroups of these woodlands based on all stems of overstorey and midstorey species.

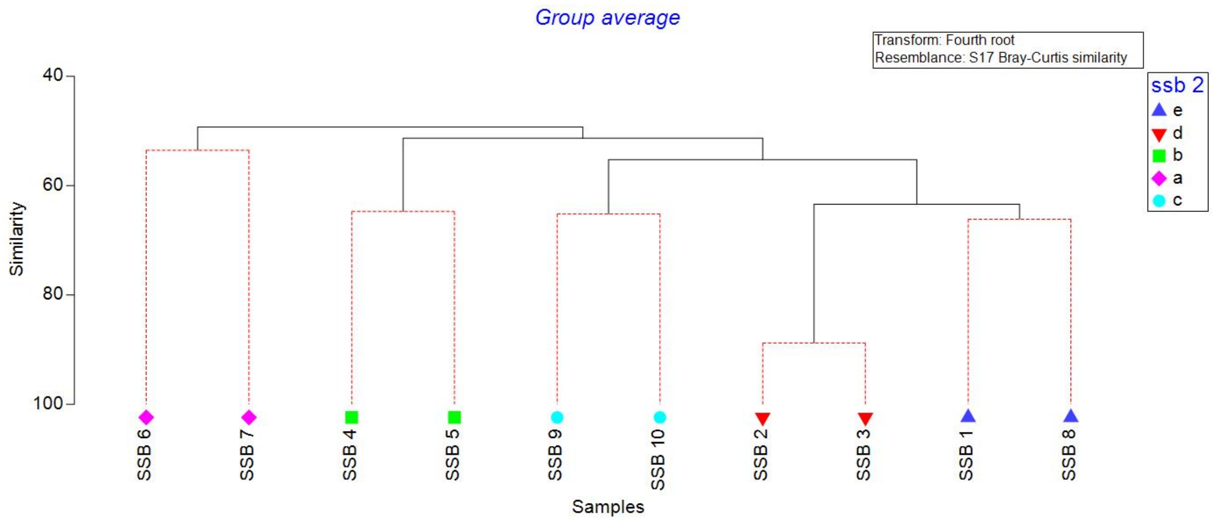


Figure 2-10: Dendogram illustrating similarity of subgroups from SSB sites near Ranger (2019/2020 data) utilising stems/ha of overstorey/midstorey species.

The data was then analysed using Clarke and Gorley (2015) Primer version 7.0.13 using Bray –Curtis similarity using the combined data from SSB (2019/2020), Humphrey *et al.* (2012) (A1 to A54 – designated as GTX A1 to GTX A54), Georgetown sites by Hollingsworth and Meek (2003) (H1 to H97, see E sites in Hollingsworth *et al.* (2007a)) and nearby sites of Saynor *et al.* (2009) (G1 to G36 noting that a few sites were missing in the series of 31 sites). Brennan (2005) sites were excluded due to the variation noted in Hollingsworth *et al*. (2007a); although these should be considered in future variations for potential extreme and localised site conditions that might arise on the RPA.

As indicated in the dendrogram (Figure 2-11) the data from some Georgetown woodland sites align with the SSB Eucalypt woodlands. Consequently these results support the combination of the SSB sites with other sites to broaden the coverage and also to allow for variations on site conditions on the RPA which may not support selected species (e.g. *Eucalyptus miniata*, due to lack of soil water holding capacity) and may support other species (e.g. *Eucalyptus tectifica* that are more drier site tolerant). The results from this modified combination as a subset of the large set of sites is summarised in Figure 2-12. These results enabled the refinement of 4 possible groupings of the Eucalypt woodland communities (ICRE based on SSB 1 to 10). A slightly modified ICRE (ACRE v1) and a modified ACRE v2 which supports species and a community that is wider in representation and ACRE v3 which includes species that may be more tolerant of drier site conditions on the RPA.

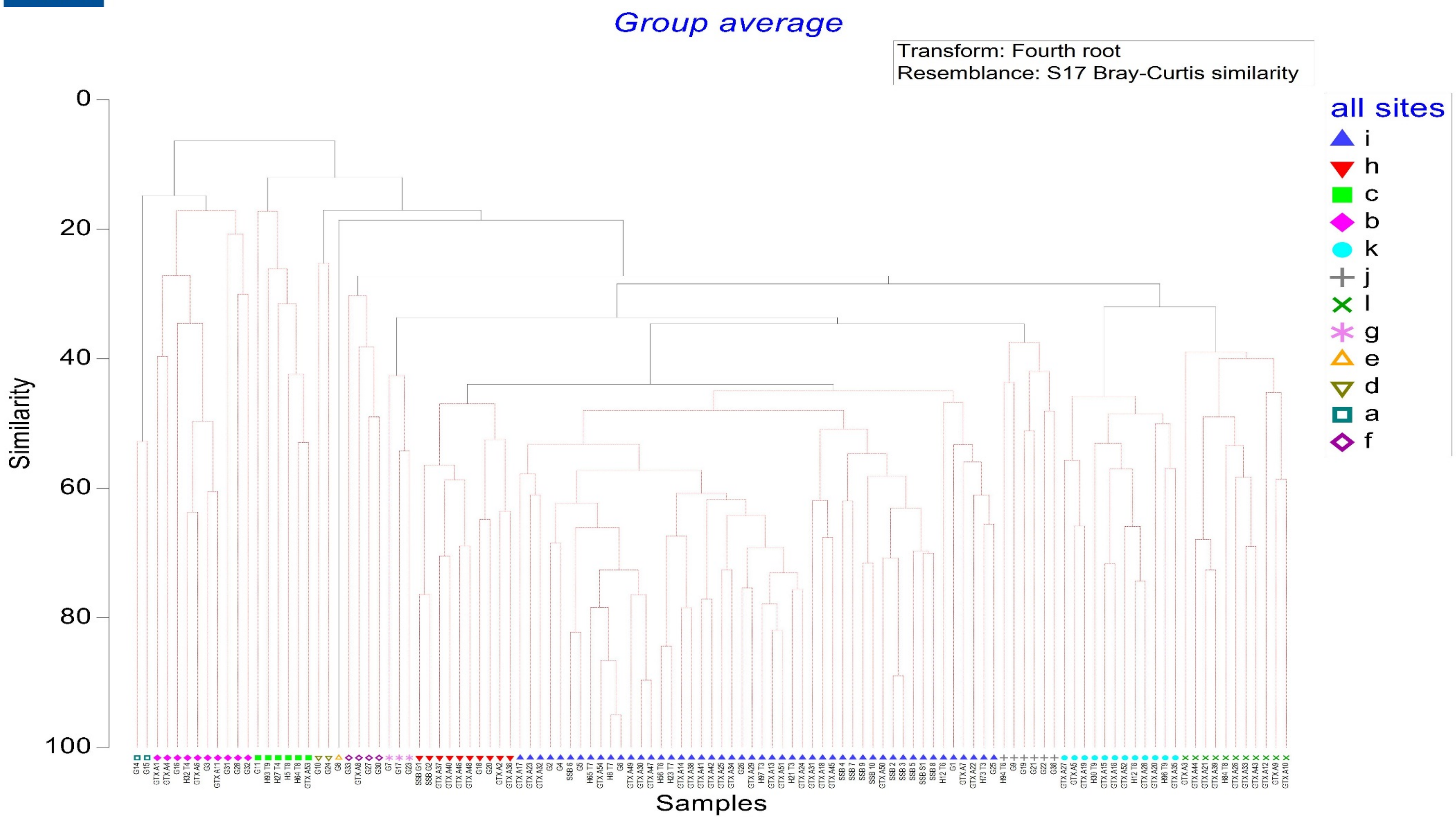


Figure 2-11: Dendogram illustrating similarity of SSB sites near Ranger (2019/2020 data) and all of Saynor et al. (2009) and Georgetown (Hollingsworth & Meek 2003, Humphry et al (2012) using stems/ha overstorey/midstorey species (Mattiske & Meek 2020).

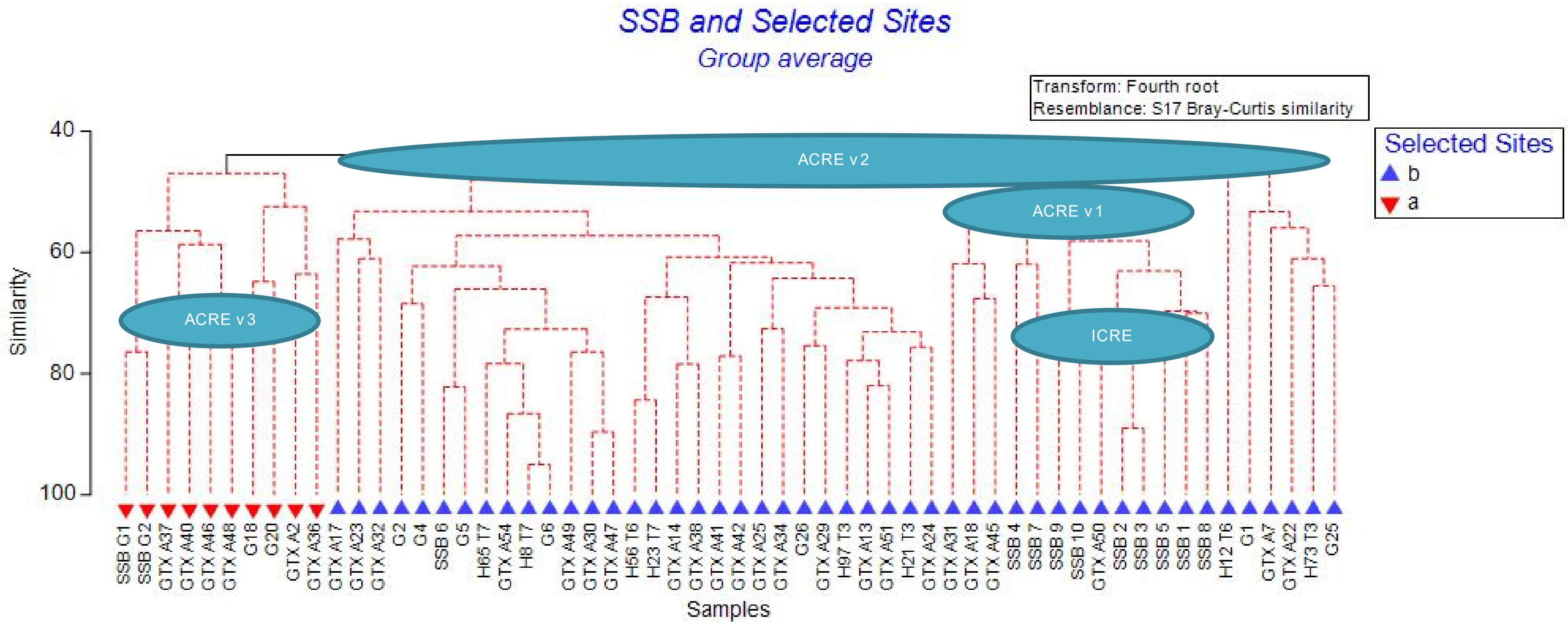


Figure 2-12: Dendogram illustrating similarity of a subset of SSB sites near Ranger (2019/2020 data), Saynor et al. (2009) and Georgetown (Hollingsworth & Meek 2003, Humphry et al (2012) using stems/ha of overstorey/midstorey species (Mattiske & Meek 2020).

These results support the approach of combining the SSB sites with selected sites from within and near the RPA. This has led to a modified list of framework species for the Eucalypt woodlands as proposed to align with the ICRE (SSB 1 – 10) dominant species; IRCE slightly modified (ACREv1) (species from grouping within the “b” group) on Figure 2-11, a modified ACRE v2 which supports species and a community that is wider in representation and a ACREv3 which allows for the inclusion of species that tolerate drier sites (a modification of group a), see Table 2-5. The lower contribution of *Eucalyptus miniata* and the contribution of *Eucalyptus tectifica* are within the ACREv3 potential option.

Such an approach can be used to refine and adapt the framework overstorey and midstorey species following rehabilitation trials. As such it can also be used to delineate and refine completion criteria from other datasets associated with the different lifeforms, midstorey and understorey species.

The inclusion of a wider range of sites is beneficial in view of the variation within the local woodland which is evident from the initial analyses on the SSB sites 1 to 10 as well as on the wider area within RPA and KNP. The inclusion of Georgetown and other sites near the RPA have assisted in the process to date and as more site specific conditions necessitate a similar approach could be undertaken for the other parameters.

ERA will take the opportunity of the 2021 Pit 1 revegetation trial to plant out areas with these different CREs as detailed in Section 6.1 below. This will enable monitoring for their suitability for revegetating waste rock landforms and also provide an opportunity to visually demonstrate the different ecosystem types to Traditional Owners and external stakeholders.

Table 2-5: Dominant Overstorey and Midstorey species for the ICRE and proposed ACREs (species order by descending mean stems/ha ± SE) (Mattiske & Meek 2020 – *in draft*).

ICRE (SSB Sites 1 – 10)		**ACREv1 (13 sites)**		**ACREv2 (48 sites)**		**ACREv3 on drier sites (10 sites)**	
Acacia mimula	182.8 ± 64.8	*Acacia mimula*	174.6 ± 60.4	*Acacia mimula*	393.4 ± 53.1	*Acacia mimula*	304.1 ± 72.4
Eucalyptus tetrodonta	140.1 ± 21.3	*Eucalyptus tetrodonta*	125.8 ± 18.9	*Eucalyptus miniata*	106.0 ± 21.6	*Eucalyptus tetrodonta*	188.8 ± 65.1
Eucalyptus miniata	86.6 ± 23.9	*Corymbia porrecta*	77.5 ± 19.8	*Eucalyptus tetrodonta*	103.3 ± 10.0	*Corymbia foelscheana/latifolia*	150.5± 57.5
Corymbia bleeseri	57.6 ± 39.3	*Livistona humilis*	67.8 ± 28.4	*Xanthostemon paradoxus*	76.7 ± 15.8	*Xanthostemon paradoxus*	137.6 ± 41.6
Corymbia porrecta	56.2 ± 15.6	*Eucalyptus miniata*	65.0 ± 20.9	*Corymbia porrecta*	75.2 ± 9.6	*Terminalia pterocarya*	115.6 ± 23.66
Livistona humilis	50.6 ± 32.8	*Xanthostemon paradoxus*	58.8 ± 16.0	*Corymbia bleeseri*	28.7 ± 10.2	*Corymbia porrecta*	59.8 ± 19.6
Xanthostemon paradoxus	28.9 ± 10.4	*Corymbia bleeseri*	46.2 ± 30.5	*Terminalia ferdinandiana*	28.1 ± 4.9	*Terminalia ferdinandiana*	41.0 ± 15.2
Erythrophleum chlorostachys	28.3 ± 7.5	*Erythrophleum chlorostachys*	41.0 ± 11.3	*Livistona humilis*	20.7 ± 8.6	*Corymbia disjuncta*	40.5 ± 15.3
Terminalia ferdinandiana	22.9 ± 4.8	*Terminalia ferdinandiana*	38.3 ± 10.2	*Erythrophleum chlorostachys*	18.0 ± 4.8	*Eucalyptus miniata*	24.1 ± 15.0
Persoonia falcata	15.7 ± 5.3	*Planchonia careya*	12.0 ± 6.6	*Melaleuca viridiflora*	10.4 ± 10.4	*Buchanania obovata*	22.4 ± 6.2
Acacia aulacocarpa	8.7 ± 8.3	*Buchanania obovata*	11.7 ± 3.8	*Planchonia careya*	10.3 ± 3.1	*Corymbia bleeseri*	18.9 ± 14.8
Buchanania obovata	7.9 ± 1.9	*Persoonia falcata*	10.5 ± 4.5	*Corymbia foelscheana/latifolia*	7.8 ± 2.8	*Calytrix exstipulata*	14.5 ± 14.4
Acacia oncinocarpa	5.4 ± 5.0	*Acacia aulacocarpa*	6.7 ± 6.4	*Corymbia dunlopiana*	7.3 ± 5.1	*Cochlospermum fraseri*	14.3 ± 10.2
Brachychiton megaphyllus	3.7 ± 2.0	*Syzygium eucalyptoides bleeseri*	6.4 ± 4.1	*Persoonia falcata*	6.6 ± 1.7	*Eucalyptus tectifica*	11.9 ± 7.1
Pandanus spiralis	3.3 ± 2.2	*Brachychiton megaphyllus*	4.8 ± 2.3	*Syzygium eucalyptoides bleeseri*	6.4 ± 4.8	*Pouteria arnhemica*	10.8 ± 5.8
Cochlospermum fraseri	3.1 ± 2.0	*Acacia oncinocarpa*	4.1 ± 3.8	*Calytrix exstipulata*	5.9 ± 4.7	*Gardenia megasperma*	10.6 ±5.1
Planchonella arnhemica	3.0 ± 1.2	*Jacksonia dilatata*	3.8 ± 3.8	*Corymbia setosa*	5.7 ± 3.7	*Planchonia careya*	9. 4 ± 5.1
Stenocarpus acacioides	3.0 ± 1.2	*Planchonella arnhemica*	3.8 ± 3.8	*Buchanania obovata*	4.6 ± 1.4	*Grevillea mimosoides*	8.0 4 ± 5.5

Table 2-6: Selection of Overstorey and Midstorey Stems/ha and Species Richness data of each Reference Ecosystem (Mattiske & Meek 2020 – *in draft*).

	Summary Data					
ICRE	**n**	**MIN**	**MAX**	**MEAN**	**MEDIAN**	**SE**
TOTAL Stems / ha	10	304	1954	725	511	167
Framework stems/ha	10	147	989	369	278	85
No. OS/MS framework spp.	10	4	5	4	4	0
No OS/MS spp.(all)	10	10	22	17	18	1
ACREv1						
TOTAL Stems / ha	13	304	1954	783	648	13
Framework stems/ha	13	147	989	356	299	13
No. OS/MS framework spp.	13	3	5	4	4	13
No OS/MS spp.(all)	13	9	22	15	17	13
ACREv2						
TOTAL Stems / ha	38	354	2100	993	900	79
Framework stems/ha	38	50	950	321	275	31
No. OS/MS framework spp.	38	2	5	3	3	0
No OS/MS spp.(all)	38	3	10	8	8	0
ACREv3						
TOTAL Stems / ha	10	500	2200	1219	1056.5	200
Framework stems/ha	10	50	1475	499	440.5	134
No. OS/MS framework spp.	10	1	8	4	4	1
No OS/MS spp.(all)	10	6	30	13	11	2

2.2 Development of Closure Criteria

Closure criteria are the qualitative or quantitative standards of performance used to measure the achievement of the rehabilitation closure objectives for the closure of the site and needed for the relinquishment of the mining lease (WA EPA 2006). They are usually expressed relative to a reference ecosystem (Young *et al.* 2019b) or, as has been covered in the preceding sections, a series of appropriate conceptual reference ecosystems adjusted to account for the known, or anticipated, constraints of the post-mining landscape.

The process of developing closure criteria is underpinned by the analyses of both analogue sites in appropriate reference ecosystems as well as the analysis of rehabilitation data sets during the initial and ongoing phases of rehabilitation activities with the continual need for adaptive management at different phases from initial establishment and growth to achievement of trajectories of key parameters towards specific closure criteria.

As part of the 2020 review, Mattiske & Meek undertook a benchmarking exercise of the approach to reference site selection and derivation of qualitative and/or quantitative closure criteria at other mining operations and jurisdictions. Utilising the reference site analyses presented earlier and the benchmarking outputs, suitable floristic parameters (or attributes) and preliminary descriptive closure criteria are proposed.

2.2.1 Benchmarking of other operations and jurisdictions

There are many guidelines and frameworks for setting and assessing mine closure objectives (Section 1.2); however the majority of closure criteria are based on processes and qualitative parameters. A review and benchmarking exercise was undertaken to identify best practices in relation to more detailed closure criteria, focussing on areas where the intention was mainly concentrated on re-establishing native vegetation. In addition, the review concentrated on previous practice in the Australian mining industry due to current standards in local and national context.

To extract this information it was necessary to rely on specific and publicly available licences, closure plans and environmental plans. Not all of details by many mining companies are explicit in public documents and there is a reliance on process rather than detailed closure criteria. In many instances there are more generic statements related to outcomes such as the re-establishment of sustainable ecosystems with similarities without specific targets or metrics to achieve such outcomes.

The pattern of increasing expectations on the industry and the studies undertaken internationally are on a similar trajectory towards greater certainty on outcomes. Criteria, where they are defined, tend to be qualitative rather than quantitative criteria. There has been a greater reliance on measurements of particular parameters that are key to re-establishing the native ecosystems. The latter include parameters such as use of local flora species, selection of key, dominant or framework species, selection of species that are known to establish and some quantitative data on species richness, density and cover. Fauna species are less regularly assessed with the exemption of a greater coverage of invertebrate species such as ants and bird species in the early phases. The more detailed best practices concentrate on process, internal outcomes and external outcomes; with clear triggers on adaptive management needs and actions. These will be discussed further for each specific parameter in the Closure Criteria (Section 8).

2.2.2 Key floristic parameters

These floristic parameters (Table 2-7) reflect current local and international industry guidelines (SRG SERA 2017), and as such reflect consistency in current operations and are in many instances comparable to those used internationally (Ruiz-Jaen & Aide 2005; Wortley *et al.* 2013).

Table 2-7 Key Attributes for assisting in alignment from local and international guidelines for rehabilitated and restored ecosystems (extracted from SRG SERA 2017)

SER (2004) Restoration Attributes (^^)	WA EPA (2006) Rehabilitation Criteria	SERA (2017) Restoration Attributes (^^)
1. Structure 3. Functional groups	9. Abundance or density 12. Canopy and keystone species 16. Habitat diversity	Community structure
1. Structure 2. Indigenous species 3. Functional groups	8. Species diversity 10. Genetic diversity 11. Ecosystem diversity **13. Effective weed control 15. Animal diversity	Species composition
Resilient Self-sustaining 5. Function	**6. Soil structure and function 7. Self-sustaining and resilient	Ecosystem function
Landscape integration External threats	**13. Effective weed control 14. Pest and disease control	External exchanges Absence of threats

** criteria repeat over different attributes; ^^ – SER (2004 and SERA (2017 use the terminology attributes rather than parameters.

One of the keys to selecting and refining the selection of attributes for the closure criteria include including key parameters (or attributes as used by some authors) that reflect and support the assessment for outcomes, be easily and consistently sampled by different researchers, are reliable indicators in line with key attributes of ecosystems, have clear and consistent analytical methods available to a wide range of technical and professional skill levels, be appropriate to time frames and be clear to assessors and those reviewing progress.

The Society for Ecological Restoration (SER 2004) recommended nine ecosystem attributes to measure restoration (rehabilitation in the mining industry context):

- similar ecosystem diversity and community structure to those of the reference sites
- the presence of indigenous (native) species
- the presence of functional groups necessary for long-term stability

- the capacity of the physical environment to sustain reproducing populations
- normal functioning
- integration within the landscape
- the elimination of potential threats
- resilience to natural disturbances
- self-sustainability

The SER Primer underpins key ecosystem attributes to formulate goals for restoration (SRG SERA 2017). Recent reviews reveal seemingly infinite numbers of indicators that have been used or could be used to reflect the ecosystem attributes in different areas and ecosystems (Ruiz-Jaen & Aide 2005; Wortley *et al.* 2013). These studies have reflected the dominance of species diversity, abundance, structure and ecological processes as key attributes. The range of possible attributes is summarised in the document by Kragt *et al.* (2019) and includes an extensive range of abiotic, biotic indicators. In Western Australia, recent standards on baseline studies has led to more consistency in approaches (WA EPA 2016a, 2016b).

The initial focus is concentrated on the dominant Eucalypt and *Corymbia* woodlands near the RPA with a view towards following a similar approach for other ecosystems associated with other post-mining conditions (e.g. riparian areas and seasonally wetter sites). The development of the concepts of domains in the pre-mining and post-mining areas has been commenced and as such relies on the underlying information on the baseline environmental values and research associated with understanding these values and how these values could be restored on highly disturbed environments.

In line with the end land use and proposed outcomes at Ranger the emphasis in the revegetation planning and processes relies on an understanding of the constraints and where these can be addressed and minimised to return the local flora and fauna species, the structure and function of the communities and the associated values on a trajectory towards such an end land use. Other mine sites have addressed some of these short-term gaps through the introduction of the following procedures:

- Selection of engineering designs (landforms, soils and drainage) that may facilitate the species and ecosystem functions.
- Selection of alternative species that may be known to prefer specific site conditions.
- Placement of values such as surface soils or logs and hollows in local scattered areas to assist with progression of species re-colonisation.
- Avoidance of some treatments (e.g. avoidance of soils that may introduce competitive native species or weed species that may increase fire risks in the early phases of the rehabilitation of the post mining sites).

In assessing the values there has been a reliance on most recent mine site rehabilitation activities to use indicators such as use of local provenance seed and seedlings, plant species richness, plant cover and plant density. Of the range of indicators these were the most commonly used to evaluate the progress of restoration programs (Ruiz-Jaen and Aide 2005).

The following parameters have been agreed by ERA and the Supervising Scientist Branch (Table 2-8). The development of the parameters and descriptive completion criteria are summarised in MCP Section 8.

Table 2-8: ERA Agreed Objectives, Outcomes and Parameters.

Objective	Outcome	Parameter
Revegetation of the disturbed sites of the Ranger Project Area using local native plant species similar in density and abundance to those existing in adjacent areas of Kakadu National Park, to form an ecosystem the long term viability of which would not require a maintenance regime significantly different from that appropriate to adjacent areas of the park	Revegetate the disturbed sites of the RPA using local native plant species.	Provenance
	Species composition and community structure is similar to adjacent areas of Kakadu NP	Species composition (tree and shrubs) and relative abundance
		Canopy architecture
		Canopy cover index, ground cover index
		Tree distribution**
	Long term, viable ecosystem requiring maintenance similar to adjacent areas of Kakadu NP	Reproduction (flowering and seeding)
		Recruitment / regeneration
		Nutrient cycling
		Fire resilience
		Resilient to wind and drought
		Weed composition and abundance
		Native fauna
		Exotic fauna

**Tree distribution is covered separately in the Cultural Criteria.

2.2.3 Future development of quantitative closure criteria

The proposed qualitative criteria are currently focussed on derivations of the local woodland ecosystems, anticipated to be suitable for the bulk of the final landform and land application areas at Ranger Mine. However, as indicated in the technical review of constraints, there are scenarios predicted that may require additional reference ecosystems to be identified, such as riparian, sedgeland and grassland, or shrubby ecosystems. This will then require the gathering of data from suitable analogue sites, which may take some effort (and time) and is required to inform revegetation activities. Refinement of qualitative and/or quantitative closure criteria and monitoring and assessment methods shall follow the process outlined below.

Following the agreement of the proposed qualitative criteria, ERA shall continue working towards quantitative closure criteria through the following steps:

- Review all available rehabilitation monitoring data from ERA including Trial landform data, previous revegetation trials, and early results from Stage 13 and Pit 1 revegetation activities.
- Access relevant rehabilitation data from other sites, such as the *Eucalyptus tetrodonta* dominated revegetation at Gove and Weipa bauxite mines (over 40 years of knowledge).
- Utilise the State-and-Transition model that has recently been developed (Richards *et al*. 2020 - in draft) to refine the trajectories for key parameters of the revegetation, to identify milestones and thresholds to inform the ERA Adaptive Management Plan.
- Review other trajectory study options as recently developed by Steedman *et al*. (2019) utilising species richness and density datasets to evaluate progress on rehabilitation areas.
- Propose quantitative closure criteria for the target 'close-out' timeframe expressed relative to the appropriate conceptual reference ecosystem.
- Undertake a statistical review and benchmarking exercise on how quantitative closure criteria should be monitored and assessed at Ranger Mine.

Once draft quantitative closure criteria are proposed, these will be reviewed by key stakeholders and key researchers in line with adaptive management of options for progressing the ecological restoration on a trajectory to meet the proposed outcomes. In view of the limitations associated with limited trials on the revegetation areas, it is important the proposed assessment methodologies and studies are developed and refined to enable ongoing testing and adaptive management and strategies for continual improvement (Mattiske & Meek 2020).

In developing these quantitative measures it is important to undertake data gathering which is scientifically rigorous without the complexity that restricts effort and coverage.

As part of the development of potential quantitative closure criteria, there is a need to review former and proposed monitoring methods to enable not only comparisons with reference ecosystem values but also with proposed closure criteria which can vary in their trajectories in the initial phases of rehabilitation. In the context of mine closure there may be leading indicators and lagging indicators. An example of using initial indicators at Alcoa of Australia Ltd bauxite mines illustrates this approach with selected indicators which has enabled remediation and supplementary treatments to be undertaken in a timely manner.

As discussed in Section 2.2.2, there are a range of options for metrics for different parameters over time. The options include, for example, means or medians with standard errors, or a range of data within the bounds of that in the appropriate reference ecosystem, or the use of percentiles within set bounds (e.g. 10% to 90% or 20% to 80%). Consideration of the interaction of post-mining conditions and the selection of appropriate closure criteria have been taken into account initially in the selection of potential descriptive qualitative criteria (MCP Section 8) and as such will require further refinement for the 2021 ERA MCP.

The rate and predicted direction of change in environmental metrics will vary between parameters over time, and will be reflected in the different rehabilitation trajectories. At ERA these components will be addressed as part of the quantitative review; however at this juncture and considering the industry benchmarking exercise, it is expected that the initial planting of seedlings of the framework species will encourage rapid growth and a range of other attributes associated with colonisation and dispersion, and litter accumulation will result. Amongst the lagging indicators will be a range of fauna species that rely on soil development and also values that will take some time to establish. This latter aspect will require further investigations in 2020/2021.

The additional key component in developing suitable quantitative closure criteria is a clear way forward on methodology of assessments, analysis and interpretation of the findings on future rehabilitation areas. The critical aspect of the latter is the need for consistency and coverage of key attributes in a scientifically rigorous approach.

3 REVEGETATION STUDIES AND KNOWLEDGE AT RANGER MINE

Over more than thirty years, a large number of small-scale revegetation trials have been undertaken at Ranger Mine by the CSIRO, ERISS, ERA and other parties in relation to final landform (FLF) morphology, revegetation and ecosystem establishment (Section 3.1). All this research has culminated in an extensive body of applied techniques, designed to give confidence that the revegetation strategy proposed for the closure of the RPA will result in a self-sustaining, long-term ecosystem. These practical techniques are summarised in MCP Section 9.4.6.

3.1 Early revegetation establishment trials at Ranger Mine

A myriad of revegetation trials were undertaken at Ranger Mine between 1982 and 2002 (refer Table 3-1 and Figure 3-1). Almost all of these trials were discontinued at various stages, due to the need by operations for additional waste rock storage areas as mining of the pits progressed. However, these trials enabled important lessons to be learned early and in turn influence subsequent trials. This historical knowledge and experience was used to inform the first Ranger Revegetation Strategy and the establishment of a dedicated waste rock revegetation research facility – the Trial Landform (TLF).

In 2001, Reddell and Zimmermann (2002) completed a comprehensive assessment of 11 earlier waste rock revegetation trials and identified a number of examples of success and failure and related key issues that are highly relevant to ERA's revegetation strategy.

In more recent years, investigative studies have been undertaken on local seed provenance for revegetation, and species composition and community structure. The outcomes of these studies are described in the following sections.

Table 3-1: Small-scale revegetation trials conducted on the RPA (1982 – 2002)

Project	Location	Date
First revegetation – germination trials	Waste rock piles	1982
Irrigation using RP2 water to 35 hectares of mature savanna woodland, along with fire exclusion	Ranger Mine lease	1984-1995
Fire trial	Waste rock piles	1986
1:5 slope erosion trial	Waste rock piles	1986-1987
Constructed wetlands experiments and aquatic plant transplantation	North-west seepage collector	1987-1988
Slope erosion trial	Waste rock piles	1988-1991
Wetland filter trials using RP4 water directed through 3 hectares of Djalkmarra Creek catchment	Djalkmarra Creek catchment	1988-1991
Topsoil spread. Hydroseeded (grass and fertiliser ± eucalypt seed). *Pandanus basedowii* planted	Waste rock piles	1988-1995
Topsoil trials ± fungi	Waste rock dump	1989

Project	Location	Date
Revegetation trials and rainfall simulation	Waste rock piles	1990-1993
Direct seeding via tractor spread of 3 ha with pasture grasses	Northern waste rock dump	1991-1992
Hydromulching, tree and grass seed spreading, and aquatic plant transplantation (*Eleocharis, Nymphaea* and *Azolla*)	RP1 wetland filter	1991-1992
Tubestocks ± inoculation. Various seed mixes, grass, aggressive and non-aggressive acacias. Planting on angle of repose batter west of plots	Ecological islands	1992
Topsoil trial	Waste rock piles	1992
Topsoil spread	RP5	1992
Application of hydromulch and grass seed to batter slopes facing Pit 1	Pit 1	1992
Tubestock planting, seedling and fungi trials	Northern waste rock dump	1992
Native seed and tubestock planting at tailings seepage sumps	North-western, north-eastern and southern seepage collectors	1992-1993
Tubestock and native tree seedling planting	VLGS (stockpile, north-west of the TSF)	1992-1994
Tubestock planting and fungi and varied density of nitrogen-fixing acacias. Inoculation of different seed mixes	RP4 irrigation	1992-1994
Seeded (grass and fertiliser with broadcaster)	Northern waste rock dump	1993
Log shelter/baits, termite baiting, pitfall trapping and casual soil fauna collecting	Northern waste rock dump	1993-1994
Native tubestock	VLG (west of Pit 1)	1993-1995
Native tubestock planted (grown by ERA and Djabulukgu Association)	Southern waste rock dump	1993-1997
Rhizobia trial	Waste rock piles	1994-1995
Effect of seed imbibition mulch, fertiliser *Scleroderma* and eucalypt applications rates	Southern waste rock dump	1994-1995
Angle of repose and 1:3 batter slopes. Randomised block hydromulched seed and *Pisolithus ectomycorrhizal* fungi	RP5	1994-1995
Establishment and growth on waste rock and magnesite to determine rate of self-thinning in high density eucalypt and non-aggressive acacias and slow release fertiliser	RP5	1994-1995
Effect of mulch type on germination and early growth	Waste rock piles	1994-1995
Native tubestock planting	Waste rock piles	1994-1996

Project	Location	Date
RP1 wetland filter expansion and aquatic plant transplanting (*Nymphaea* and *Eleocharis*)	RP1 wetland filter	1995
Effect of mycorrhizal associations on survival and growth of *Eucalyptus miniata* seedlings.	RP5	1995
Direct seedling fertiliser and tubestock planting	Sleepy Cod Farm Dam walls	1995-1996
Transplanting native tree root section trials	Southern waste rock dump	1996
Irrigation with RP4 water, introduced grasses (*Chloris gayana*), tubestock and seed mix trials	Waste rock dump	1996
Large-scale planting (seed and tubestock) composition, density, irrigation, mulch, fungi, fertiliser	Waste rock and Retention Pond	1996-1997
Hydromulch and native grass trials ± fertiliser	Northern waste rock dump	1996-1997
Elevated wetland trials, tubestock, seed and herb transplanting	Southern waste rock dump	1997
Measure indicators of rehabilitation success on the RPA. Fauna surveys and landscape function analysis	Ranger Mine lease	1997
Direct seeding	Old light industrial area road	1997-1998
Hydromulch with native grass seed and fertiliser applied to 3 kilometres of table drain	Main access road	1997-1998
Direct seeding, tubestock and fertiliser application	Northern waste rock dump	1997-1998
Hydromulch with native grass seed and fertiliser application	TSF waste rock dump	1997-1998
Direct seedling, tubestock and fertiliser application	Southern waste rock dump	1997-1998
Direct seeding and tubestock planting following deep ripping	Borrow pit north-west of Pit 3	1998
Seed (*Grevillea* spp.) under erosion control matting	RP5	n.d.
Removal and remediation/rehabilitation of road infrastructure. Tubestock and direct seeding trials of native woodland species on freshly cultivated waste rock	Various roads, tracks and former low-grade ore stockpiles	1998 - 1999
Grass direct seeding trials with and without fertiliser	Borrow pits	1999 - 2002

ERA

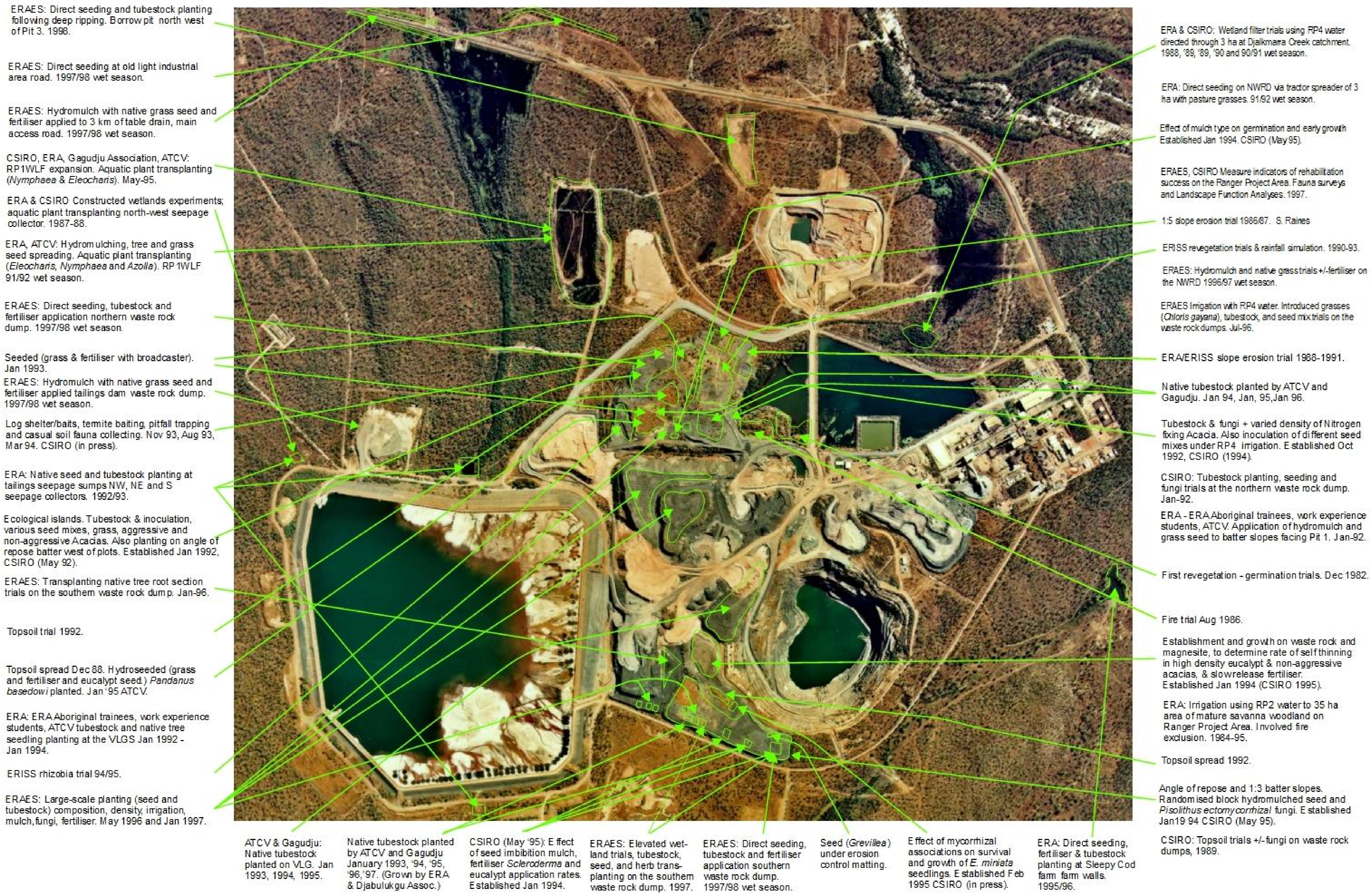


Figure 3-1: Revegetation conducted on Ranger Mine (1982 – 1998)

3.2 The Trial Landform: An ongoing (11-year long), large-scale field test of the Revegetation Strategy

The 8 ha TLF, situated near the north-western corner of the TSF (Figure 3-2), was constructed in 2008/2009 to allow for testing of landform design, substrate types, and revegetation strategies (Daws *et al.* 2009). It also has provided the opportunity to investigate and implement adaptive-management during ecosystem establishment (Humphrey 2013). An extensive monitoring system was installed to assess the soil water holding capacity, runoff and infiltration of the landform (Daws *et al*. 2008, Shao 2015) as well as the revegetation performance.

The TLF has enabled the Ranger Revegetation Strategy to be tested and refined. It has also informed many of the physical and biophysical features of the FLF design, including but not limited to: its waste rock construction, erosion, bedload, stability, water management, radiological aspects, revegetation and ecosystem development.

The following sections provide an overview of the construction and purpose of studies to date on the TLF.

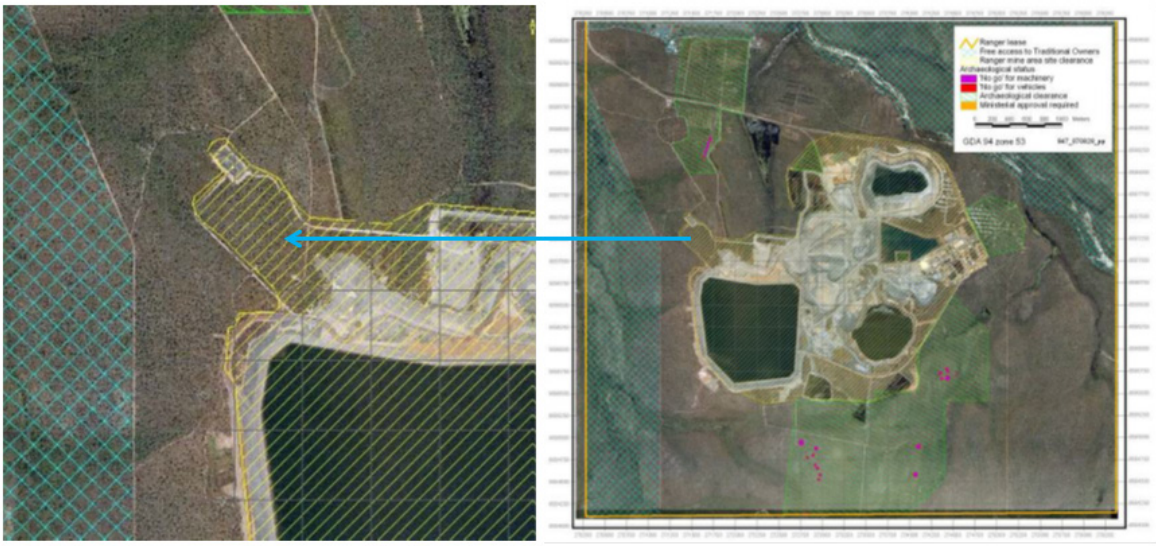


Figure 3-2: Location of the trial landform, north-west of the TSF (Pugh *et al.*, 2008)

3.2.1 Design and construction

The TLF was designed based on studies undertaken by ERA and ERISS on analogue sites and previous revegetation work conducted at Ranger Mine. It stands four to seven metres above the original natural ground surface and was constructed using 800,000 tonnes of primary and weathered waste rock and laterite material. The design has allowed testing of the performance of different types of surface substrates, different depths of mixed materials over the waste rock only layer, different planting methods and different irrigation regimes (Figure 3-3; adapted from Pugh *et al* 2008).

The TLF 1A was built by first constructing a base layer approximately 2 m thick, by tip-head dumping, and then placing another layer 2 m thick over it, by paddock dumping. As a result of this construction method, a sub-surface consolidated horizon was created by the activity of the dozers and dump trucks on the surface of the TLF base layer, underneath the final paddock dumped layer. Construction records show that the surface of the base layer of the TLF (prior to the commencement of paddock dumping) had a high proportion of visible fines compared to underlying material.

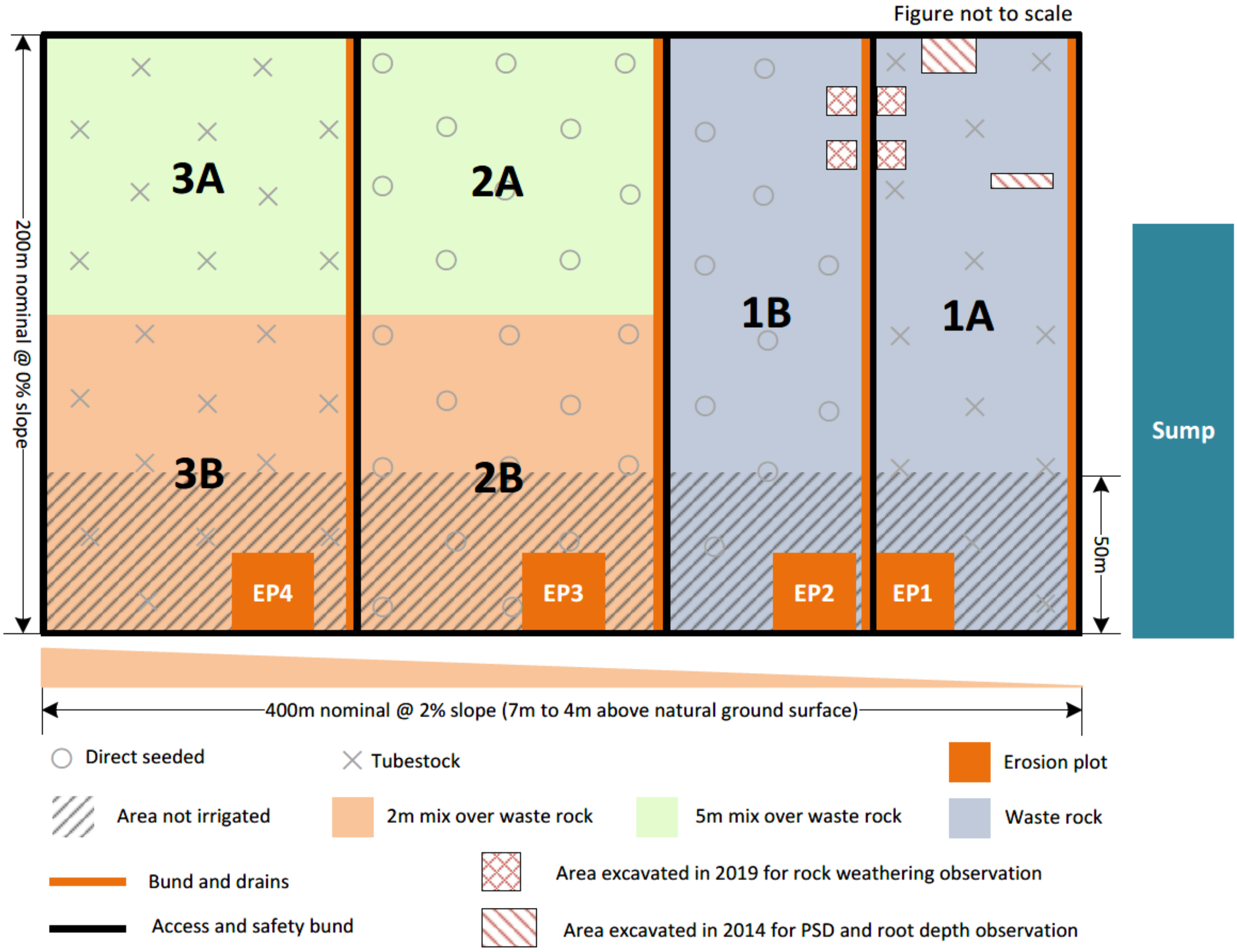


Figure 3-3: Trial landform – treatment design and associated infrastructure

The three main rock types in Ranger waste rock stockpiles are primary, weathered and laterite materials, all of which were used in the construction of the TLF. Primary material consists of unweathered host rock, which primarily consists of altered quartz-feldspar schists and to a lesser extent cherts and carbonaceous materials. Weathered material consists of friable rock (usually quartz-feldspar schist) with altered mineral assemblages, but generally still low in clay content. Laterite is a near surface, highly weathered and sometimes reconsolidated material that is generally high in iron and aluminium clays (ERA 2018). Photos of the 1s primary material and weathered rock used for construction of the TLF are shown in Figure 3-4.

The surface substrates trialled on the TLF were: waste rock only; and waste rock blended with 30 percent volume/volume of laterite rock. To facilitate treatments, the trial landform was divided into several areas according to treatment (Daws & Poole 2010). The Area 1A and 1B of the TLF were constructed with the waste rock only. Areas 2(2m) and 3(2m) were constructed as a two-metre thick layer of laterite /waste rock mix over a base of 1s rock 3 to 5 metres thick. Areas 2(5m) and 3(5m) were constructed as a five-metre thick layer of laterite/waste rock mix over a 1's rock base 0 to 2 metres thick. The Ranger FLF surface layer will be primarily constructed with primary and weathered waste rock without purposely mixing in laterite. This design and construction is similar to the waste-rock only section of the TLF (i.e. section 1A), presented in Figure 3-5.

Bulk density of the substrate layer of the TLF is estimated at about 2.0 t/m^3, with a specific gravity of solids of 2.65 t/m^3 (Stephen Pevely, Senior Resource Geologist, ERA, *pers. comm.* Oct 2017). This equates to a void space of about 25% (void volume/ total volume). In its natural state this void space will be filled partially by air and water.

The TLF was constructed with a 2% slope and was ripped at 2 metre intervals down to approximately 0.5 m deep.

Vegetation establishment commenced in March 2009 and an area 50-metres wide on the front, north-eastern side of the TLF was left unirrigated; this is further described in Section 3.2.3 below.

(source: Daws & Poole 2010)

Figure 3-4: Rock types used to construct the trial landform

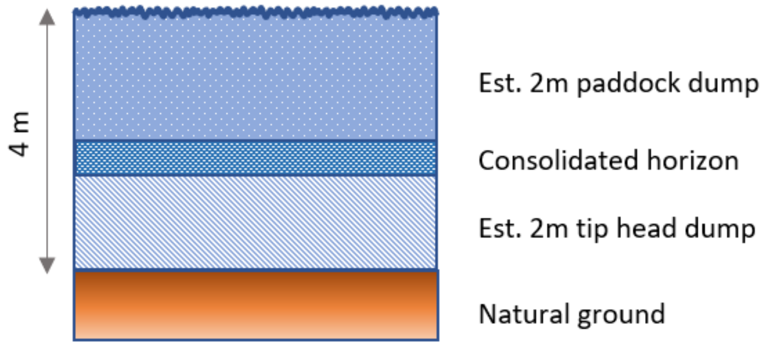


Figure 3-5: Profile of the waste-rock only section 1A of the TLF

3.2.2 Instrumentation

The landform design incorporates runoff and catchment management features, and monitoring systems to provide water quality data to inform decision-making on future water management strategies. These include:

- 66 soil moisture probes.
- A weather station.
- Four erosion plots (Supervising Scientist, 2010), featuring:
  - A tipping bucket rain gauge.
  - A primary shaft encoder with a secondary pressure transducer to measure stage height.
  - A turbidity probe.
  - Electrical conductivity probes located at the inlet to the stilling basin and at the entry to the flume to provide an inferred measure of the concentration of dissolved salts in runoff.
  - An automatic pump sampler to collect event-based water samples.
  - A data logger with mobile phone telemetry connection and a rectangular broad-crested flume to accurately determine discharge from the plots.

3.2.3 Vegetation establishment trials

A range of trials have been undertaken on the TLF (Table 3-2). Overstorey (OS) and midstorey (MS) species were initially introduced in 2009 in both the waste rock and laterite mix areas of the TLF; tubestock planting was conducted in March and direct seeding occurred in July. This resulted in the entire TLF being revegetated except for a 40 - 50 m strip along the northern edge of the direct seeded areas, which was not seeded since it was outside the irrigated zone. This area was direct seeded when rainfall commenced in December 2009 (Daws & Poole 2010). In January 2010, additional tubestock was planted in the tubestock areas to fill gaps left by an initial high mortality (Daws & Gellert 2011). In January 2011, tubestock was planted in the direct seeding areas to increase plant density and correct for the skewed species composition due to the low success rate of some of the species (Gellert 2012). In February 2020, additional tubestock were planted to increase OS and MS diversity, to trial species that hadn't been grown in waste rock before, and to trial 'secondary' introductions for species that failed to establish during the initial revegetation.

There have been multiple attempts to establish understorey (US) species on the TLF. Grass seeds were sown in January 2011 on the tubestock areas and in November 2012 on the waste rock tubestock section, both times without fertiliser or irrigation. In 2018, a comprehensive research project was undertaken to investigate optimal protocols for establishing native US grass and legume species on waste rock (Parry 2018). Both direct seeding and tubestock planting were trialled on sections 1A and 1B of the TLF, which by then had considerably different stem densities and canopy covers. In addition, five different amelioration treatments to the waste rock were investigated with the direct seeding trials. A well-watered shade house trial was also conducted in 2018 investigating the same waste rock amelioration treatments. In January 2019, the US plants left over from the shade house trial were planted in 'islands' on the waste rock section. Lastly, in February 2020, a mixture of grasses, legumes, shrubs and herbs were planted and sown to increase US diversity and to trial 'secondary' introduction methods.

Controlled burns were performed in May 2016 (Wright 2019a) and June 2019 (Wright 2019b) on the laterite mix areas of the TLF as a means of weed management and to measure the resilience of the established vegetation.

Table 3-2: Vegetation establishment activities conducted on the Ranger Mine TLF, 2009 – 2020

Month/Year	Action	Details	Reference
March 2009	Tubestock planted on the TLF	1473 tubestock planted in section 1A, 3029 planted in section 3 – each with 21g slow release fertiliser tablet	2
July 2009	Direct seeding of TLF (irrigated sections)	Seed mixes, made up of 31 species, sown at a rate of 3 kg ha^{-1} in sections 1B and 2	3
December 2009	Direct seeding of TLF (unirrigated sections) Fertiliser application	Direct seeding of the northern edge in sections 1B and 2, using the same sowing rate and species mix as the previous areas 50 kg ha^{-1} of Osmocote Plus to whole landform – applied at the base of tubestock and broadcasted in direct seeded areas	4
January 2010	Infill tubestock planted	699 tubestock planted in section 1A, 1317 planted in section 3 – each with 21g slow release fertiliser tablet	3
November 2010	Fertiliser application	50 kg ha^{-1} of Osmocote Plus to whole landform – applied at the base of tubestock and broadcasted in direct seeded areas	3
January 2011	Infill tubestock planted	1449 tubestock planted in section 1B, 2432 planted in section 2 – each with 21g slow release fertiliser tablet	5
January 2011	Understorey trials	Five grass species were sown in section 1A and 3	6
January 2012	*Xanthostemon* tubestock planted	Approximately 300 planted in the track between sections 1A and 1B; 75 planted in section 3	7
November 2012	Understorey trials Fertiliser application	Seven grass species were sown in section 1A Small handful of Osmocote applied to each of the Jan-2011 infill planted tubestock. Smaller amount applied to direct-seeding plants on an ad-hoc basis	8, 9 6
May 2016	Weed management	Cool burn of the laterite mix sections (2 and 3)	10
April 2018	Understorey direct seeding trial	Five understorey species were sown in sections 1A and 1B with six WR amelioration treatments	11

[2] Daws and Gellert (2010) Initial revegetation monitoring on the trial landform
[3] Daws and Poole (2010) Construction, Revegetation and Instrumentation of the Ranger Uranium Mine Trial Landform: Initial Outcomes
[4] Daws and Gellert (2011) Ongoing revegetation monitoring on the trial landform
[5] Gellert (2012a) Ongoing revegetation monitoring on the Trial Landform 2011
[6] Gellert (2012b) Establishment trials for five native grasses on the Ranger Trial Landform
[7] Gellert (2013) Ongoing revegetation monitoring on the Trial Landform 2012
[8] Gellert (2013) Ongoing revegetation monitoring on the Trial Landform 2012
[9] Gellert (2014) Ongoing revegetation monitoring on the Trial Landform 2013
[10] Wright (2019a) Effects of the 2016 prescribed fire on revegetation at the trial landform (2016 and 2018 surveys)
[11] Parry (2018) Treatments to improve native understorey establishment in mine waste rock in northern Australia

June 2018	Understorey tubestock trial	Five understorey species were planted in sections 1A and 1B	11
January 2019	Understorey planting in 'islands'	Nine understorey species that were grown in 2018 nursery trials were planted in 'islands' on sections 1A and 1B – some with litter	NA
June 2019	Weed management	Cool burn of the laterite mix sections (2 and 3)	12
February 2020	'Secondary' introductions	Eighteen species tubestock planted (10x US and 8x MS/OS), and seven understorey species seeded in patches with and without added mulch (21 species total, mostly 1A and 1B)	13
February 2020	Understorey direct seeding trial	Twelve understorey species were sown in section 1A in plots with and without naturally occurring organic matter	14

3.2.4 Ecosystem monitoring programs on the TLF

The TLF has been continually monitored over the last decade to assess revegetation performance and ecosystem development.

In September 2009, five 15 x 15 m Permanent Monitoring Plots (PMPs) were established in each of the different sections of the TLF; a further 15 PMPs were established in February 2011 after infill planting was performed in 1B and 2 (Figure 3-6 and Table 3-3). The OS and MS plants inside the PMPs have been monitored annually (excluding 2017) for survival, growth, and density. In addition to the PMPs monitoring, two large-scale surveys measuring every single OS and MS plant on the TLF have been conducted, once in 2009 and again ten years later in 2019.

From 2010 to 2014, TLF monitoring also included Landscape Function Analysis (LFA) to measure stability, infiltration and nutrient cycling.

Starting in September 2018, regular walk-throughs have been performed in every section of the TLF to opportunistically capture and/or monitor patterns and changes. Some of the observations include whether established plants are flowering, fruiting and recruiting, and whether new species have been able to naturally colonise the TLF from external sources.

[12] Wright (2019b) Technical Memo: TLF (laterite mix areas) weed control burn – June 2019

[13] Trial Landform Research and Monitoring Plan 2020 – 2026 (in draft)

[14] Parry (2020) Project plan for 'secondary introduction' understorey direct seeding trials on TLF – in draft

Figure 3-6: Permanent Monitoring Plot Locations on the Ranger Mine TLF

Table 3-3: TLF Permanent Monitoring Plot details

Plots	Substrate Type	Establishment Method
0 – 4	Waste rock only	Tubestock
5 – 9	Laterite mix (5m depth)	Tubestock
10 – 14	Laterite mix (2m depth)	Tubestock
15 – 19	Waste rock only	Direct seeding
20 - 24	Laterite mix (2m depth)	Direct seeding
25 – 29	Laterite mix (5m depth)	Direct seeding
30 – 34	Waste rock only	Tubestock & Direct seeding
35 – 39	Laterite mix (2m depth)	Tubestock & Direct seeding
40 - 44	Laterite mix (5m depth)	Tubestock & Direct seeding

3.3 Species establishment research program

The ERA revegetation strategy is to initially establish framework overstorey species along with a subset of important and predictable midstorey and understorey species (Section 1). Once these species have established, they will control much of a site's nutrient and water resources, and will provide many of the core habitat values for other plants and animals to colonise (Reddell & Hopkins 1994). Based this approach, the species establishment research program (SERP) has been developed to systematically work through all of the potential revegetation species and identify the best way to establish them in the rehabilitation at the Ranger Mine.

The SERP will undertake a series of progressive trials to determine the most efficient and effective establishment method for each species or for an indicative species for a group of related or similar species. Priority will be placed on framework species that are required for initial introductions as this will result in the majority of species and stems per hectare in the revegetation program. Other species, particularly understorey species, will be progressively tested in small trials (e.g. pot trials or small-scale field trials) due to very limited seed.

The SERP is continuously working to increase the number of species included in the revegetation implementation program (either as initial or secondary introductions), through improved understanding of practical aspects such as seed collection, storage and usage strategies, propagation tactics, planting and irrigation methods, and species-specific ecological characteristics in terms of substrate, water availability and competition.

3.3.1 The SERP species list

Plant species composition and relative abundance based on appropriate reference sites (Section 2.1.3) was used to develop a revegetation species list with relative density for the revegetation of the TLF in 2007 by ERA in collaboration with ERISS and was provided to GAC for consultation in 2014 (Lu 2014). In 2015, the Mirarr developed a list of culturally important flora based on various criteria that pertain to an end use continuum, including but not limited to whether the plant is used as a cultural resource (e.g. for food, medicinal, aesthetic, material culture and/or ritual purposes), provides faunal linkages, and promotes biodiversity (Garde 2015).

In March 2016, the flora and fauna closure criteria technical working group (TWG) reached a consensus on a Ranger Mine revegetation tree and shrub species list (MCP Section 9.4.6.1) This revegetation species list was developed based on:

- previous analogue vegetation studies in undisturbed RPA and surrounding areas by ERISS and ERA (125 studied analogue sites, including 10 sites from Kakadu NP with a land surface similar to the Ranger Mine final landform) (Section 2.1.3, Figure 2-4);
- culturally-important plant species, as identified by the Mirarr Traditional Owners in Garde (2015), and
- learnings from progressive revegetation activities and in particular the learnings from the TLF.

The current SERP species list (Table 3-4) comprises 121 species, including 17 overstorey tree species, 61 midstorey tree and shrub species, and 43 understorey species. All species are initially assessed as framework or 'other', and likely suited to an initial or secondary introduction strategy. The list is based on the 2016 agreed revegetation tree and shrub list and expanded with the addition of understorey species based on early surveys by Brennan (2005) and further modified after consultation with Peter Christophersen (*pers comm.*, 2019) and Dr Sean Bellairs (Lu *et al.* 2017; *pers comm.* 2019).

The species included in this list will continue to be refined as outcomes from ongoing reference site survey and data analysis (e.g. Mattiske & Meek 2020 – *in draft*), revegetation trials (e.g. TLF, Stage 13 and Pit 1), risk assessments and further stakeholder consultations are completed (including appropriate formal review by stakeholders).

3.3.2 Culturally significant plant species

A number of species have been included in the agreed revegetation list following cultural consultation with the Mirarr Traditional Owner group (Garde 2015). While fifteen species identified by Garde (2015) do not occur in any of the historically surveyed reference sites (e.g. Georgetown, Brennan, OSS surveys; Section 2.1.2.1), their cultural significance warrants inclusion in the revegetation list. An additional eight species are on the list that were identified as culturally important plant species by the Mirarr Traditional Owners, however these are out of scope or of taxonomic uncertainty. In this context, it is acknowledged by the Mirarr that it may not be possible to propagate and establish all species. Nevertheless, the intention is to plant as many species identified by the Mirarr on the final landform as practicable, to address cultural and other values such as aesthetics.

Table 3-4: The SERP revegetation species listed with framework status and potential introduction strategy

Species	Framework / Other	Initial / Secondary Establishment
Overstorey trees		
Corymbia bleeseri	Framework	Initial
Corymbia chartacea (setosa)	Framework	Initial
Corymbia dichromophloia	Other	Initial
Corymbia disjuncta (confertiflora)	Framework	Initial
Corymbia dunlopiana (setosa)	Other	Initial
Corymbia foelscheana	Framework	Initial
Corymbia latifolia	Framework	Initial
Corymbia polycarpa	Other	Initial
Corymbia polysciada	Other	Initial
Corymbia porrecta	Framework	Initial
Corymbia ptychocarpa	Other	Initial
Erythrophleum chlorostachys	Framework	Initial
Eucalyptus miniata	Framework	Initial
Eucalyptus phoenicea	Framework	Initial
Eucalyptus tectifica	Framework	Initial
Eucalyptus tetrodonta	Framework	Initial
Eucalyptus tintinnans	Other	Initial
Understorey		

Species	Framework / Other	Initial / Secondary Establishment
Midstorey trees and shrubs		
Acacia aulacocarpus	Other	Secondary
Acacia difficilis	Other	Initial
Acacia dimidiata	Other	Initial
Acacia hemignosta	Other	Initial
Acacia lamprocarpa	Other	Secondary
Acacia latescens	Framework	Initial
Acacia mimula	Framework	Initial
Acacia oncinocarpa	NA	
Allosyncarpia ternata	Other	Secondary
Alphitonia excelsa	Other	Initial
Antidesma ghesaembilla	Other	Secondary
Asteromyrtus symphyocarpa	Other	Secondary
Banksia dentata	Other	Secondary
Brachychiton diversifolius	Other	Initial
Brachychiton megaphyllus (paradoxus)	Other	Initial
Buchanania obovata	Framework	Initial
Calytrix achaeta	Other	Secondary
Calytrix exstipulata	Other	Secondary

Species	Framework / Other	Initial / Secondary Establishment
Overstorey trees		
Acacia gonocarpa	Framework	Initial
Alloteropsis semialata	Framework	Initial
Ampelocissus acetosa	Other	Initial
Aristida holathera	Other	Secondary
Aristida inaequiglumis	Other	Secondary
Chrysopogon fallax	Framework	Initial
Crotalaria brevis	Other	Secondary
Cymbopogon refractus	Other	Secondary
Ectrosia leporina	Other	Secondary
Eragrostis rigidiuscula	Other	Secondary
Eragrostis schultzii	Other	Secondary
Eriachne armittii	Other	Initial
Eriachne avenacea	Other	Secondary
Eriachne basedowii	Other	Secondary
Eriachne obtusa	Other	Initial
Eriachne schultziana	Other	Secondary
Eriachne sulcata	Other	Secondary
Eriachne triseta	Other	Secondary
Ficus aculeata (opposita)	Other	Initial
Fimbristylis caloptera	Other	Secondary

Species	Framework / Other	Initial / Secondary Establishment
Midstorey trees and shrubs		
Clerodendrum floribundum	Other	Secondary
Cochlospermum fraseri	Other	Initial
Coelospermum reticulatum	Other	Initial
Dodonaea hispidula	Other	Secondary
Elaeocarpus arnhemicus	Other	Secondary
Ficus racemosa	Other	Initial
Gardenia fucata	Other	Initial
Gardenia megasperma	Other	Initial
Grevillea decurrens	Other	Initial
Grevillea dryandri	Other	Initial
Grevillea goodii	Other	Secondary
Grevillea pteridifolia	Other	Initial
Hakea arborescens	Other	Initial
Hibbertia dealbata	Other	Secondary
Jacksonia dilatata	Other	Secondary
Livistona humilis	Framework	Initial
Livistona inermis	Framework	Initial
Lophostemon lactifluus	Other	Initial
Melaleuca argentea	Other	Initial
Melaleuca cajuputi	Other	Initial

Species	Framework / Other	Initial / Secondary Establishment
Overstorey trees		
Fimbristylis sp.	Other	Secondary
Flemingia parviflora	Other	Secondary
Galactia megalophylla	Other	Secondary
Galactia tenuiflora	Framework	Secondary
Haemodorum coccineum	Other	Initial
Heteropogon triticeus	Framework	Initial
Indigofera saxicola	Framework	Secondary
Marsdenia sp.	Other	Initial
Mnesithea formosa	Other	Secondary
Panicum mindanaense	Other	Secondary
Schizachyrium fragile	Framework	Secondary
Sehima nervosum	Other	Secondary
Senna leptoclada	Other	Secondary
Sorghum intrans	Other	Secondary
Tephrosia nematophylla	Other	Secondary
Tephrosia polyzyga	Other	Secondary
Tephrosia remotiflora	Other	Secondary
Tephrosia reticulata	Other	Secondary

Species	Framework / Other	Initial / Secondary Establishment
Midstorey trees and shrubs		
Melaleuca dealbata	Other	Initial
Melaleuca leucadendra	Other	Initial
Melaleuca nervosa	Other	Initial
Melaleuca viridiflora	Framework	Initial
Owenia vernicosa	Other	Initial
Pandanus spiralis	Framework	Initial
Persoonia falcata	Other	Secondary
Petalostigma pubescens	Other	Initial
Petalostigma quadriloculare	Framework	Initial
Planchonia careya	Framework	Initial
Stenocarpus acacioides	Other	Initial
Sterculia quadrifida	Other	Secondary
Syzygium eucalyptoides subsp. bleeseri	Other	Initial
Syzygium eucalyptoides subsp. eucalyptoides	Other	Initial
Syzygium suborbiculare	Framework	Initial
Terminalia carpentariae	Framework	Initial
Terminalia ferdinandiana	Framework	Initial
Terminalia pterocarya (canescens)	Other	Initial

Species	Framework / Other	Initial / Secondary Establishment
Overstorey trees		
Thaumastochloa major	Other	Secondary
Themeda triandra	Other	Secondary
Uraria lagopodioides	Other	Secondary
Vigna lanceolata	Other	Secondary
Vigna vexillata	Other	Secondary

Species	Framework / Other	Initial / Secondary Establishment
Midstorey trees and shrubs		
Verticordia cunninghamii	Other	Initial
Vitex glabrata	Other	Secondary
Wrightia saligna	Other	Initial
Xanthostemon eucalyptoides	Other	Secondary
Xanthostemon paradoxus	Framework	Secondary

3.3.3 Trial establishment methods

Compared to most surface mining operations where topsoil return followed by broadcasting of seed during rainy seasons is standard, a non-typical range of establishment options is available in Ranger Mine rehabilitation. A few key options are described below, and additional revegetation methods or tactics for investigation are included within the description of each particular trial, e.g. Stage 13.1 (Section 3.4).

3.3.3.1 Topsoil return and direct seeding

Vegetation is reintroduced to most strip-mines in the wet-dry tropics by both transport of propagules in fresh topsoil and by direct seeding, using a range of methods (from hand broadcasting to tractor mounted seeders to aerial sowing). Occasionally 'enrichment' planting of nursery-grown stock is used to increase the density of important framework species. The success of direct seeding at these strip-mines can be variable, but in general, with good topsoil handling techniques (minimising weed presence in the transported seed bank) and the use of an appropriate seed mix dominated by framework species, good early establishment results have been obtained.

In contrast, on some hard-rock mines direct seeding has been more problematic and unreliable for establishing framework species (Reddell & Zimmermann 2002). Reddell and Hopkins (1994) found that tubestock planting was more successful than direct seeding, and follow-up trials confirmed that the reliability and predictability of vegetation establishment was very low with direct seeding (Reddell & Spain 1995, Gordon *et al* 1995), likely due to the extreme and variable climatic condition on the waste rock surface. Amelioration using mulch treatments were also unsuccessful and results suggest that the interaction between high ambient temperature and fluctuating moisture levels were probably critical factors affecting the success of vegetation establishment from seed. Another limitation with direct seeding is the amount of seed required to establish vegetation at appropriate densities. Considering establishment from seed in the field is often very low (<10 % reported in Merritt & Dixon 2011), a significantly greater quantity of seed is needed for direct seeding as compared to tubestock planting. The revegetation of the Ranger Mine is limited to seed from local provenance, therefore commercial supply of seed is extremely limited.

Although experience shows that direct seeding is not suitable for initial establishment of framework species, it is still an option in some situations (e.g. later establishments with the substrate conditions have improved) due to its:

- potential high cost effectiveness, and
- operational simplicity for 'broad scale' application.

Investigations are underway and shall continue into the environmental conditions and species best suited to this method of establishment.

3.3.3.2 Establishment from tubestock

Based on experience cited above, the Ranger revegetation has (since e.g. Reddell & Meek 2004) focussed on establishment via tubestock. Based on current technology tubestock planting will:

- significantly reduce the risk of planting failure associated with erratic rainfall and extreme temperatures
- accelerate the speed of vegetation development
- overcome the poor predictability of establishing a final revegetated landform from direct seeding techniques

This strategy has proven to be the most cost-effective method for the initial establishment of framework species at the Ranger Mine and is reasonable given the constraint imposed by greatly limited seed availability within Kakadu NP. However, where reliable and predictable direct seeding success can be achieved for some species, such as Pandanus and Kapok *(Cochlospermum* spp.*)*, this method will be used.

Whilst tubestock planting has proven very successful for a range of overstorey and midstorey species, a number of taxa have failed to establish using this method and many remain untested.

3.3.3.3 Litter islands

One opportunity for increasing the diversity of species able to colonise the waste rock final landform would be the establishment of fresh litter islands which would provide a number of valuable elements:

- act as a seed source for growth and further dispersal of a range of (particularly understorey) species (as long as the collection method ensured some of the surface 'soil', including much of the seed store, was obtained)
- introduce an array of microbes (especially mycorrhizae and rhizobia species) present in surface soils and litter of natural eucalypt-dominated woodlands that, by definition, will likely be suited to the native species being established in the waste rock.
- act as a mulch (by reducing surface temperatures and reflectance, and increasing surface soil moisture) and provide small 'micro-niches' where seeds or tubestock of plants that struggled to establish on bare waste rock are able to establish.
- include organic material that could kick start decomposition, support soil microbes and accelerate the soil development process.
- act as a source of future seed for further spread into the rehabilitation area.

This method might be an opportunity early in the initial establishment of revegetation, but it most likely has greatest potential to significantly assist with increasing diversity in the future, underneath the existing canopy of semi-developed overstorey framework trees.

A number of considerations must be made prior to this method being implemented at scale:

- timing (seasonality) of litter collection will highly influence the makeup of the seed store (particularly for annual species) and perhaps the makeup of the microbial population being transplanted with the litter.
- size of islands should be large enough to ameliorate the harsh impacts of the waste rock surface temperature, reflectance and so on, yet small enough to be able to be placed in and around established trees and shrubs.
- the thickness of the litter being applied must not be so thick that it will create a barrier for seedling emergence (e.g. as discussed by Parry 2018).
- suitable material may be limited (sources will be limited to natural sites on the RPA with no weeds) and so judicious use is advised. The number and size of islands must be carefully decided.
- the methods of litter collection and island 'construction' would need to be further developed to suit any large-scale rollout of the method.

Commencing in 2018, a small litter island trial at Jabiluka revegetation site has already shown potential in terms of introduction of target species but also non-target native species (deemed too problematic or non-dominant to warrant active introduction), with an early emergence of over 12 species observed including *Livistona*, *Grevillea*, *Phyllanthus* and *Spermacocce* species.

A series of investigations into this method will continue given that it is showing such potential to increase biodiversity, particularly of the understorey.

3.3.3.4 Passive or voluntary establishment

The Ranger Mine revegetation strategy includes deferring the introduction of competitive or 'sensitive' species (Section 1) until conditions improve. Thus, it is anticipated that understorey species richness will be low for a number of years after initial revegetation. ERA is committed to ensuring that target species composition and densities are achieved, and will develop and implement and, where required, innovative methods to actively ensure they establish on the final landform.

However, the potential role of 'passive' introductions of some of these species should not be overlooked, as this may enable resources to be focussed on the more 'recalcitrant' species requiring active introduction. A common experience in mining revegetation is the 'passive' establishment of what are termed 'volunteer' species, usually through dispersal by insects, animals and wind. These species often include grasses and fruiting species such as figs.

This was demonstrated at Pine Creek mine rehabilitation (on waste rock) where no understorey grasses or herbaceous species were actively introduced in the 1988-1996 seed mixes (Tony Scherer, *pers comm* May 2019) and yet the mature revegetation includes a 'some representative understorey' and 'limited evidence of invasive plants' (Dixon *et al.* 2019).

Observations from previous revegetation on waste rock at the Ranger Mine and more recently on the TLF have recorded a variety of native species establishing that were not actively introduced as part of the trial, these include *Ficus racemosa*, *Alstonia actonophylla*, *Eragrostis cumingii*, *Marsdenia* sp., and more that have yet to be identified.

The Species Establishment Research Program will continue to systematically review and improve the successful introduction of these species in revegetation at the Ranger Mine.

3.4 Early final landform trials (Stage 13.1)

As part of the SERP, a series of opportunistic, small-scale tubestock trials are currently being conducted at Stage 13.1 as a precursor to the large-scale Pit 1 revegetation trials (discussed in Section 6.1). For more information regarding the Stage 13.1 landform characteristics and trial layout see MCP Section 9.

The overall objective of the Stage 13.1 trial is to investigate different potting and planting techniques with the aim of improving tubestock survival during the 6 – 12 months after planting. However, this study will also provide an opportunity to:

- propagate and plant tubestock during different times of the year
- fine tune nursery propagation methods, such as germination rates, required growing times, irrigation requirements etc.
- improve planting methodology, trial new planting equipment, and collect information on ergonomics and HSE considerations
- obtain improved data on predicted species performance and adjust planting strategy (species, density, locations) accordingly
- obtain baseline performance data for species that have not been grown on FLF media previously
- inform future trials for Pit 1 and scaling up for operational planning for Final Landform (2023-2025)

The study consists of two distinct trials: wet season trials to investigate seven different potting and planting methods (treatment descriptions and rationale in Table 3-5); and unseasonal revegetation trials.

Thus far, propagating and planting of tubestock has only been performed for revegetation in the wet season. However, in 2024/2025 when revegetation is at its peak, tubestock will need to be grown and planted all year round. Revegetating between September – November (the 'build-up') may present some challenges; propagation will be needed during the cooler, dry months when seed germination and plant growth are typically very slow, and planting will occur during the hottest and most humid time of year when there is still minimal rainfall. Propagation for the unseasonal trials began in April 2020 and planting is scheduled for October 2020.

Table 3-5: Stage 13.1 Wet season treatments and rationale

Treatment		Rationale
1- 4	Different sources of microbes [1] local microbes [2] no microbes [3] commercial only [4] combination of local and commercial microbes	Microorganism inoculation has become standard practice in many commercial nurseries due to the vital role microbes perform in plant nutrient acquisition. The 2009 TLF planting tubestock potting mix included spores of locally collected fungi. These treatments are to assess whether tubestock seedlings have improved growth/survival when inoculated with microbes from different sources. Commercially produced microbial additives for potting mix are becoming routinely used by nursery and horticultural industries. Locally sourced microbes may perform better than commercial microbes because they are adapted to the environmental conditions of Kakadu and have evolved with the plant species that are being used for revegetation. However, there is concern that inoculation with a local microbe mix sourced from inside the RPA (which is regularly disturbed by fire) will not have sufficient quantities or diversity of micro-organisms. It may be that a combination of local and commercial microbes are needed for improved plant growth and survival.
5	Plastic nursery tubes (50 x 120 mm)	Although nursery tubes are the commercial standard for revegetation, past experience at Ranger suggests biodegradable pots may be a preferable option as they eliminate the need to depot.
6	Irrigation “hardening off”	By slowly reducing the frequency of watering a few weeks before transplanting, the tubestock may be better adapted to ‘cope’ with the harsh field condition of the FLF.
7	Additional material in planting hole	Plant available water is a key concern for plant survival. Provision of additional growth medium may result in increased success. The additional material consists of approximately 2L of regular potting mix combined with the residual solid material used for local microbe application (collected mulch, puffballs, manure).

Approximately 1200 tubestock of 22 different species were planted at Stage 13.1 in April 2020 for the wet season trials (Table 3-6). All of the species had treatments 1 and 4, however the remaining treatments were only applied to select species so that the size of the study was manageable. Three of the treatments (2, 3 & 6) were trialled with three framework species: *Eucalyptus tetrodonta, Petalostigma quadriloculare* and *Terminalia ferdinandiana*. These species were also trialled with the remaining two treatments (5 & 7) along with three other species, *Brachychiton megaphyullus, Buchanania obovata* and *Grevillea decurrens*. These 'focus' species were chosen as they represent different community stratum (OS, MS, and US), they are from different Families (Myrtaceae, Picrodendraceae, Combretaceae, Malvacea, Anacardiaceae and Proteaceae), and are a combination of evergreen or deciduous.

Table 3-6: Stage 13.1 Trial Species

Species	Lifeform	Family	Treatments
Midstorey and Overstorey			
Acacia difficilis	Shrub	Fabaceae	1 & 4
Acacia dimidiata	Shrub	Fabaceae	1 & 4
Acacia mimula	Shrub	Fabaceae	1 & 4
Brachychiton megaphyllus	Shrub	Malvaceae	1, 4, 5 & 7
Buchanania obovata	Shrub	Anacardiaceae	1, 4, 5 & 7
Corymbia bleeseri	Tree	Myrtaceae	1 & 4
Corymbia chartacea	Tree	Myrtaceae	1 & 4
Corymbia dunlopiana	Tree	Myrtaceae	1 & 4
Corymbia foelscheana	Tree	Myrtaceae	1 & 4
Corymbia latifolia	Tree	Myrtaceae	1 & 4
Corymbia porrecta	Tree	Myrtaceae	1 & 4
Erythrophleum chlorostachys	Tree	Fabaceae	1 & 4
Eucalyptus miniata	Tree	Myrtaceae	1 & 4
Eucalyptus phoenicea	Tree	Myrtaceae	1 & 4
Eucalyptus tetrodonta	Tree	Myrtaceae	1 – 7
Grevillea decurrens	Shrub	Rubiaceae	1, 4, 5 & 7
Melaleuca viridiflora	Tree	Myrtaceae	1 & 4
Terminalia ferdinandiana	Shrub	Combretaceae	1 – 7
Understorey			
Cymbopogon bombycinus	Grass	Poaceae	1 & 4
Eriachne obtusa	Grass	Poaceae	1 & 4
Heteropogon triticeus	Grass	Poaceae	1 & 4
Petalostigma quadriloculare	Shrub	Picrodendraceae	1 – 7

3.5 Vegetation performance in waste rock

The revegetation trials conducted over the last decade have continued to reinforce many aspects of the first ARRTC-endorsed Ranger Revegetation Strategy (Reddell & Meek 2004), which was first formed over 15 years ago based on research conducted in the 80s, 90s and early 2000s. However, as ERA gather additional data and further revegetation experience, the Revegetation Strategy evolves. Some of the key learnings from recent revegetation trials (discussed in greater detail in the following sections) include:

- The FLF growth medium layer to be predominately grade 1 waste rock material with no purposely mixed laterite incorporated as was previously considered (over a decade ago). This is due to: 1) a lack of suitable laterite material of sufficient quantity for the FLF; 2) vegetation generally performing better on waste rock only substrates in terms of germination and survival; and 3) areas with high proportions of laterite material showing higher risk of weed infestation.

- The majority of revegetation will be performed through tubestock planting. In almost all cases, tubestock areas have out-performed direct seeded areas in terms of plant survival, height, DBH (diameter at breast height), stem density, species diversity, production of flowers and fruit, and recruitment (Daws & Gellert 2010, 2011, Gellert 2012, 2013, 2014, Gellert & Lu 2015, Lu 2015, 2016 unpublished reports, Wright & Parry 2018, 2019, 2020 unpublished survey data).

- Initial revegetation will be irrigated during the first few months following introduction, regardless of season, as plant survival can be significantly impacted by water availability.

3.5.1 Overstorey and midstorey performance

3.5.1.1 Survival and establishment

Plant mortality is often highest in the first few months following planting, as the seedlings recover from any transplant shock and adjust to the new, harsher field conditions. At the TLF, initial mortality of the 2009 tubestock was very high. Overall survival after six months was 40% in section 1A and 36.3% in section 3 with irrigation; this was still significantly greater than the non-irrigated areas, which had 13% and 22.7% survival in 1A and 3 (Daws & Gellert 2010). It should be noted that there were issues in the 2009 planting relating to tubestock quality and irrigation reliability that may have contributed to this high initial mortality. Overall initial survival was considerably better for the tubestock planted in January 2010, with 73.6% and 55.3% survival in the irrigated areas of 1A and 3 eight months after planting (Daws & Gellert 2011). Surprisingly, survival in the non-irrigated areas was not significantly different to the irrigated areas; this is presumably because of the high and consistent rainfall between January – April in 2010, which was 16 % above the mean for that period (Jabiru Airport, Bureau of Meteorology 2020) (Figure 3-7) (Daws & Gellert 2011). Over 109% more rainfall was delivered in March and April 2010 compared to the same period in 2009 (Jabiru Airport, Bureau of Meteorology 2020). This clearly demonstrates that annual rainfall variability can have a significant impact on initial tubestock survival, and that irrigation is critical to avoid complete revegetation failure in the event that Jabiru experiences a poor wet season.

After three months, the Stage 13.1 wet season revegetation seems to be tracking similarly to the 2010 tubestock on section 1A (Table 3-7). Mortality appeared to slow after 10 weeks with overall survival stabilising around 75% in the following four weeks; overall health of the tubestock also increased during that time as they slowly became less stressed (Figure 3-8). Some of the best performing MS and OS species thus far are *Brachychiton megaphyllus* (88%), *Buchanania obovata* (91%), *Grevillea decurrens* (90%), *Melaleuca viridiflora* (95%), and *Terminalia ferdinandiana* (88%).

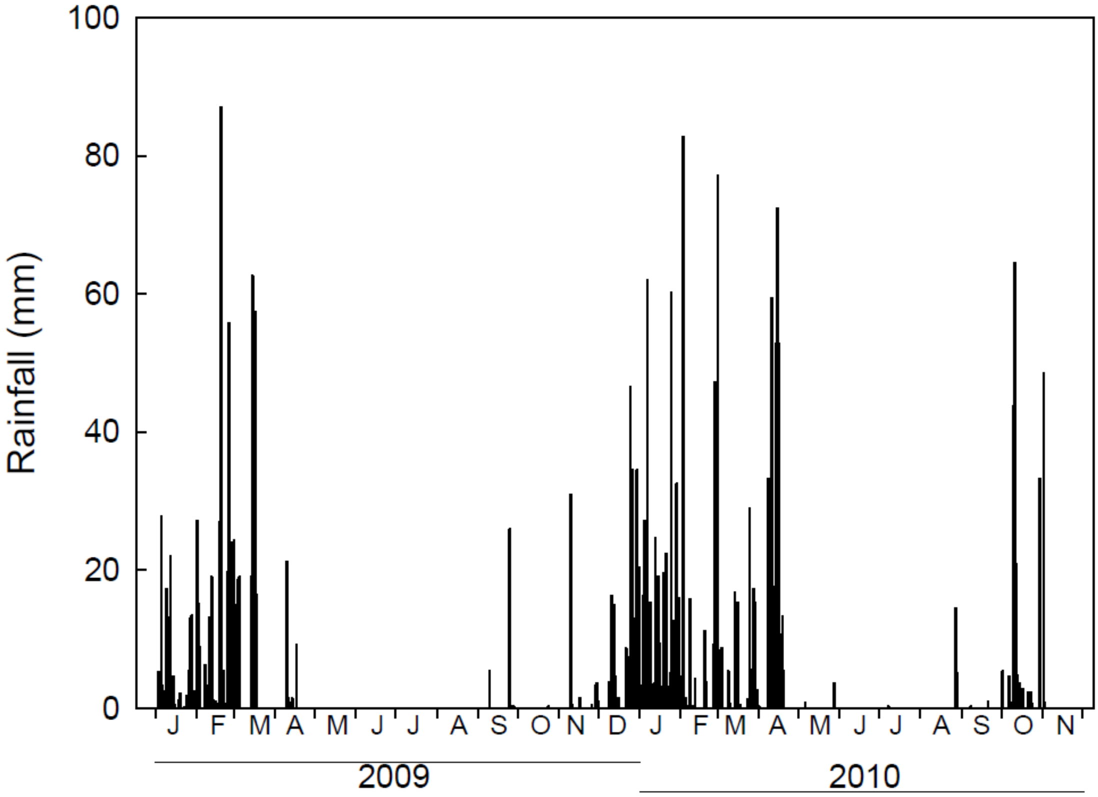


Figure 3-7: Daily rainfall for 2009 – 2010. Data up to 17 April 2009 from Jabiru Airport (Bureau of Meteorology): subsequent data from the TLF.

Initial results from the TLF direct seeding appeared promising. Although sowing was performed during the dry season, a considerable number of seedlings emerged in both sections of the TLF (approximately 25% greater density in the waste rock only substrate). Interestingly, the irrigated seeding in July 2009 was significantly more successful than the non-irrigated seeding in December 2009, despite the above-average rainfall over the 2009/2010 wet season (Daws & Gellert 2011). It's possible that the lower temperatures experienced in July were actually beneficial for germination, as the waste rock substrate surface can reach well over 50°C in the heat of the day during the build-up (September – December, depending on the year). However, it is likely that the consistent irrigation also contributed to the success of the July seeding.

Table 3-7: Initial overall survival (%) of tubestock planted on the TLF and Stage 13.1.

	Initial Overall Survival (%) of Irrigated Revegetation		
Areas	**2009 Tubestock (6 months post-planting)**	**2010 Tubestock (8 months post-planting)**	**2020 Tubestock (3 months post-planting)**
TLF waste rock (1A)	40	73.6	na
TLF laterite mix (3)	36.3	55.3	na
Stage 13.1	na	na	75

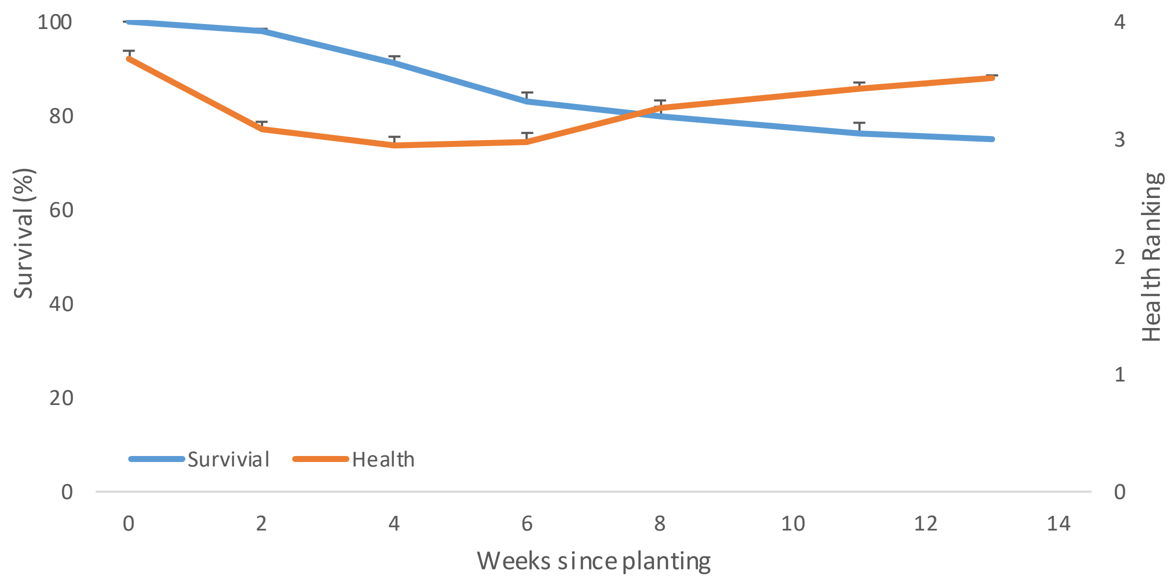


Figure 3-8: Stage 13.1 overall tubestock survival and health ranking (based on visual assessment) at 13 weeks after planting (includes OS, MS and US species data).

Whilst the TLF direct seeding seemed successful in the first year due to the high initial stem density, species compositions were skewed due to the different rates of germination. In both sections, *Acacia sp.* and *Terminalia* were amongst the more 'successful', with many of the framework Myrtaceae overstorey species germinating at lower rates (Daws & Gellert 2011). Within 18 months of seeding, infill planting was required to improve both sections' species compositions and stem densities.

Overall, 40 of the 42 tree and shrub species that were planted or direct seeded on the TLF are still present in 2020 (Table 3-8). The two species which completely failed to establish, *Erythrophleum chlorostachys* and *Stenocarpus acacioides,* were only direct seeded; *E. chlorostachys* germinated in section 2 but failed to persist beyond two years, and *S. acaciodes* seed failed to germinate despite being ~94% viable (Daws & Gellert 2011). Some species established but over time disappeared from one section of the TLF (*Acacia dimidiata, Asteromyrtus symphyocarpa*, *Grevillea sp., Hakea arborescens* and *Planchonia careya*), and others have established but have very few individuals (*Jacksonia dilitata*, *Petalostigma pubescens* and *Owenia vernicosa*).

Mean survival after ten years in the tubestock planted areas is relatively low (32 ± 4.4% in section 1A; 18 ± 3.3% in section 3) (Figure 3-9). This is partly due to the high initial mortality rates of the 2009 tubestock and the shorter-lived species naturally senescing in recent years (e.g. some of the *Acacias* and *Grevilleas*). One of the species that had particularly low survival during the revegetation of the TLF was *Xanthostemon paradoxus.* Mortality was extremely high in the six months following planting (over 95 %) which prompted a master's research project. It was found that *X. paradoxus* tubestock survival and growth was significantly improved with shading, likely due to less light and reduced heat stress (Gellert 2014). These results indicate that this species is better suited for introduction once the overstorey has had time to develop canopy and provide shade.

The species with the greatest survival on both sections of the TLF is *Eucalyptus tintinnans*. This species naturally grows on rocky ridges and appears well adapted to the Ranger waste rock media. Although *E. tintinnans* does not occur in the ecosystems adjacent to the RPA, it is native to Kakadu National Park and has been included in the Ranger Revegetation Strategy at small densities as a climate change contingency species.

3.5.1.2 Stem density

Throughout the life of the TLF, stem densities have consistently been greater in the waste rock sections compared to the laterite mix sections due to better germination and/or survival of the trees and shrubs (Figure 3-10). As of 2019, section 1A had the greatest stem density (of individuals >1/5m height) at approximately 727 stems/ha^{-1}, followed by 1B, 3 and 2 at 534, 354, and 200 stems/ha^{-1} respectively (Table 3-9). Self-recruitment was also highest in 1A, with approximately 290 recruits, followed by sections 3, 1B and 2 with approximately 146, 98 and 75 recruits respectively.

Table 3-8: The status of overstorey and midstorey species that were planted and/or direct seeded on the TLF between 2009 and 2011 (as of May 2020).

Species	Family	1A	1B	2	3
Acacia dimidiata	Fabaceae	Present	Not Present	Present	Present
Acacia hemignosta	Fabaceae	Present	Present	Present	Present
Acacia latescens	Fabaceae	Present	Present	Present	Present
Acacia mimula	Fabaceae	Present	Present	Present	Present
Asteromyrtus symphyocarpa	Myrtaceae	Present	na	na	Not Present
Brachychiton diversifolius	Malvaceae	na	Present	Present	na
Brachychiton megaphyllus	Malvaceae	Present	Present	Present	Present
Buchanania obovata	Acantahc	Present	Present	Present	Present

Species	Family	1A	1B	2	3
Cochlospermum fraseri	Bixaceae	Present	Present	Present	Present
Corymbia bleeseri	Myrtaceae	Present	Present	Present	Present
Corymbia disjuncta	Myrtaceae	Present	Present	Present	Present
Corymbia dunlopiana	Myrtaceae	Present	Present	Present	Present
Corymbia foelscheana	Myrtaceae	Present	Present	Present	Present
Corymbia latifolia	Myrtaceae	Present	Present	Present	Present
Corymbia polysciada	Myrtaceae	Present	Present	Present	Present
Corymbia porrecta	Myrtaceae	Present	Present	Present	Present
Erythrophleum chlorostachys	Fabaceae	na	Not Present	Not Present	na
Eucalyptus miniata	Myrtaceae	Present	Present	Present	Present
Eucalyptus phoenicea	Myrtaceae	Present	Present	Present	Present
Eucalyptus tectifica	Myrtaceae	Present	Present	Present	Present
Eucalyptus tetrodonta	Myrtaceae	Present	Present	Present	Present
Eucalyptus tintinnans	Myrtaceae	Present	Present	Present	Present
Gardenia megasperma	Rubiaceae	na	Present	Present	na
Grevillea decurrens	Proteaceae	Present	Present	Not Present	Present
Grevillea pteridifolia	Proteaceae	Present	Present	Present	Not Present
Hakea arborescens	Proteaceae	Present	Present	Present	Not Present
Jacksonia dilatata	Fabaceae	na	Present	Failed	na
Livistona humilis	Arecaceae	Present	Present	Present	Present
Livistona inermis	Arecaceae	Present	Present	Present	Present
Melaleuca viridiflora	Myrtaceae	Present	Present	Present	Present
Owenia vernicosa	Meliaceae	na	Failed	Present	na
Pandanus spiralis	Pandanaceae	Present	Present	Present	Present
Petalostigma pubescens	Picrodendraceae	na	Present	Present	na
Planchonia careya	Lecythidaceae	Present	Present	Not Present	Present
Stenocarpus acacioides	Proteaceae	na	Failed	Failed	na
Syzygium eucalyptoides sp. bleeseri	Myrtaceae	Present	Present	Present	Present
Syzygium eucalyptoides sp. eucalyptoides	Myrtaceae	na	Present	Present	na
Syzygium suborbiculare	Myrtaceae	Present	Present	Present	Present

Species	Family	1A	1B	2	3
Terminalia carpentariae	Combretaceae	na	Present	Present	na
Terminalia ferdinandiana	Combretaceae	Present	Present	Present	Present
Wrightia salingna	Apocynaceae	Present	Present	Not Present	Present
Xanthostemon paradoxus	Myrtaceae	Present	Present	Not Present	Not Present

Present = At least one individual present; *Not Present* = was once in that section, but no non-recruits currently present; *Failed* = species never observed despite being introduced; *na* = species never introduced

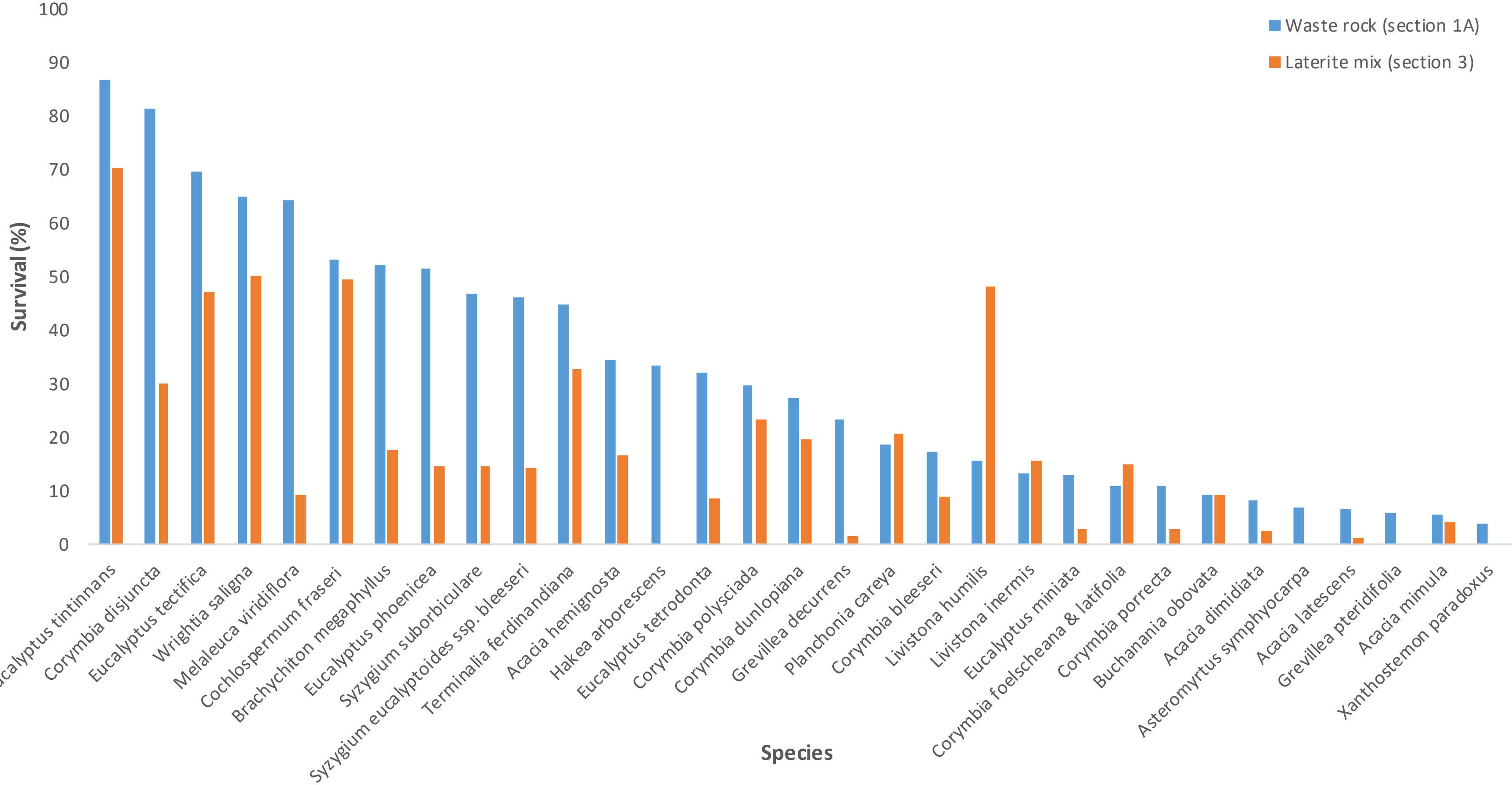


Figure 3-9: Tubestock Survival on 1A and 3 after ten years.

Calculated = (# of non-recruits present in 2019 / # planted in 2009 + 2010) * 100

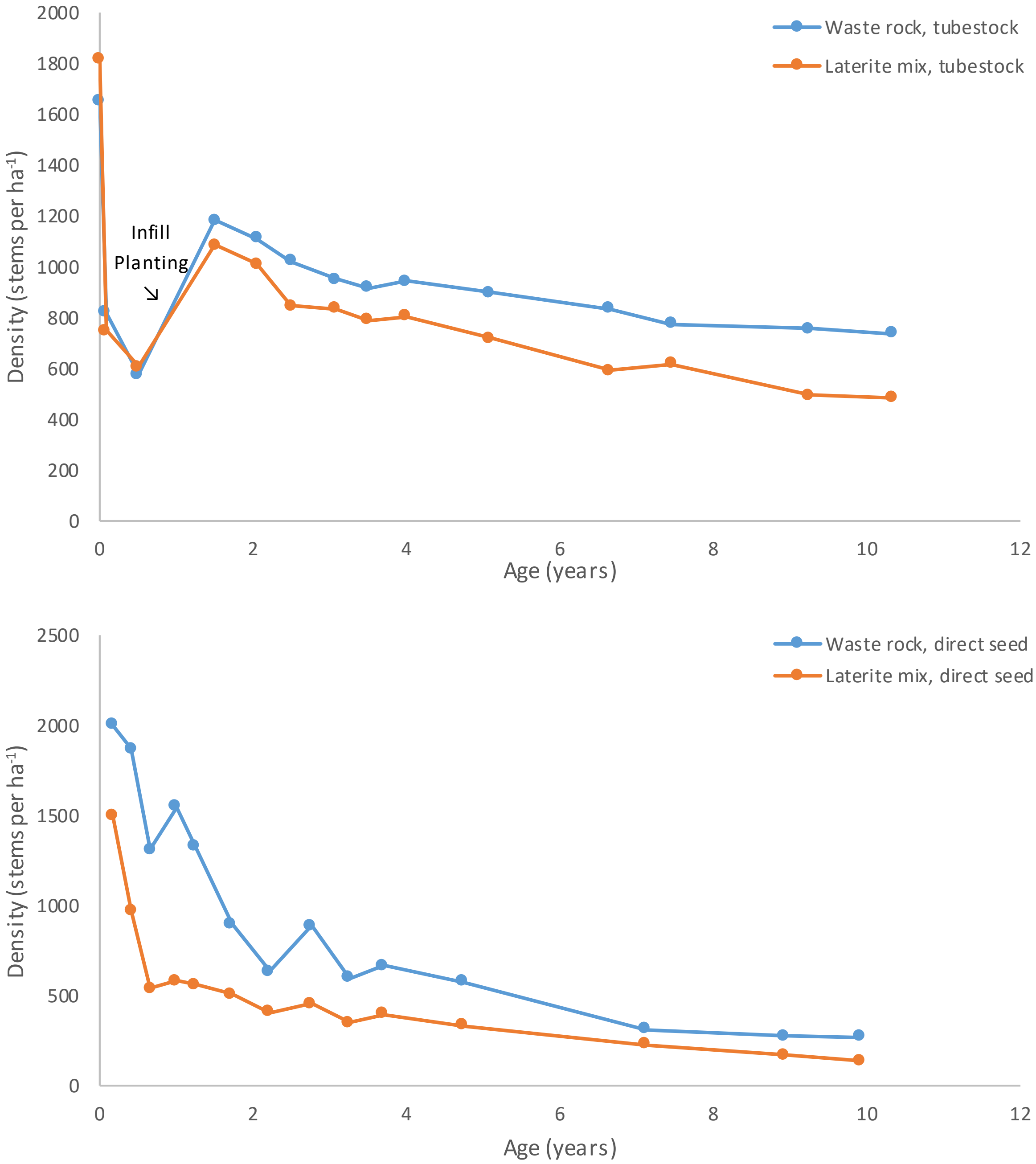


Figure 3-10: Longitudinal plant density (stems per ha^{-1}) based on the tubestock only (0 -14) and direct seeding only (15 – 29) Permanent Monitoring Plots on the TLF, not including recruits.

Note: Density is based on all introduced individuals inside the PMP regardless of height. Density before 0.5 years was calculated using the total number of seedlings in each section (estimates for direct seeded areas); the direct seeding densities do not include infill planting. It is believed that the increases in density in the directly seeded areas during the first few years were likely due to ongoing germination of the broadcast seed.

Table 3-9: Approximate total overstorey and midstorey stems on the TLF, including recruits.

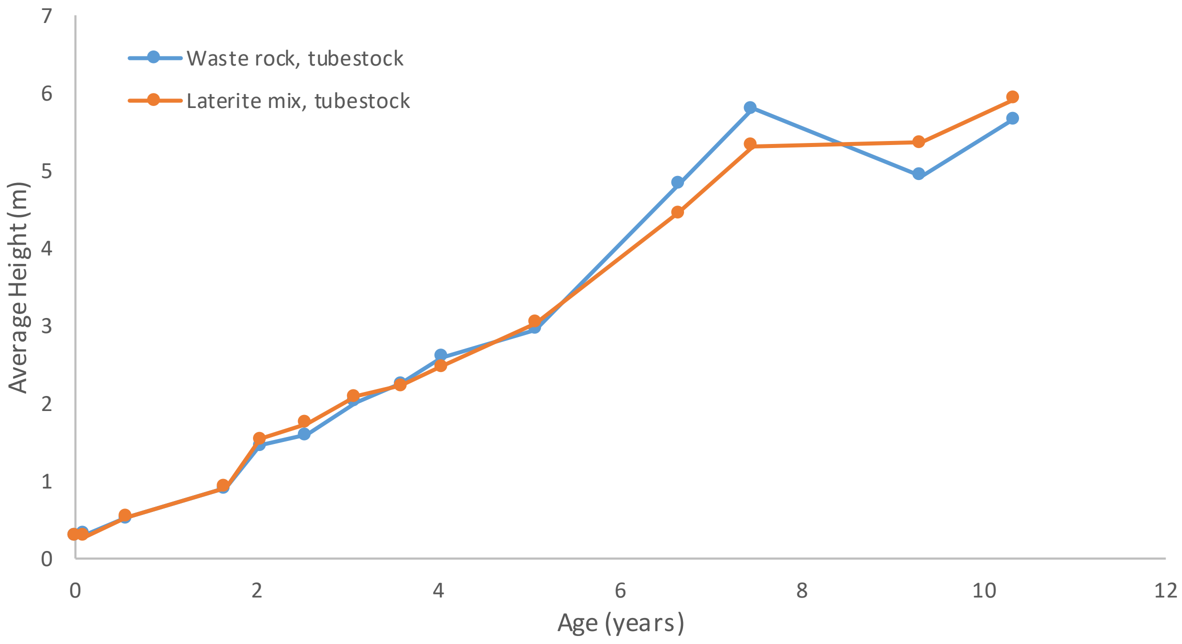
	Total # of individuals (approx.)	Total # of individuals >1.5 m	Stems per hectare (>1.5 m)
1A	967	727	727
1B	863	534	534
2	564	400	200
3	864	708	354
Total	3258	2369	296

3.5.1.3 Growth

Plant height on the TLF has not varied significantly by substrate in the tubestock areas (Gellert & Lu 2015, Parry 2019 unpublished data; Figure 3-11). In the first five years, mean height in the waste rock and laterite mix tubestock sections was almost identical, with around 60 cm of plant growth per year. Mean height almost doubled in the following 2.5 years, reaching a peak average height of 5.8 m in the waste rock section in August 2016. Cyclone Marcus brought heavy destructive winds to the area in March 2018, disproportionately effecting the waste rock end of the TLF. This combined with tall *Acacias* reaching the end of their natural life-span, accounts for the reduction in height between August 2016 and June 2018. Diameter at breast height is slightly greater in the laterite mix substrate, with a mean DBH of 8.6 ± 0.4 cm in section 3 compared to 8.05 ± 0.46 cm in 1A (based on 2019 PMP data).

Growth differences between the substrates is more pronounced in the direct seeded areas of the TLF, with lower mean plant height in the waste rock section. Plant DBH is also lower in the waste rock, with a mean DBH of 6.11 ± 0.8 cm in 1B compared to 7.73 ± 0.92 cm in section 2 (based on 2019 PMP data).

The considerable differences in growth between the two direct seeded areas are likely due (at least partially), to a greater proportion of taller species in section 2 (Gellert 2013). It is also possible that the TLF's mean plant height and DBH has been somewhat skewed towards larger plants in the laterite mix areas (particularly the direct seeded section), considering a greater proportion of smaller plants died in the 2016 burn conducted on those areas (Figure 3-12 and Figure 3-13).



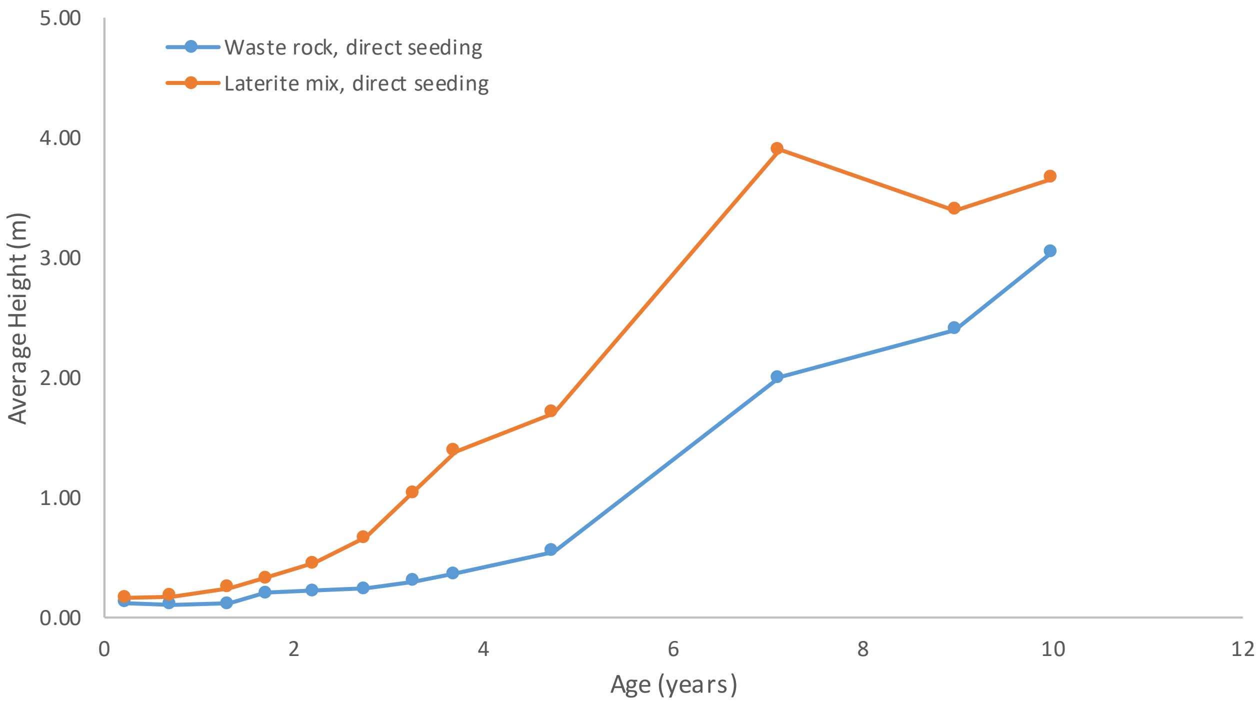
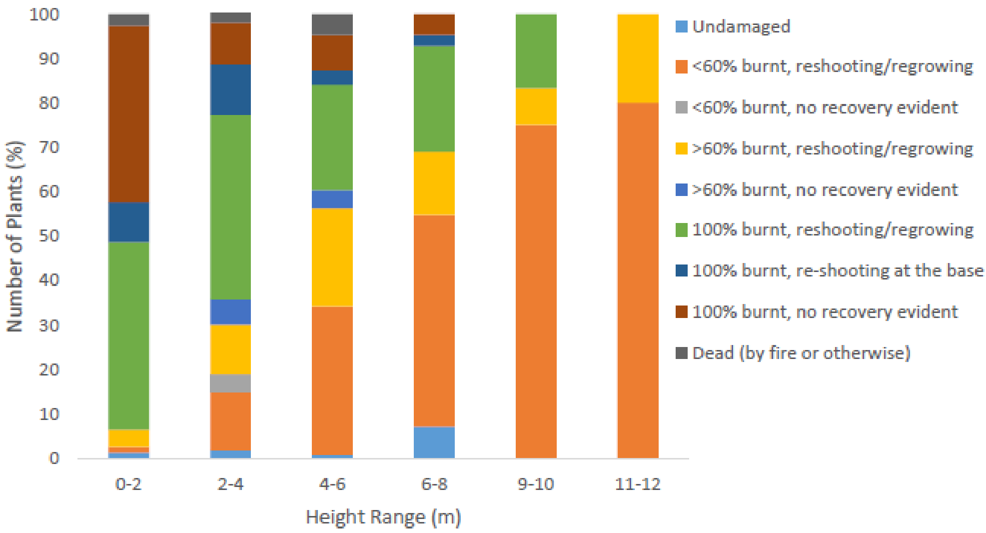


Figure 3-11: Longitudinal plant growth (height) based on the tubestock only (0 -14) and direct seeding only (15 – 29) Permanent Monitoring Plots on the TLF, not including recruits

Note: For the tubestock graph, the data points at 0.1 and 0.6 years are the average heights of the 2009 tubestock; from 1.5 years onwards, the graph is the combined average height of the 2009 and infill 2010 tubestock. Direct seeding height does not include the 2011 infill planting.

Figure 3-12: Recovery of the revegetation from a prescribed burn in May 2016. View of the burnt vegetation on the trial landform 12 days post fire (left) and 6 months post fire (right)



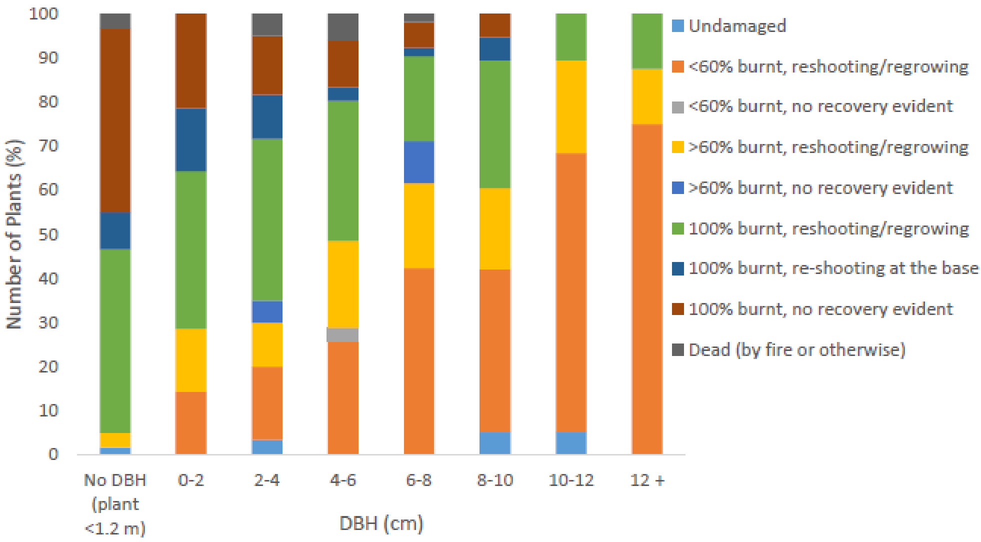


Figure 3-13: Height and DBH ranges and associated health classes after the 2016 burn on laterite mix areas of the TLF (Wright 2019a)

3.5.1.4 Flowering, fruiting and self-recruitment

Of the 40 OS/MS species that were introduced on the TLF between 2009 and 2011 and are still present today, 37 have flowered and fruited at least once since September 2018 (when monthly walk-throughs began) (Table 3-10). Over half of the species have flowered and fruited in every section that they are still present, including the majority of *Corymbias* and *Eucalyptus*. The three species that have not flowered and fruited at all include *Gardenia megasperma, Owenia vernicosa* and *Pandanus spiralis*, all of which were direct seeded. These species have grown very slowly (most <1 m) and are generally still too small to flower and fruit. Overall, species appear to flower and fruit most consistently in 1A, and least consistently in 1B.

Almost three-quarters of the OS/MS species on the TLF have self-recruited, either via seed and/or vegetative reproduction (suckering) (Table 3-10). Nine of the species have recruited in every section that they are present and another twelve have recruited in at least half of the sections they are present. Twelve species have had no observed recruitment. This includes the three species that have not fruited, and another five that have very few individuals on the TLF (*Acacia dimidiata*, *Asteromyrtus symphyocarpa, Jacksonia dilatata, Petalostigma pubescens* and *Xanthostemon paradoxus*). It is less clear why *Brachychiton diversifolius, Planchonia careya, Syzygium eucalyptoides* subsp. *eucalyptoides* and *Terminalia carpentariae* have had no self-recruitment; however, it is possible that these species actually have recruited, but the seedlings were either missed or died before the next walk-through was conducted.

Although the majority of the MS/OS species have had at least one observed instance of self-recruitment, most seedlings survive for a few months before disappearing, typically towards the end of the dry season. Only nine of the TLF species, many of which began self-recruiting within five years (Gellert 2014), have obvious recruits that have survived for over twelve months (Table 3-10).

The species with the greatest levels of self-recruitment are *Acacia hemignosta* and *Cochlospermum fraseri*. It appears that *C. fraseri* in particular is very suitably adapted for the waste rock only substrate, with almost one hundred recruits greater than 1.5 m in section 1A (Parry 2019 unpublished data). Not only does this significant level of recruitment contributed to 1A's high stem density, it also skews the section's species composition, which Gellert (2014) predicted may occur. It should be noted that *C. fraseri* recruitment is considerably lower in the other three sections of the TLF. It appears that the head-start the species received being tubestock planted rather than direct-seeded, combined with the rocky substrate, allowed *C. fraseri* to thrive and aggressively recruit in the 1A. This information is important for planning future planting densities.

Fire also appears to be an important factor influencing self-recruitment. *Eucalyptus tetrodonta* and *Wrightia saligna* in particularly have considerably more recruitment in the laterite mix sections compared to the waste-rock only sections, with the recruitment being almost entirely through vegetative reproduction (suckers) in section 2 and 3, versus seed in sections 1A and 1B.

Overall, section 1A has had the greatest number of species recruit (79% of the species present), followed by section 3, 1B and 2 (48%, 44% and 39% respectively). There may be several reasons why the level of recruitment is considerably higher in 1A than the other areas of the TLF. Section 1A has the most species fruiting and the highest density of shrubs and trees, therefore more individuals to potentially drop seed and recruit. Section 1A also has greater canopy cover and ground litter than the other sections of the TLF; although in natural systems shade and litter may impede recruitment, it is possible that on the harsh conditions of the TLF they provide a beneficial microclimate for early seedling establishment (Parry 2018). Lastly, section 1A has never had a dense weedy groundcover, unlike sections 2 and 3, which can outcompete young emerging recruits.

Table 3-10: Flowering, fruiting and self-recruitment of the MS/OS species on the TLF

Species	Flowering and Fruiting	Self-recruiting
Acacia dimidiata	At least 1 section	Not observed
Acacia hemignosta	All sections species is	All sections species is
Acacia latescens	All sections species is	All sections species is
Acacia mimula	At least 1 section	At least 1 section
Asteromyrtus symphyocarpa	All sections species is	Not observed
Brachychiton diversifolius	At least 1 section	Not observed
Brachychiton megaphyllus	All sections species is	At least 1 section
Buchanania obovata	All sections species is	All sections species is
Cochlospermum fraseri	All sections species is	All sections species is
Corymbia bleeseri	At least 1 section	At least 1 section
Corymbia disjuncta	All sections species is	At least 1 section
Corymbia dunlopiana	All sections species is	At least 1 section
Corymbia foelscheana	All sections species is	At least 1 section
Corymbia latifolia	All sections species is	At least 1 section
Corymbia polysciada	All sections species is	At least 1 section
Corymbia porrecta	All sections species is	At least 1 section
Eucalyptus miniata	At least 1 section	All sections species is
Eucalyptus phoenicea	All sections species is	At least 1 section
Eucalyptus tectifica	All sections species is	At least 1 section
Eucalyptus tetrodonta	All sections species is	All sections species is
Eucalyptus tintinnans	All sections species is	At least 1 section
Gardenia megasperma	Not observed	Not observed
Grevillea decurrens	At least 1 section	At least 1 section *
Grevillea pteridifolia	All sections species is	At least 1 section
Hakea arborescens	All sections species is	At least 1 section
Jacksonia dilatata	All sections species is	Not observed
Livistona humilis	At least 1 section	At least 1 section
Livistona inermis	At least 1 section	At least 1 section
Melaleuca viridiflora	All sections species is	All sections species is
Owenia vernicosa	Not observed	Not observed
Pandanus spiralis	Not observed	Not observed
Petalostigma pubescens	At least 1 section	Not observed
Planchonia careya	At least 1 section	Not observed
Syzygium eucalyptoides sp. bleeseri	At least 1 section	At least 1 section
Syzygium eucalyptoides sp. eucalyptoides	At least 1 section	Not observed
Syzygium suborbiculare	At least 1 section	At least 1 section
Terminalia carpentariae	All sections species is	Not observed
Terminalia ferdinandiana	All sections species is	All sections species is
Wrightia salingna	All sections species is	All sections species is
Xanthostemon paradoxus	At least 1 section	Not observed

* Species with recruits >12-months-old

Figure 3-14: Flowering and fruiting on the Trial Landform. Top right to bottom left: *Brachychiton megaphyllus, Jacksonia dilatata, Eucalyptus tectifica, Cochlospermum fraseri*

3.5.2 Understorey performance

Experience at Ranger suggests that understorey species are more likely to establish successfully when tubestock planted rather than direct seeded, particularly during the initial revegetation of waste rock when there is no shade or organic matter.

All attempts at direct seeding grasses on the TLF in the first few years following construction were ultimately unsuccessful. The grass trials either had minimal seed germination (Gellert 2014), or when germination did occur, seedlings failed to recruit and persist for longer than a year (Gellert 2012b). It's likely that irrigation and/or fertiliser would have improved the outcome of these trials. The 2012/2013 wet season was particularly dry and warm, with 21% less rainfall than normal and December - February being in the 95th temperature percentile (December 2012 the hottest on record) (Jabiru Airport, Bureau of Meteorology).

During a 1993 directly-seeded grass trial, some native understorey cover was able to establish and persist on an old waste rock dump capsite (Gray & Ashwath 1994). However, multiple factors likely contributed to this trial's success, including:

- A favourable study site – the trial was conducted on a 'substantially weathered' section of the dump located below the upper level batter slope. The site was ripped and graded, and each plot was raked to remove as many rocks with a >20cm diameter as possible;
- Irrigation – substantial irrigation was provided throughout the first few months of the trial;
- Favourable microsite conditions – shade cloth was secured over the experimental plots during germination and early establishment of the seedlings (for up to two months). This was to protect against seed loss from wind, but it also would have provided shade, which likely reduced irradiance, surface temperatures and soil water evaporation.

Direct seeding on the TLF has been somewhat more successful in recent years. In the 2018 trial, mean emergence from germinable seed ranged from 0 – 19 % for all species with the exception of *Galactica tenuiflora* in the surface litter treatment, which had 46 % emergence from germinable seed (Parry 2018). All the species had greatest emergence and number of surviving seedlings in the surface litter treatments, likely because the litter improved the seedlings microclimate by retaining water and reducing surface temperature. The surface litter may also have protected the seeds/seedlings from rain wash or uprooting, and predation. There has been significant mortality over the two years following seeding, with the best performing plots having fertiliser, surface litter, or a combination treatment (Figure 3-15) (Parry 2019 unpublished data).

Although some amelioration treatments have been found to improve directly seeded understorey establishment (Parry 2018), tubestock planting has consistently better survival and significantly higher rates of self-recruitment (Parry 2018, 2019 & 2020 unpublished data). The tubestock planted in 2018 still had up to 92 % survival after one year for all the species; after two years, the legumes had begun flowering and fruiting and the grasses had produced 2 – 3 generations of recruits (Parry 2020 unpublished data).

Initial survival of the US species tubestock planted at Stage 13.1 is generally high. Two of them are amongst the best performing species, *Heteropogon triticeus* and *Cymbopogon bombycinus,* with 100 ± 0 % and 90 ± 4 % survival respectively. The other two US species, *Eriachne obtusa* and *Petalostigma quadriloculare* have lower initial survival rates, but in the case of *E. obtusa,* it is likely that more seedlings are actually alive and have simply browned off due to the dry season.

It appears that some understorey species are more suited for 'secondary' establishment, even when tubestock planted. On the TLF, some species performed much better when planted in section 1A compared to those planted in the more open areas of 1B. It is likely the greater density of trees in 1A provided shade, reduced evaporation and surface temperature, protected the plants from drying, damaging winds, and made the area less accessible to herbivorous animals (Parry 2018). *Alloteropsis semialata* and the legumes had considerably lower mortality (particularly *I. saxicola*) and greater growth (particularly *G. tenuiflora*) in section 1A. The *Eriachne* grasses were the most successful in terms of recruitment, and had similar levels of survival, growth and recruitment in 1A and 1B. These results indicate that *A. semialata* and the two legumes likely require a more developed overstorey/soil for optimal establishment, whereas *Eriachne* could be introduced on less developed, more open landscapes if needed.

Overall, of the 24 understorey species that have been actively introduced to the TLF via seed and/or tubestock, eight have persisted, flowered and fruited, and a further four have recruited (Table 3-11). This number will likely increase in the next 12 months as the species introduced in February 2020 have the chance to establish.

Figure 3-15: Directly seeded *Galactica tenuiflora* in mixed treatment plot with fallen tree

Table 3-11: Timeline and method of understorey species introductions on the TLF.

Species	Jan-11	Nov-12	Apr-18	Jan-19	Feb-20
Acacia gonocarpa				Tubestock	Seed & Tubestock
Alloteropsis semialata			Seed & Tubestock	Tubestock	Seed
Aristida holathera		Seed		Tubestock	
Aristida inaequiglumis					Seed & Tubestock
Chrysopogon fallax		Seed			
Cymbopogon bombycinus					Seed & Tubestock
Dichanthium sericeum	Seed	Seed			
Ectrosia leporina					Seed & Tubestock
Eriachne armittii	Seed		Seed & Tubestock	Tubestock	Seed
Eriachne avenacea	Seed				
Eriachne ciliata	Seed				
Eriachne obtusa		Seed	Seed & Tubestock	Tubestock	Seed & Tubestock
Eriachne schultziana					Seed
Eriachne triseta	Seed	Seed			Seed & Tubestock
Galactia tenuiflora			Seed & Tubestock	Tubestock	
Haemodorum coccineum					Seed & Tubestock
Heteropogon triticeus					Seed & Tubestock
Indigofera saxicola			Seed & Tubestock	Tubestock	Seed & Tubestock
Petalostigma quariloculare				Seed	Seed & Tubestock
Pseudopogonatherum contortum		Seed			
Rhynchospora sp.					Seed
Templetonia hookeri					Seed
Tephrosia oblongata				Tubestock	
Triodia bitextura		Seed			

Species	Jan-11	Nov-12	Apr-18	Jan-19	Feb-20
Outcome	Failed to persist	Failed to germinate/ persist	Establishment successful via tubestock, some seeding also successful	Successful establishment	TBD, but preliminary results appear promising with tubestock and seeding

3.5.3 Ecosystem development

3.5.3.1 Exotic and weedy species

Weeds have been an ongoing issue on the TLF. In May 2009, the waste rock/laterite mix section had a weed density of 7,083 +/- 1,828 weeds/ha, whereas no weeds were identified in the waste rock only areas (Daws & Poole 2010). Daws and Poole (2010) concluded that a substantial weed seed bank was introduced with the laterite material used in constructing the landform. In addition, the waste rock only substrate was quite hostile to self-colonisation by weed species. There is still minimal weed cover on the waste rock areas in 2020, however, species have slowly begun colonised from the laterite mix areas into 1B and 1A in recent years. Paradoxically, the high ground cover contributed to higher early LFA indices on the laterite mix area, albeit confounded due to the high presence of weedy understorey (Gellert & Lu 2015).

Nineteen exotic /weedy species have been observed on the TLF since September 2018. Five of these species have not been observed since March 2020, including *Crotalaria goreensis, Cyanthillium cinereum, Echinochloa colona, Euphorbia hirta* and *Sida acuta;* however, it will take multiple months of no observations to consider them eradicated. Most of the species present today were growing in the laterite mix areas within two years after the TLF was constructed (Daws & Gellert 2010, 2011; Daws & Poole 2010). Although the number of exotic and weedy species on the TLF is similar across the four sections, the cover is significantly different. Sections 2 and 3 have recurringly dense, groundcovers of weed, whereas 1A and 1B have sparsely scattered weeds with very few dense patches.

Acacia holosericea and *Urochloa sp.* are generally considered native/naturalised species in the Northern Territory. However, due to their aggressive colonisation and dominance of disturbed areas they are considered weeds on the TLF. Within two years of the TLF construction, *A. holosericea* had germinated, grown, set seed (Gellert 2012), and were cut back at the end of 2010 to manage their spread (Daws & Gellert 2011). The cool burn performed in the laterite mix areas in July 2019 has proven to be a successful management tool for controlling *A. holosericea*. Approximately 90% of the *A. holosericea* did not recover from the burn, drastically reducing its number to only a few pockets that were protected from fire (eg. very rocky patches that did not burn, Figure 3-16). The prescribed burn also considerably changed the composition of the groundcover weed layer. Pre-burn the ground layer was dominated by buffalo clover whereas now it's predominately *Urochloa* grass, a more manageable species.

Figure 3-16: *Acacia holosericea* exposed to fire (top) and protected from fire (bottom), four months after 2019 June burn.

3.5.3.2 Species self-colonisation

At least thirty-eight native species have naturally colonised the TLF. The majority of these are understorey (Figure 3-17), however eight MS/OS species have also been observed.

Five of the OS/MS species, *Acacia difficilis, A. oncinocarpa, Alstonia actinophylla, Ficus racemosa* and *Lophostemon lactifluus* colonised the TLF well before the walk-throughs began in 2018, and are now several metres tall.

Understorey species with the greatest presence have been *Boerhavia coccinea, Brachyachne convergens, Phyllanthus sp.* and *Sporobolus australasicus* followed by *Blumea tenellula*, *Ectrosia leporina, Eragrostis cumingii* and *Marsdenia sp.* Much of the understorey diversity, particularly in 1A, comes from annual grasses, sedges and herbs. However, an increasing number of perennial species are also appearing, most recently *Indigofera linifolia* (Figure 3-18), *Tacca leontopetaloides* and *Triodia bitextura.*

Figure 3-17: Various grasses, herbs, sedges and vines that have naturally colonised the TLF.

Figure 3-18: Leguminous understorey self-colonisers on the TLF, *Indigofera linifolia* (left) and *Tephrosia sp.* (right).

As of March 2020, section 1A has a significantly greater diversity of native species colonising from external sources than the other sections of the TLF (Figure 3-19). This is likely due to 1A having a more favourable microclimate for seedlings (increased shade and litter) and having minimal weedy groundcover. Another possibility is that the 2018 Honours trial may have inadvertently increased recruitment from external sources due to increased foot traffic and low-intensity dry season irrigation. However, if this was the primary cause of increased recruitment it would reason that 1B would show similar increases, as it was watered and monitored at the same frequency as 1A.

The rate of recruitment on 1A has increased exponentially over the 18-month monitoring period. It may be that the ecosystem has reached a certain level of development were it can now sustain a native understorey. This would support the theory that species richness, particularly the understorey, will increase over time as the ecosystem develops (e.g. soil formation, nutrient cycling, overstorey canopy etc). The other waste-rock only section, 1B, has also shown an increase in the number of species recruiting over the 18-month period, however only slightly. This is another indication that section 1A is further along in its ecosystem development than 1B, undoubtedly stemming from being initially tubestock planted rather than direct-seeded.

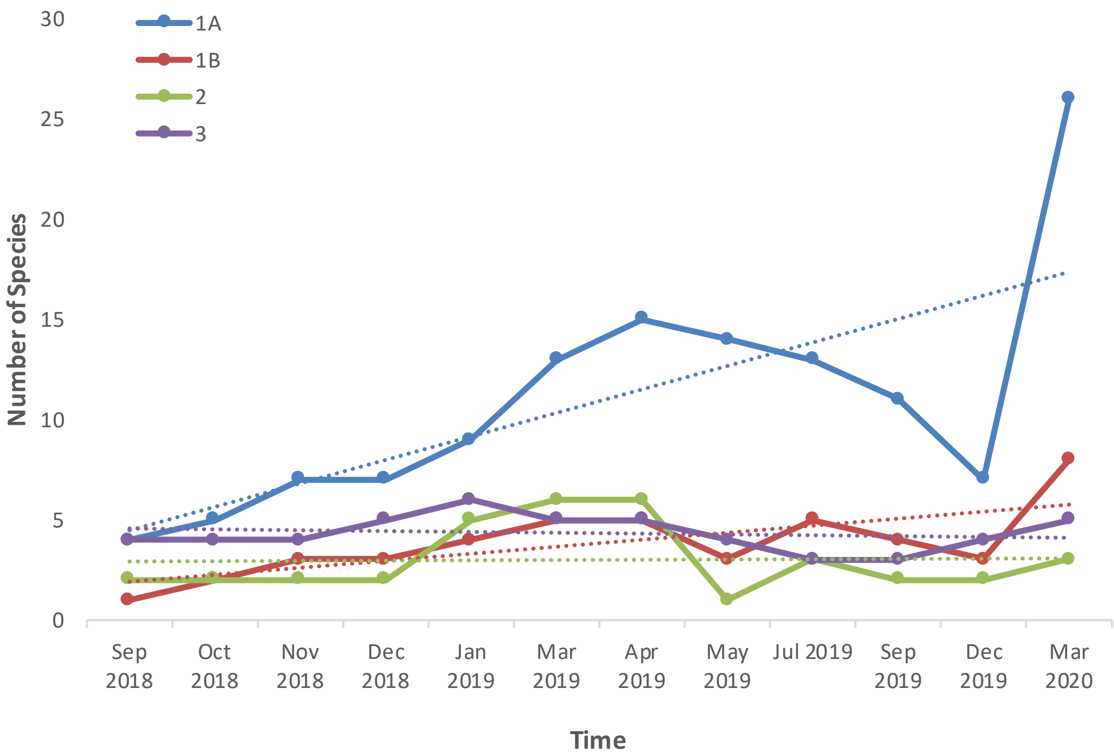


Figure 3-19: Rate and diversity of species colonising from external sources on the TLF

3.5.3.3 Fauna sightings

A variety of different faunal guilds have been observed utilising the TLF. Anecdotal observations include insects, arachnids, reptiles, birds, amphibians and mammals (Figure 28), with some occurring within the first year after construction – i.e. rock rats (Collier & Hooke 2011).

Figure 3-20: Fauna visitations on the TLF.

4 WASTE ROCK AS A GROWTH MEDIUM

At the Ranger Mine, the final landform of the disturbed area will be constructed of stockpiled run-of-mine waste rock with limited laterite and topsoil. The revegetation strategy of the final landform is therefore based on the assumption that most of the growth media will be waste rock only.

The physical characteristics of waste rock as a growth medium affects seed germination, initial survival of the young seedling (tubestock and direct seeded) and subsequent plant growth. This can make establishing diverse vegetation, especially shallow rooted understorey species, difficult. Waste rock, which has high proportions of coarse fragments, has low water-holding capacity which can cause severe surface drought and stressful growth conditions (eg. heat) for plants (Bradshaw & Chadwick 1980; Sheoran *et al*. 2010; Tordoff *et al*. 2000). Media with large sized particles can also have poor nutrient retention, and may not provide adequate root-soil contact needed for seedling establishment and survival (Chambers & MacMahon 1994).

The chemical and biological properties of waste rock can also inhibit seedling emergence, plant establishment and growth. Limiting chemical characteristics can include low organic matter content, low concentrations of plant-essential macronutrients such as nitrogen, phosphorus and potassium, acidity, salinity, and elevated bioavailability of metals (Ashwath *et al*. 1993, Bolan *et al*. 2017; Singh *et al*. 2002; Sheoran *et al*. 2010). Waste rock is also virtually devoid of soil microorganisms, such as mycorrhizal fungi, which limits mine waste revegetation by impacting nutrient cycling and microbial processes (Huang *et al*. 2012; Reddell & Milnes 1992).

The Ranger Mine is located in the seasonally wet-dry tropics, where approximately 95% of rainfall occurs between November and April, followed by an essentially rainless dry season, lasting from May to September. In this region, the most important factor shaping the landscape and determining the type of savanna ecosystems is the soil water availability and whether vegetation can survive the half-year dry season. Soil water availability is a key challenge for Ranger Mine site ecosystem re-establishment because the majority of the final landform will be constructed of waste rock growth media which often lack structure and contain large amounts of rock fragments and macro-pores that reduce their water holding capacity (compared to natural soils, Figure 4-1).

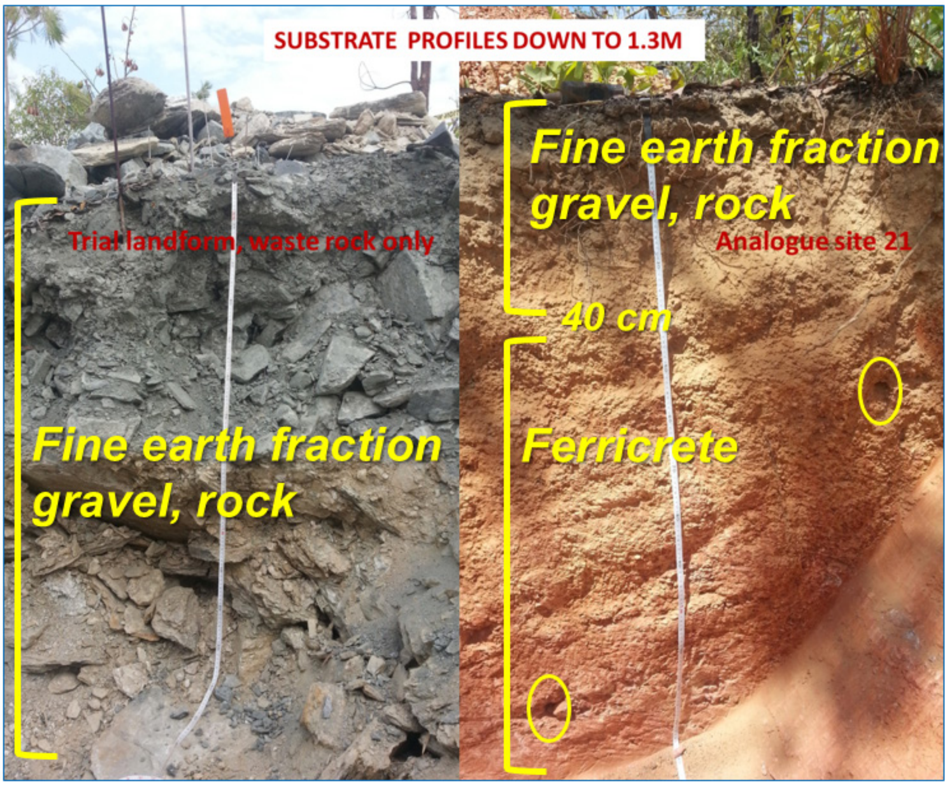


Figure 4-1: The waste rock substrate at the Ranger TLF Section 1A in 2014 (left) is fundamentally different to local substrates at Ranger's Georgetown Reference area (site 21) (right)

To address the question of whether the waste rock growth media of the Ranger final landform cover can supply sufficient plant available water (PAW) and nutrients to sustain a local native woodland, ERA has undertaken extensive research in the past three decades (Johnston & Milnes 2007, ELA 2017) on growth media particle size distribution, soil water dynamics, root depth, soil chemistry and nutrient cycling, and vegetation performance on the Ranger trial landform (TLF) (ELA 2017, Huang and You 2018, Huang *et al.* 2020, Lu 2017, Lu *et al*. 2019). This section will summarise the key knowledge on waste rock as a growth medium.

4.1 Waste rock particle size distribution

For the purpose of assessing water holding capacity of the growth media (waste rock), a key parameter is the % of the fines that are smaller or equal to 2mm in size. In soil science, only this portion of the material is considered to be able to store water for plant use.

During the construction of the Ranger TLF in 2009, waste rock samples were taken in triplicate from the surface of the TLF and at depths of one, two, three and four metres from the TLF pits (there was one pit in each of the 1A and 1B subsections that were constructed of waste rock only). These samples were sieved to determine the weights of the fraction greater than 2 mm (>2 mm) and less than 2 mm (<2 mm). Sub-samples of the fine earth fraction (i.e. <2 mm) were sent to the University of Melbourne for particle size analysis using the Bekham Coulter LP

13320 laser sizer. Particle sizes were grouped into the sand, silt and clay fractions according to the USDA size classes.

Particle size distribution (PSD) results from the TLF section 1A profile are presented in Table 4-1. Note that sand, silt and clay make up 100 % of the fine earth material particles (i.e. particles <2 mm), commonly referred to as '*fines*'. The rock content (i.e. particles >2 mm) ranges from 61 to 73 % with an average of about 67 %. This is consistent with SSB observed 70 % rock content (Mike Saynor, *pers. comm.*). A breakdown of the fines content is shown in the three right-hand columns in Table 4-1, and is similar to values published by Saynor & Houghton (2011) (Figure 4-2). Saynor & Houghton (2011) described the determination of the particle size statistics of the surface material from different parts of the TLF. In 2009 two surface material samples were collected from each of two different sample sites within each of the six treatment areas, with 24 samples collected in total (Saynor *et al*. 2012a).

Table 4-1: Particle size distribution data from TLF 1A section at construction in 2009

Depth (cm)	Total volume of material (rock and fines)		Classification and breakdown of fines portion (particles <2 mm)		
	Rock %$_{v/v}$	**Fines %$_{v/v}$**	**Sand %**	**Silt %**	**Clay%**
0	66.2	33.8	83.8 ± 1.4	14.9 ± 1.3	1.3 ± 0.2
100	68.0	32.0	82.8 ± 2.5	15.8 ± 2.4	1.3 ± 0.2
200	63.8	36.2	82.9 ± 1.2	15.7 ± 1.1	1.4 ± 0.1
300	73.0	27.0	83.6 ± 0.3	15.0 ± 0.2	1.4 ± 0.1
400	61.6	38.4	82.9 ± 2.1	15.7 ± 1.9	1.3 ± 0.2

(Source: Segura 2017)

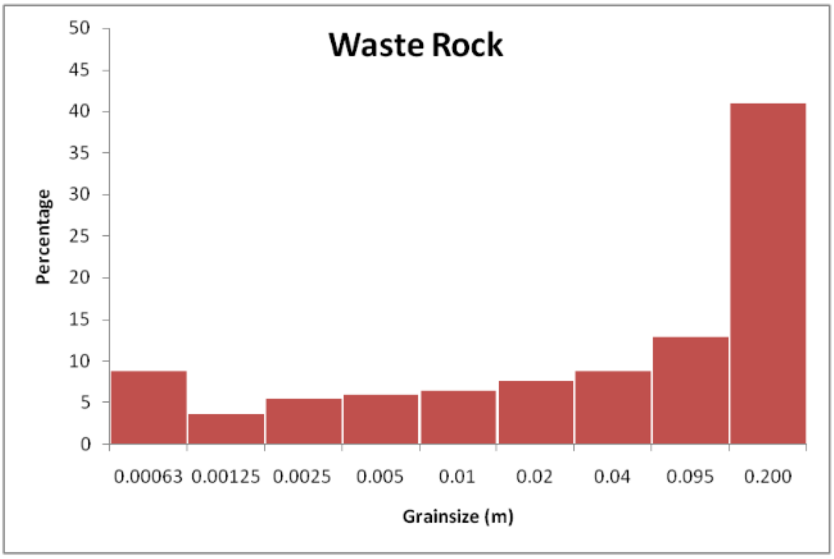


Figure 4-2: Surface grain size distribution for waste rock samples from sections 1 and 2 of the trial landform

Hollingsworth (2010) measured PSD, water content and water potential of the substrates in an experimental waste rock cover established on the northern Ranger Mine waste rock dump of the Pit 3 materials. It was reported that the substrate contains 36% of fines (<2 mm) and 64% of gravels/rocks from 24 core samples.

In an early CSIRO study on revegetated waste rock dumps at Ranger, Emerson and Hignett (1986) found that the rock fractions (> 2 mm) of the samples taken from trenches in three rock piles of Pit 1 materials were 'surprisingly' uniform and the mean was 61%, 54% and 57%, respectively (Table 1 in Emerson & Hignett 1986). These rock contents are comparable to, though consistently lower than, the TLF finding of 67% for the Pit 3 materials. These findings suggest that waste rock materials are similar in terms of fractions <2 mm particles (fines) between Pit 3 materials used for the TLF and Hollingsworth (2010), and even between Pit 1 materials (Emerson & Hignett 1986) and Pit 3 materials.

With the assistance of the Douglas Partners Geotechnical & Environmental Consultants, ERA has undertaken a PSD sampling campaign of stockpiled waste rock (2019) and also progressive sampling of the waste rock material being placed in the Pit 1 upper 6 m growth layer during the 2019/2020 construction activity (note: not all of the Pit 1 samples have been analysed yet). Figure 4-3 provides a comparison of the results available to date, and indicates that the Pit 1 backfill material is significantly finer (averaging about 40% <2mm fraction) than the estimated stockpile average (about 21% fines). The stockpile sampling data suggests that the stockpiles used for backfilling Pit 1 ('Stage 10 and some Stage 6 stockpiles) have unusually high fines compared to the other stockpiles, and so it is expected that the remainder of the material used in construction of the final landform will be more like the overall average of 20-25% fines.

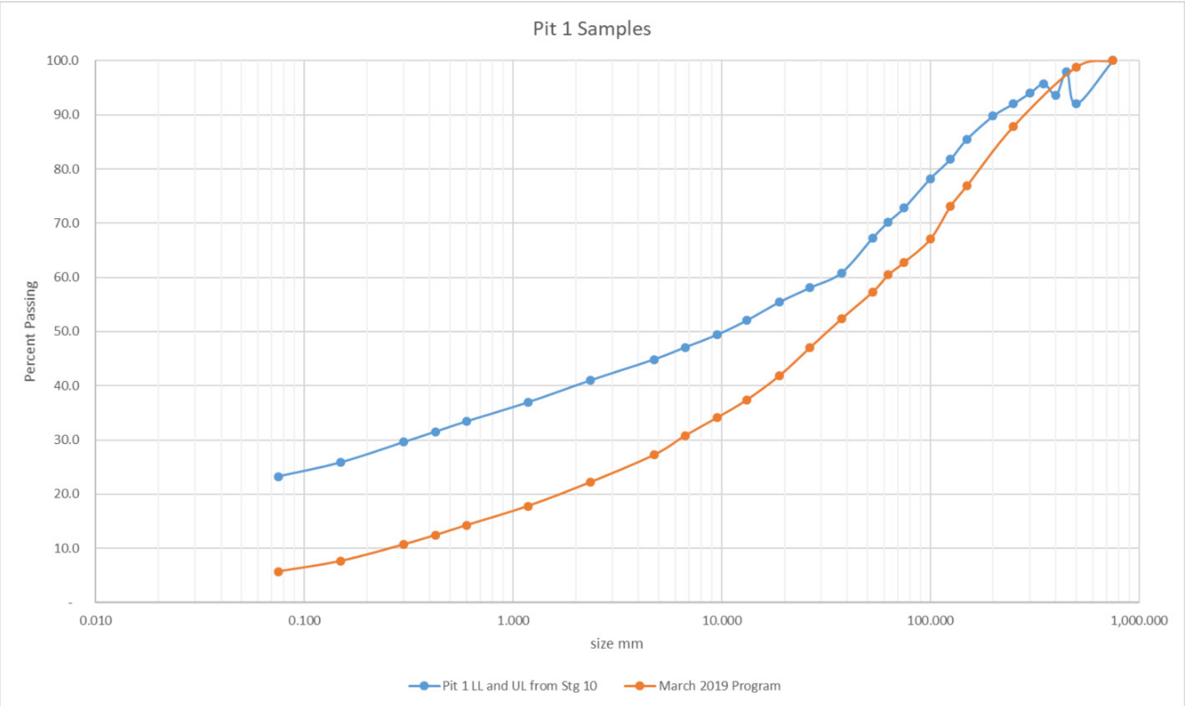


Figure 4-3: Particle size distribution for Pit 1 growth layer materials compared to 2019 stockpile samples.

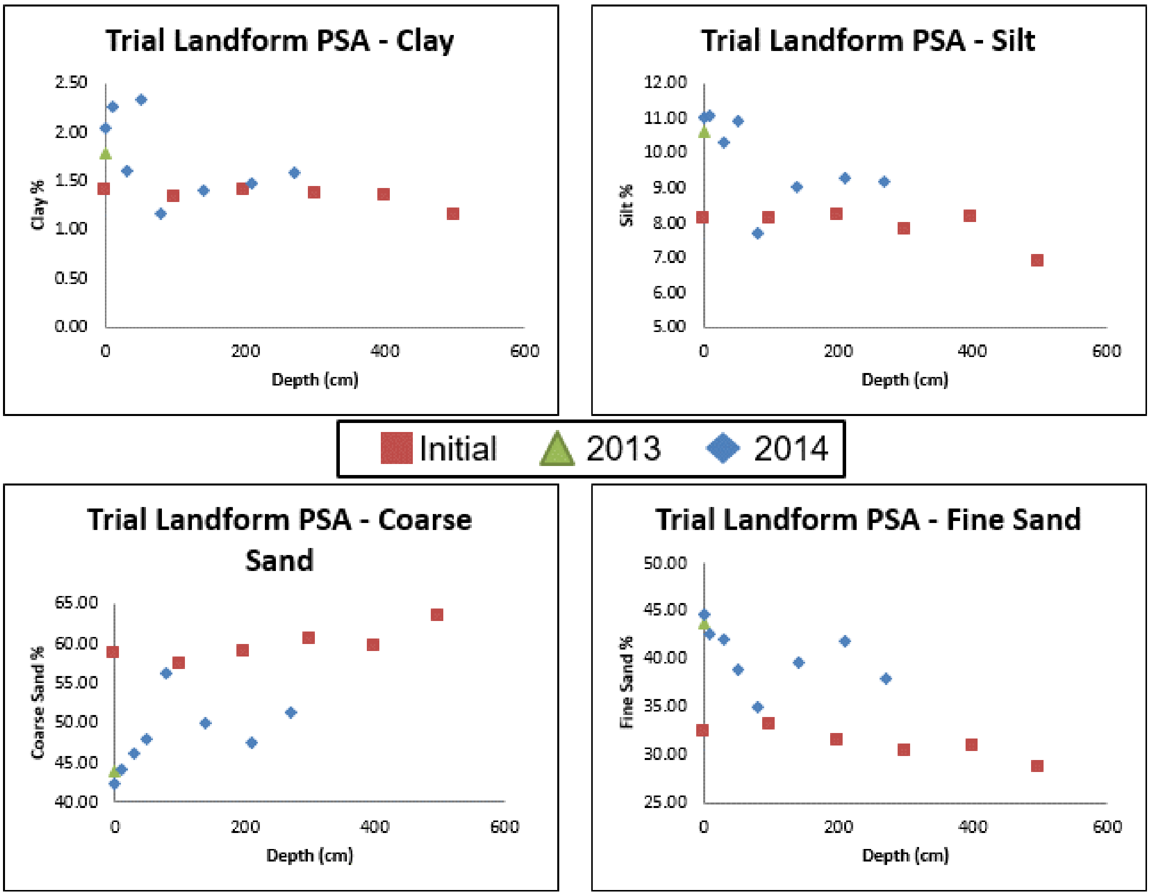
4.2 Weathering and soil development

Development of a waste rock ‘soil’ able to sustain native vegetation is a result of the complex interactions between the waste rock, plant roots, leaf litter, a range of microbial organisms and other environmental and climatic factors. Production of rock fines through weathering is one component of this, however generation and infiltration (illuviation) of organic matter is another important process (Tony Milnes, *pers. comm*. 2019).

Weathering of the waste rock over time will increase the proportion of fines in the profile, which increases the water holding capacity of the material. General observations indicate ROM waste rock on the TLF has been breaking down since initial placement as a consequence of physical, chemical and biological weathering processes, and also due to vegetation establishment and litter accumulation, and decomposition by microbial activity in the substrate. The increased proportion of fines will create a suitable substrate for understorey development. Some natural establishment of understorey species in the waste-rock-only section has been observed since 4-5 years after revegetation, which supports this theory.

Johnston and Milnes (2007) reviewed a number of early CSIRO investigations into the formation of waste rock ‘soils’ to inform the revegetation strategy and summated that weathering of much of the rock materials exposed on the surface of the stockpiles was rapid. Within two years of construction of waste rock stockpiles, properties such as colour mottling due to increased hydromorphy, variations in soil texture as a result of water erosion of fine material, structure development, decrease in pH due to pyrite oxidation and sulfate weathering were recognised by Fitzpatrick *et al* (1989).

In 2013 the University of Queensland and Charles Darwin University conducted a small-scale excavation of the TLF section 1A and particle size analysis (PSA) was undertaken to determine particle size distribution. A slight increase in fines was observed compared to proportions measured during initial construction of the TLF in 2009 (Figure 4-4and Figure 4-5).

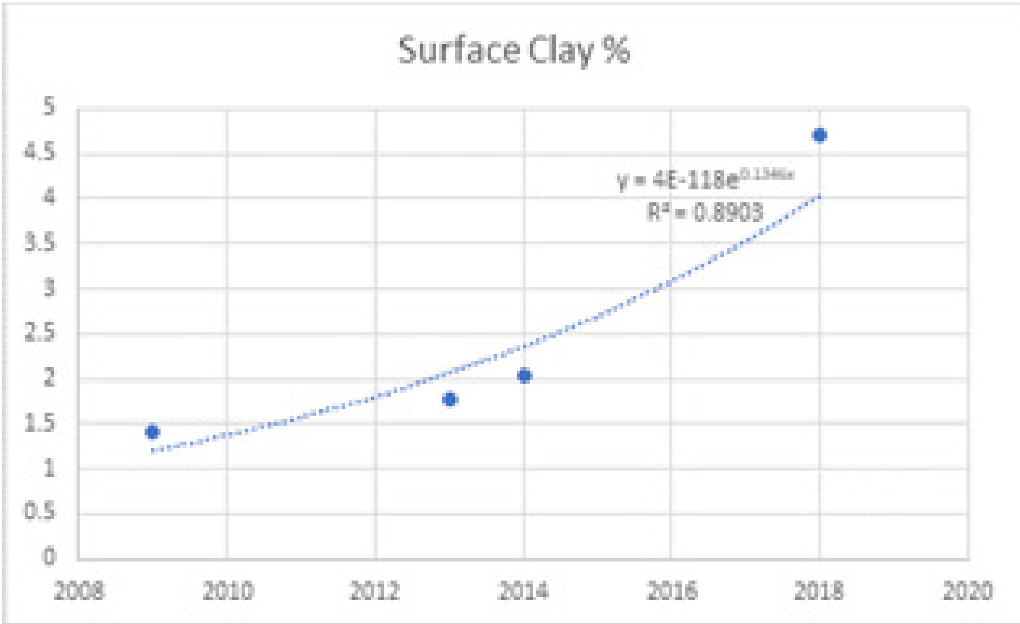


(Source: Lu *et al*. (2018)

Figure 4-4: Changes in PSD on TLF from 2009 to 2014 inclusive

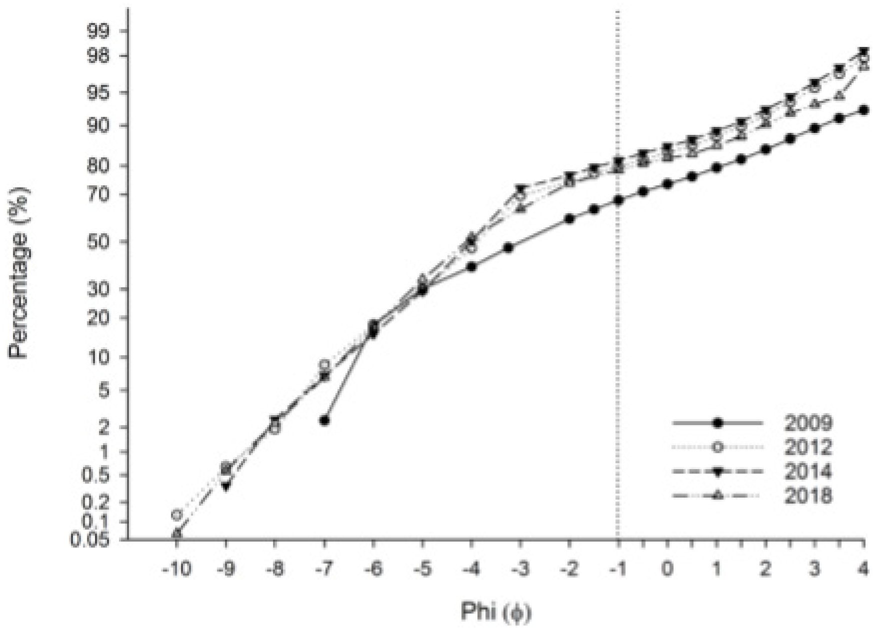
The SSB has measured, *in situ*, particle size of waste rock on the surface of the TLF since 2009 (Saynor 2019). Results indicate that the samples are exhibiting a trend of very little weathering over the five-year period (2012-2018). Measuring only surface samples risks missing the important fines that move vertically into the substrate profile, however Saynor (2019) suggests that this is only a minor 'loss', despite not having been measured. It is explained that "the near-uniformity of the cumulative particle size class distributions over time indicates such potential loss is minor over the sampling period" (Figure 4-6).

Nevertheless, the weathering measured as above did not account for the fines that were removed from the surface so the rate of material weathering is potentially underestimated.



(Source: Lu *et al.* 2018)

Figure 4-5: Changes in PSD on TLF1 (including 2018 surface soil samples) at 5 cm depth



(phi)	(mm)
-10	1024
-9	512
-8	256
-7	128
-6	64
-5	32
-4	16
-3	8
-2	4
-1	2
0	1
1	0.5
2	0.25
3	0.125
4	0.063

Figure 4-6: Cumulative percentage of particle size for waste rock on TLF (from Saynor 2019)

4.3 Chemical characteristics and nutritional processes

4.3.1 Chemical characteristics

Chemicals in substrates can play a critical role in revegetation success, including: as a limiting nutrient; a toxicant above a threshold effects level; a modifier or facilitator of other chemical processes/interactions; or a combination (Bayliss 2018).

Overall, the waste rock material at Ranger Mine differs from natural soils by having higher pH, EC, CEC, Mg, total P and SO4 concentrations, and having lower levels of nitrogen and extremely low organic carbon at the beginning of the landform establishment because the materials were just run-of-mine without topsoil (Ashwath *et al*. 1993, Gellert 2014, Table 4-2**)**.

Table 4-2: Chemical analysis of waste rock samples taken in January 2010 compared to natural soils (source Gellert 2014)

	Section 1A TLF	**Analogue sites**
paste pH	8.0 (±0)	6.3 (±0.1)
paste EC (uS/cm)	260 (±49.2)	14.4 (±2.2)
Organic C (%)	0 (±0)	0.54 (±0.08)
P (ppm)	410 (±6.6)	0.2 (±0.1)
Total P (mg/kg)	460 (±25)	64.8 (±12.6)
Total S (%)	0.03 (±0.02)	0.02 (±0.01)
NO2-N (mg/kg)	BDL	0.28 (±0.05)
NO3-N (mg/kg)	0.64 (±0.48)	0.24 (±0.08)
paste NH3-N (mg/kg)	0.07 (±0.01)	1.27 (±0.30)
Total N (mg/kg)	45.1 (±14.0)	422 (±20.5)
Ca (mg/kg)	85.8 (±23.8)	0.8 (±0.1)
K (mg/kg)	20.3 (±1.9)	4.9 (±0.0)
Mg (mg/kg)	61.7 (±18.3)	BDL
Na (mg/kg)	17.0 (±3.8)	1.2 (±0.1)
CEC	5.3 (±0.5)	3.2 (±0.2)
Al (me/100g)	0.4 (±0.1)	1.8 (±0.1)

Worth noting is that compared to waste rock from other mines in the Alligator Rivers Region, or natural soils, the Ranger Mine waste rock has higher total, exchangeable and water soluble Mg, and higher total P (Ashwath *et al*. 1993). Ashwath *et al*. (1993) also found that C:N ratio is significantly higher in Ranger waste rock (58:1) than in the natural soil (19:1). The presence of high ratio of C:N in mine waste rock than in natural soils may restrict the net release of N to plants and soils.

As part of the 2018 *Cumulative ecological risk assessment for the rehabilitation and closure of Ranger uranium mine* (Bayliss 2018) assessments of potential chemical effects on seedling plant growth and survival were made. The assessments related to toxicity thresholds reported in the literature for species (or at least genera) that will be used in revegetation at the Ranger Mine, and their potential roles as either limiting nutrients, toxicants or chemical facilitators. Bayliss (2018) arrived at the following conclusion:

> *In summary, the potential chemical risks from poor pH range (for ectomycorrhizal fungi at least) and low values of N, Ca and Mg can be discounted in the assessment given that TS can be enhanced at planting with fertilisers (e.g. broadcast or directed application) and water crystals whose effects may last up to 14 months (Daws & Gellert 2011; Gellert 2012). Additionally, Fe was discounted as a potential toxicant given the higher concentrations found on the Miniata and Heritage analogue sites, albeit closer to the minesite compared to Georgetown. Hence, in our assessment,* ***risks to revegetation from mine-derived chemicals is assumed zero*** *and, needless to say, a more thorough screening process needs to be undertaken of potential effects on seedling growth and survival to test that critical assumption. This may require experimental in situ research and pot trials to fill knowledge gaps.*

ERA presented their conclusion to ARRTC (May 2018) on vegetation growing in the waste rock on the TLF and other areas around the mine site exposed to pond water (waste rock runoff and leachate). The observations and studies of the LAAs, irrigated with pond water for over a decade, indicate there are no observed negative effects on vegetation from waste rock contaminants.

Despite these positive conclusions, it is always preferred to have site specific and species-specific information on the nutrient requirements, and toxicity risks, of target species for rehabilitation of the Ranger Mine final landform. General findings and observations may obscure specific effects that could cause sub-optimal vegetation establishment and development. For example, investigations into the effect of magnesium sulfate salinity on the germination of seeds of twenty plant species native to the Kakadu NP (Malden *et al*. 1994) found that the presence of magnesium sulfate salinity severely decreased the final germination percentages of most species and decreased the rate of germination of most species. Whilst use of tubestock planting can decrease these specific germination impacts, these effects may impact subsequent growth or impact the subsequent establishment of mid storey and under storey species from seed. Thus, as was discussed at ARRTC (May 2018), studies on plant establishment and growth rates for specific species may inform future management practices that could mitigate nutrient and toxicity effects.

4.3.2 Nutrient cycling

The diversity and sustainable growth of revegetated plants is closely related to nutrient cycling in soil-plant systems, which is driven by functional microbial communities in litter, surface soil and the rhizosphere. Microbial driven processes are critical to *in situ* litter decomposition and N/P mineralization in soil and plant uptake.

Rehabilitated sites rapidly redevelop nutrient pools in the soil, litter and understorey vegetation, but the pool contained within trees takes longer to develop. Litter accumulates rapidly in rehabilitated sites, sourced mainly from eucalypt and legume species. At bauxite mines in WA, rehabilitated areas have accumulated the same amount of litter within three to five years as unmined forest sites contain after the same period of time following burning (Ward 2000). Surface roughness (for example provided by scarification or ripping) aid these processes by ensuring that resources such as water, leaf litter and nutrients are captured and used *in situ* or recycled. The furrows also concentrate the litter, allowing decomposition processes to commence earlier.

Research by Grant *et al.* (2007) found that a critical aspect of re-establishing a self-sustaining jarrah (*Eucalyptus marginata*) forest ecosystem to mined areas is to ensure that vital ecosystem functions such as litter decomposition and nutrient cycling are returned. Significant research has been undertaken over the past twenty years relating to litter decomposition and nutrient cycling. Studies have shown that litter accumulates rapidly in restored areas (1–4 t/ha/year) and the accumulated litter tends to be richer in nitrogen due to intentionally elevated densities of nitrogen-fixing species. This leads to a lower carbon:nitrogen ratio (60:1 compared to 130:1 in unmined forest) that may promote mineralization of organic nitrogen to inorganic forms in restored areas. The major nutrient store in the unmined forest is in the soil and returning soil during the rehabilitation process largely conserves this resource, particularly in relation to phosphorus. Short-term plant macronutrient requirements for growth are readily restored by fertilizer application. Studies on the re-accumulation of nutrient pools in the successional development of restored areas have shown that pools equivalent to the unmined forest are established within ten to twenty years. Ongoing research is focusing on the rates of cycling processes in burnt and unburnt restored areas and comparing these to the unmined forest to ensure that key functions have been re-established.

4.3.2.1 Nutrient cycling studies at the Trial Landform

ERA recently commissioned a study (Huang & You 2018, Huang *et al.* 2020a) into nutrient cycling of the revegetation at the Ranger TLF compared to the Ranger Georgetown Creek reference sites. The 2018 study compared TLF-1A and Georgetown Site 21 while the 2019 study looked at TLF-1A and Georgetown Site 30, where soil is more gravelly and shallower than at Site 21. The key findings of the 2018 study are summarised in Table 4-3.

Huang and You (2018) suggest that the low mineralisation rates in the 9 year-old revegetated TLF soils may be attributed to the consequence of combined abiotic stress selection (e.g. solar radiation associated heat stress, rapid evaporation and water deficit in the surface “soil” – fine fractions of weathered rock and organic matter debris at the surface due to low ground cover (vegetation and/or litter). Water deficit could be one of the key factors limiting microbial growth and functions in soil.

In 2019 the study aimed to assess key microbial and nutrient cycling attributes of litters and surface soils from 10 year-old revegetated waste rock (TLF-1A and 1B) in comparison with a natural vegetation reference Site 30 (Huang *et al.* 2020a). The investigation characterised litter properties (e.g. elemental and organic compound composition) and a range of key soil molecular microbial, chemical and biogeochemical indicators for assessing the potential capacity of organic carbon decomposition and nutrient (particularly nitrogen (N)) cycling processes in surface soil of trial landform (TLF 1A and 1B).

The litter collected from the sites mostly contained 40-50% organic carbon and low concentrations of N and P. The organic compounds within the litter were dominant by carbohydrate, followed by protein (especially the C=O amide I) and lipids. The differences of litter chemistry were not statistically significant between the reference site and the TLF sites (Table 4-4).

Table 4-3: Key findings of 2018 nutrient cycling study (TLF-1A and Site 21)

Area	Finding
Nutrient status in litter and surface soil	After 9 years of revegetation, litter accumulated in the trial landforms showed relatively higher levels of nutrients concentrations than those collected from the analogue. Soil in the trial landforms showed lower level of nutrients concentrations than those in the analogue.
Characteristics of bacterial and fungal decomposers	Microbial communities in both litter and surface soil of the three sites were dominated by heterotrophic bacteria. Bacterial and fungal communities in trial landforms appeared to be more diverse than those in the analogue soil, however seemed to be under selection pressure which constrained their functions. Some N-fixing and plant growth-promoting bacteria were 3 times more abundant in the analogue soil than in TLF. TLF soils had abundant bacteria colonizing nutrient limiting environment, and Rozellomycota associated with early stage of soil development. Also, there was a smaller portion of stress response stain assigned to class of Bacillus enriched in soils from TLF-1A than the analogue site.
Nutrient cycling processes in the surface soil	As is expected for a 'new soil', the microbial functions related to C and N cycling in the surface soil of trial landforms were constrained, compared to the soil from the analogue site. The TLF surface soil exhibited significantly lower levels of net mineralisation rates and higher levels of metabolic quotient (representing lower carbon utilization efficacy) than those of analogue site in the wet season when microbial biomass was supposed to be significantly boosted with increased moisture and availability of C and N.

Surface soil at the reference site was more fertile compared to the rehabilitated waste rock sites (Figure 4-7). It was slightly acidic and associated with relatively high levels of organic matter (4.5% organic C) and N (>20mg/kg), especially in the form of ammonium-N. This might be attributed to long-term organic matter decomposition and humic compound accumulation, as a high density of understorey annual/perennial plant species was present at the reference site. This is consistent with the findings that surface soil at the reference site had the highest diversity of bacteria and fungi, particularly with abundant actinobacteria associated with N enrichment and fungi genera associated with woody and later stage organic matter decomposition. Metagenome prediction and *in situ* enzymatic activities showed that bacterial communities from the reference sites also had the highest capacity to drive organic matter metabolism (as an indicator of nutrient cycling).

Table 4-4. Elemental composition in the litter among sites

Element	Reference site	TLF-1A	TLF-1B
OC (%)	42.3	47.8	42.9
N (%)	0.71	0.68	0.78
P (g/kg)	0.30	0.27	0.31
K (g/kg)	0.72	0.76	0.97
Ca (g/kg)	14.19	13.36	13.80
Mg (g/kg)	1.86	2.95	5.69
Fe(g/kg)	8.70	0.68	3.28
Al (g/kg)	2.51	0.85	4.02
S (g/kg)	0.63	0.74	0.69
Mn (g/kg)	0.38	0.12	0.15
Cu (mg/kg)	7.8	4.4	10.2
Zn (mg/kg)	18.5	16.4	20.6

The surface soil from the TLF sites is slightly alkaline and less fertile than those from the reference site, as the surface soil layer is formed from the freshly formed/weathered rock fines and decomposed organic matter. The levels of organic matter of TLF soil samples were only about one third of the reference site, with even much lower levels of total nitrogen (<5mg/kg). Microbial communities in the surface soils were highly diverse and dominated by organoheterotrophs, regardless of sampling sites. Bacterial and fungal communities in the soils from the reference site had the highest diversity. The microbial communities from the reference site appeared structurally different from those of the other sites, while a few Actinobacteria associated with N enrichment and fungi associated with later stage of decomposition were abundant in the soil from the reference sites, which are capable of decomposing woody organic matter. The soils from the site of TLF-1A and TLF-1B were enriched with microbes well adapted to habitats of low moisture and infertile soils.

The surface soil from the reference site also showed the highest capacity of microbial driven organic matter decomposition and N metabolism among the sites sampled. Both the metagenome prediction and induced metabolic activities suggested that microbial communities from the reference site had the highest capacity to metabolise simple carbohydrate. The activities of selected enzymes involved in cellulose, hemicellulose and protein decomposition were not significantly different among the sampling sites.

The TLF soil microbial communities expressed a lower potential capacity of organic matter decomposition, especially for simple carbohydrate (eg. sugar), but the selected enzymes involved in cellulose, hemicellulose and protein decomposition were at a similar level as those from the reference site. As sugar metabolisms is usually associated with opportunistic bacteria that require moist habitats, enhancing the water availability and the accumulation of organic matter with favourable C:N ratios (eg. understorey plant biomass) is critical to enhance the microbial functions and coupled nutrient cycling.

The 2018 and 2019 findings collectively point to the importance to establish productive understorey species (including N_2-fixing leguminous species) to increase labile organic matter (ie. biomass residues and root debris) and N inputs. This is critical to restoring the nutrient pools and maintaining the biological functions in surface soil. Importantly, the increased understorey vegetation would provide shading effects, to help alleviate radiation heat stress and drought stress in the surface soil of the TLF sites in future, which are favourable for soil microbial activities and nutrient cycling in the surface soil.

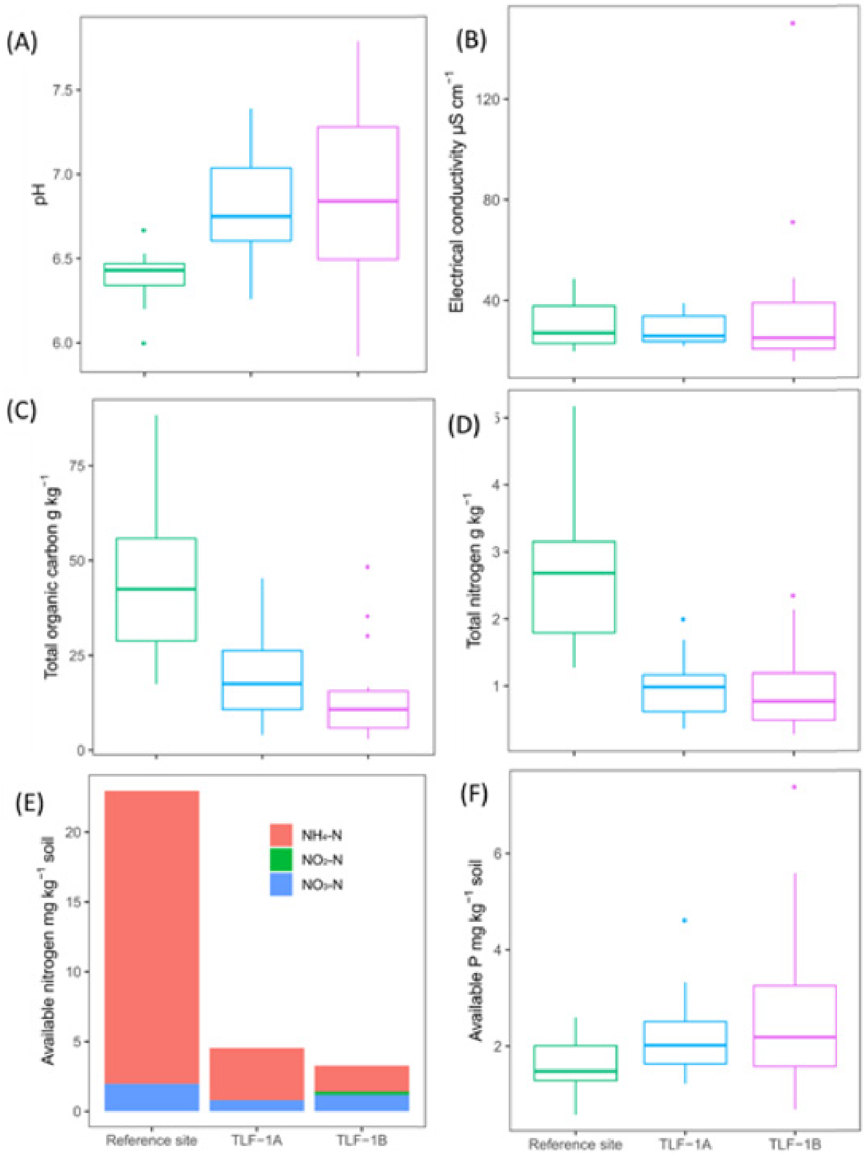


Figure 4-7: Selected soil chemical properties pH (A), EC (B), and nutrient availability, including total organic carbon (C), total nitrogen (D), Available N in the form of NH4+-N, NO2--N and NO3--N (E) and Available P (F) among reference Site 30, TLF-1A and TLF-1B.

In summary, 10 years after the revegetation, the TLF growth media have significantly improved their nutrient level compared to the initial stage of the revegetation and the microbial communities in the surface soils were highly diverse which is similar to the reference site. However, the TLF soil microbial communities expressed a lower potential capacity of organic matter decomposition, especially for simple carbohydrate (eg. sugar), due mainly to relatively dry surface material, and relatively low accumulation of organic matter with favourable C: N ratios (eg. understorey plant biomass). To improve the TLF nutrient status and cycling, it was recommended that the most important strategies were to:

(1) Minimize surface drought and heat;

(2) Enrich high quality organic matter by understorey growth; and

(3) Improve N-supplying capacity by introducing diverse deep-rooting understorey legumes.

4.4 Infiltration, runoff, and erosion

Four erosion plots (approximately 30 m × 30 m) were constructed on the TLF during the 2009 dry season (Saynor *et al.* 2009) (Figure 4-8). The TLF surface was ripped on the contour before the erosion plots were constructed, and plots were located to represent two types of potential final cover layers (waste rock, or waste rock – laterite mix) and planting methods (direct seeding and tube stock).The plots were physically isolated from runoff from the rest of the landform by raised borders.

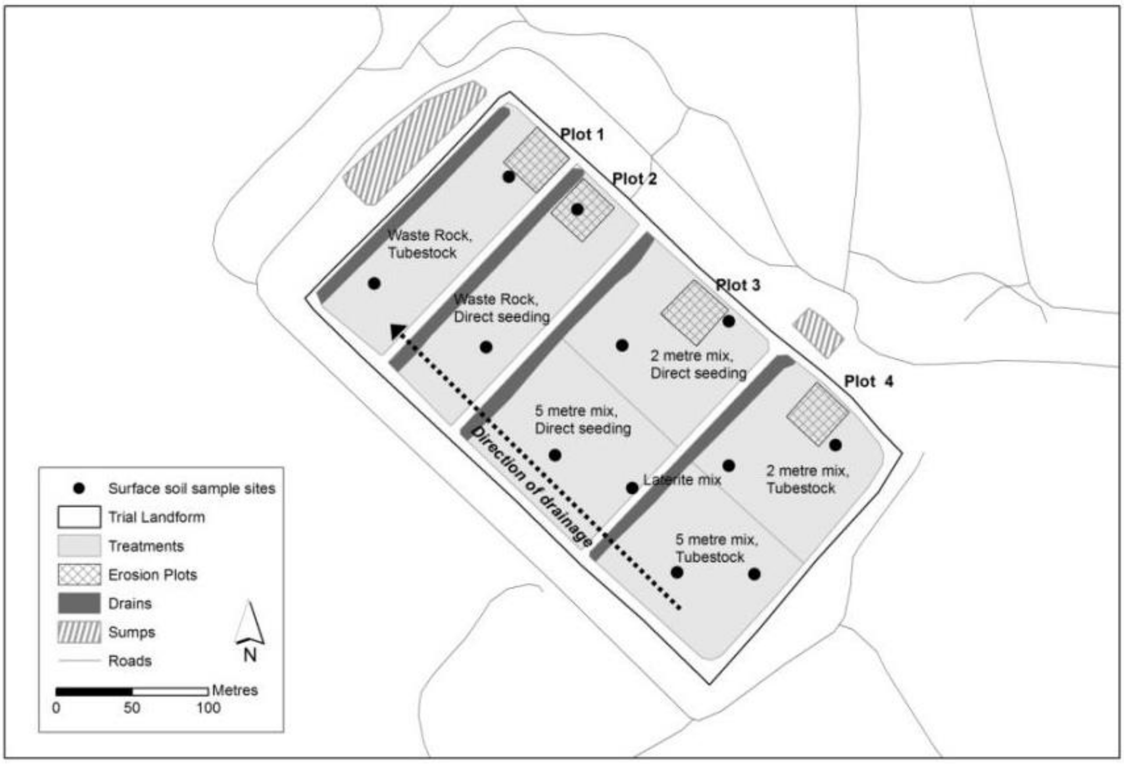


Figure 4-8: Layout of the erosion plots on the trial landform (Boyden *et al.*, 2016, Saynor *et al.*, 2016)

Sensors installed in each plot included: a tipping bucket rain gauge, a primary shaft encoder with a secondary pressure transducer to measure stage height, a turbidity probe to measure suspended sediment concentration, electrical conductivity (EC) probes located at the inlet to the stilling basin and at the entry to the flume to provide a measure of the concentration of dissolved salts in the runoff, an automatic pump sampler to collect event based water samples, a data logger with mobile phone telemetry connection and a rectangular broad-crested flume to accurately determine discharge from the plots (Saynor *et al.* 2014) (Figure 4-9).

Figure 4-9: Runoff through the flume on the trial landform erosion plot 3 during a storm event (Saynor *et al.*, 2014)

Monitoring results including generation and transport of solutes, hydrology and bedload yields, have been reported regularly (Saynor *et al.* 2009, Saynor *et al.* 2011, Saynor *et al.* 2012, Saynor *et al.* 2014, Saynor *et al.* 2015).

4.4.1 Infiltration

Studies have been undertaken involving some field measurements of infiltration and runoff rates of the TLF. In his PhD study into surface hydrological modelling for rehabilitated landforms, Shao (2015) developed a modified runoff model (RunCA) and then applied it to the Ranger Mine TLF as a case study. Good agreement was achieved between the simulated and observed discharge volumes, runoff curves and flow distributions for the rainfall events monitored during four wet seasons from 2009 to 2013. The study utilised the existing SSB erosion plots on the TLF (e.g. Saynor *et al*. 2012b) and carried out additional field infiltration measurements (September 2013) to determine the hydraulic properties of the TLF and the infiltration parameters for the RunCA model.

The following is an excerpt from Shao (2015) and details the field methods used to obtain infiltration measurements on the TLF in September 2013:

> *Due to the large width of the rip lines, four measurements were conducted on the rip lines at randomly selected areas on the waste rock cover, using a ring infiltrometer with a large diameter of 1 m. Another four measurement were also conducted randomly on the non-ripped areas between the rip lines, using a smaller ring infiltrometer with a diameter of 0.4 m. The falling head method was employed in all these measurements. Each measurement lasted until a stable infiltration state was reached, and then the final steady infiltration rate i_f was calculated by averaging the last three measured infiltration rates. Core samples were also taken in the areas immediately adjacent to the infiltration measurements for the laboratory determination of various properties. Specifically, the total porosity TP was assumed to be equal to the saturated water content, which was reached by leaving the core samples in a tray filled with shallow water for 2-4 days, and field capacity θ_{FC} was achieved by leaving the saturated core samples on a suction plate with 33 kPa (0.33 bar) suction pressure for 7 days. Initial soil moisture θ_0, TP and θ_{FC} were then determined by weighing the core samples before and after oven-drying at 105°C for 24 hours in the laboratory.*

Discharge volumes, runoff curves and flow distributions for the rainfall events monitored during four wet seasons from 2009 to 2013 were used to determine the hydraulic properties of the TLF (Shao 2015) (Table 4-5 and Table 4-6) Shao’s direct measurements from the TLF were used to calibrate the WAVES model (Section 4.5).

Table 4-5: Statistical values for the observed rainfall events in the four wet seasons (water years) from 2009 to 2013

Water year [a]	Annual rainfall (mm)	Annual runoff (mm)	Number of events	Event duration (min)		Runoff coefficient (%)	
				Range	Mean	Range	Mean
				Plot 1			
2009-10	1528.1	77.7	68	15-534	113.1±104.2	0.7-14.2	5.6±2.5
2010-11	2205.4	300.2	96	15-631	139.0±140.3	2.6-88.2	6.0±9.1
2011-12	1481.0	101.2	78	16-713	87.5±127.6	2.2-40.3	5.4±4.4
2012-13	1283.0	121.8	62	8-2135	88.1±275.8	1.2-29.9	4.6±4.3
				Plot 2			
2009-10	1531.5	132.0	68	26-543	156.2±114.3	1.1-22.3	8.0±4.0
2010-11	2293.6	328.5	96	31-760	177.5±148.5	3.7-78.2	8.7±7.9
2011-12	1531.4	166.3	78	26-1017	130.2±154.0	2.5-30.9	8.9±5.0
2012-13	1274.2	196.4	62	13-2154	127.8±270.8	2.2-57.9	11.7±9.7

[a] A water year is defined as the period from 1 September to 31 August of the following year

Table 4-6: Summary of field infiltration parameters for the TLF

Measurement No.	Infiltration parameters [a]						*RMSE* (mm h^{-1})	R^2
	i_f (mm h^{-1})	θ_{FC} (m^3 m^{-3})	θ_0 (m^3 m^{-3})	*TP* (m^3 m^{-3})	*a* [b] (mm)	*D* [b] (mm)		
			Rip lines					
1	25.20	0.09	0.07	0.30	0.60	180	7.37	0.84
2	24.00	0.12	0.09	0.26	0.50	90	5.09	0.84
3	18.00	0.11	0.07	0.30	1.30	100	6.79	0.82
4	30.00	0.09	0.08	0.26	2.50	120	7.76	0.95
Mean	24.30	0.10	0.08	0.28	1.23	122.50	6.75	0.86
SD	4.94	0.02	0.01	0.02	0.92	40.31	3.35	0.03
			Non-ripped areas					
5	7.50	0.08	0.06	0.23	0.75	100	9.38	0.83
6	19.20	0.08	0.07	0.23	1.50	150	6.23	0.96
7	12.00	0.06	0.06	0.21	1.50	50	5.00	0.96
8	14.00	0.11	0.07	0.25	1.00	80	7.73	0.85
Mean	13.18	0.08	0.07	0.23	1.19	95.00	7.08	0.90
SD	4.85	0.02	0.01	0.01	0.38	42.03	1.90	0.07

[a] i_f: final steady infiltration rate (mm h^{-1}); θ_0: initial soil moisture (m^3 m^{-3}); θ_{FC}: field capacity (m^3 m^{-3}); *TP*: soil porosity (m^3 m^{-3}); *a*: a constant (mm$^{-0.4}$ h^{-1}) in modified Holtan model; *D*: depth of control zone which affects the infiltration process (mm).

[b] unmeasurable parameters determined by curve-fitting with observed infiltration rates.

4.4.2 Runoff

Annual runoff from the TLF was determined to be the greatest in the wettest year, and there is a close relationship between event rainfall and event runoff over the full range of rainfall for all monitored years.

There is an apparent exponential relationship between event rainfall and event runoff over the full range of rainfall for five years monitoring of plot 1 (Figure 4-10), however due to technical issues with large events this has not yet been tested statistically (Saynor *et al.* 2015). Saynor *et al.* (2015) hypothesised that event rainfall greater than 30 mm generates proportionally greater runoff as smaller events do not totally infill the rip lines with water. Event rainfall greater than 30 mm can totally infill the surface storage, hence generates runoff from the whole plot surface.

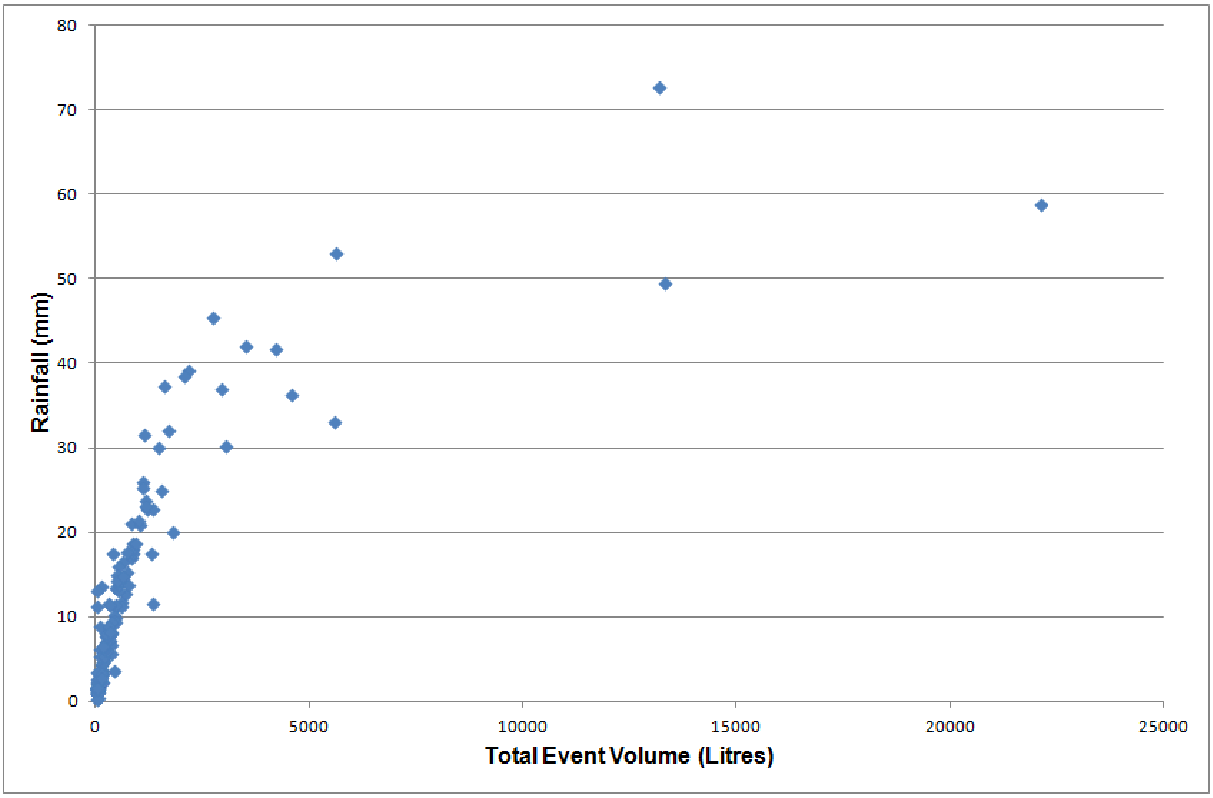


Figure 4-10: Relationship between total event rainfall and runoff for erosion plot 1 for 156 runoff events in the 2013–14 wet season (Saynor *et al.* 2015)

4.4.3 Erosion

Run-off and erosion rates measured on the trial landform have been used to assess the long-term geomorphic stability of the trial landform and have been applied by extension to the final landform (comparing measured export rates with those modelled from the landform evolution model).

Bedload samples were collected at weekly to monthly intervals during each wet season, depending on the magnitude of runoff events and staff availability. In general, sediment yields for major land disturbances, such as construction or landslides, are characterised by an initial pulse followed by a rapid decline (Duggan 1994 cited in Saynor *et al.* 2015). This is true for the trial landform annual bedload yield, which is characterised by an exponential decline since construction (Figure 4-11). Saynor *et al.* (2015) also noted that since construction, eroded material has been washed into the rip lines, but there is still a large amount of potential sediment storage before the rip lines are diminished. Fine materials and fines earth accumulated in the rip lines and other depressions are important for the soil formation on the final waste rock landform and sustainability of the revegetation.

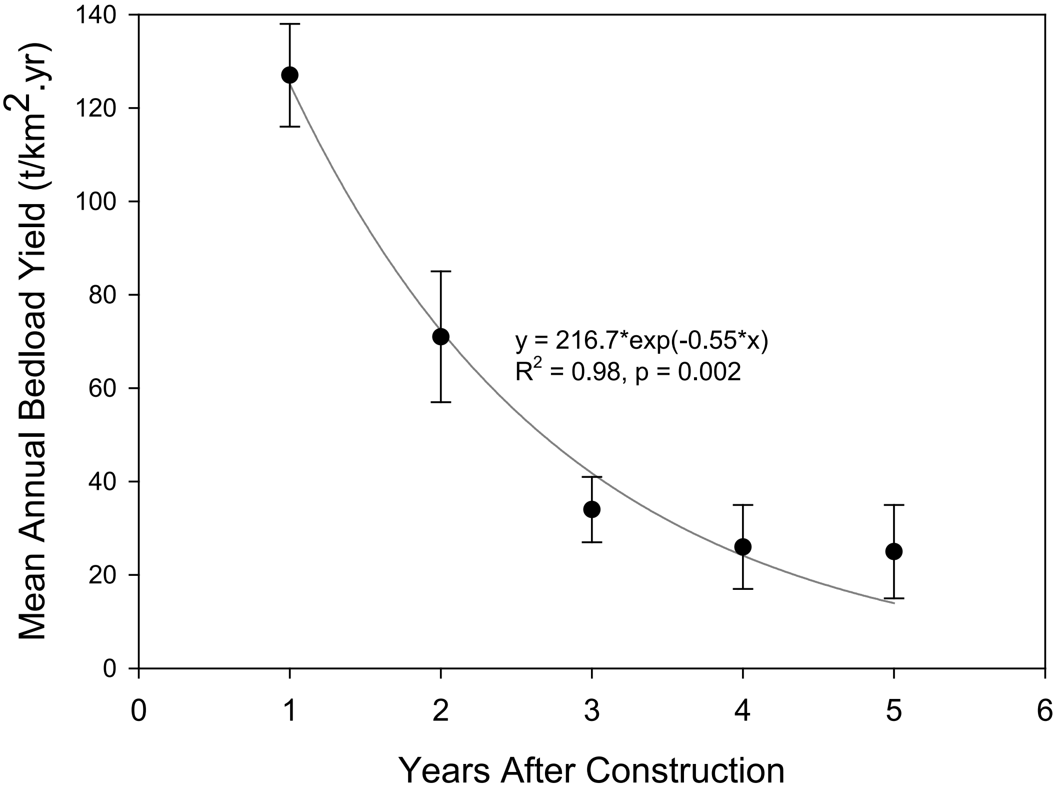


Figure 4-11: Exponential decrease in mean annual bedload yield with time since construction for the four plots on the trial landform. Data represent annual mean and standard error of estimate for all plots (Lowry & Saynor, 2015)

4.5 Plant available water (PAW) studies

Ranger Mine is located in the seasonally wet-dry tropics of northern Australia, where ~95 % of rainfall occurs between November and April (Section 4). In this tropical region, the most important factor shaping the landscape and determining the type of savanna ecosystems is the soil water availability and whether vegetation can survive the half yearly dry season. This presents the most critical challenge for Ranger Mine site revegetation as post-mining soils often lack structure or contain large amounts of rock fragments that reduce their water holding capacity.

To address this critical question of whether the waste rock substrate of the Ranger Mine final landform can supply sufficient plant available water (PAW) to sustain a local native woodland, ERA has undertaken extensive research over the past three decades, especially in the last two decades (Hollingsworth 2010, Lu 2017, Lu *et al*. 2019). ERA has undertaken long-term ecohydrological studies in the Georgetown Creek Reference Ecosystem area since 2008 (MCP Section 5.3.3.5) and studied soil water dynamics and vegetation performance on the Ranger Mine TLF since 2009.

Since 2011, ERA has engaged Charles Darwin University to undertake a modelling approach to study the water balance of the TLF. The study used hydrologic characteristics of the waste rock substrate and the outcomes of the above ecohydrological studies to model the water balance using the Commonwealth Scientific and Industrial Research Organisation (CSIRO) WAVES model (Zhang & Dawes 1998). This modelling focussed on estimating the required PAW in the waste rock surface layer to meet the anticipated demand for sustaining the rehabilitated ecosystem.

PAW is the amount of available water that can be stored in soil and be available for growing plants (within the rooting zone). Water availability on the waste rock final landform cover is going to be a challenge for the Ranger Mine ecosystem re-establishment as waste rock growth media often lack structure or contain large amounts of rock fragments and macropores that reduce their water holding capacity (compared to natural soils).

A range of ecohydrological research and modelling has been undertaken at the Ranger Mine to support the intention to use waste rock to construct the final landform and establish a range of sustainable vegetation communities similar to those in Kakadu National Park.

4.5.1 Volumetric soil moisture content

After construction of the TLF in 2009, a pit was dug to the natural ground level by an excavator to allow vertical installation of soil moisture probes to integrate a measure over the 0.3 m length of the probe at 0 to 0.3 m, 0.3 m to 0.6 m, 0.6 m to 0.9 m, 0.9 m to 1.2 m, 1.2 m to 1.5 m, 2.7 m to 3.0 m, and 3.7 m to 4 m below ground surface in the TLF 1A section. For other sections, additional 1 probe per metre was added in depth until reaching the nature ground surface. Another four probes were installed horizontally at 0.1 m below ground surface to monitor shallow soil moisture.

Soil volumetric water content at different depths in the waste rock only substrate in the TLF 1A section over a two year period are shown in Figure 4-12 and Figure 4-13. Soil volumetric water content at the TLF 1A section show significant seasonal variations. The entire soil profile is recharged with rainfall water during the wet season and gradually dries out during the dry season. The landform substrate acts as a ‘store and release’ reservoir for the establishment and development of vegetation.

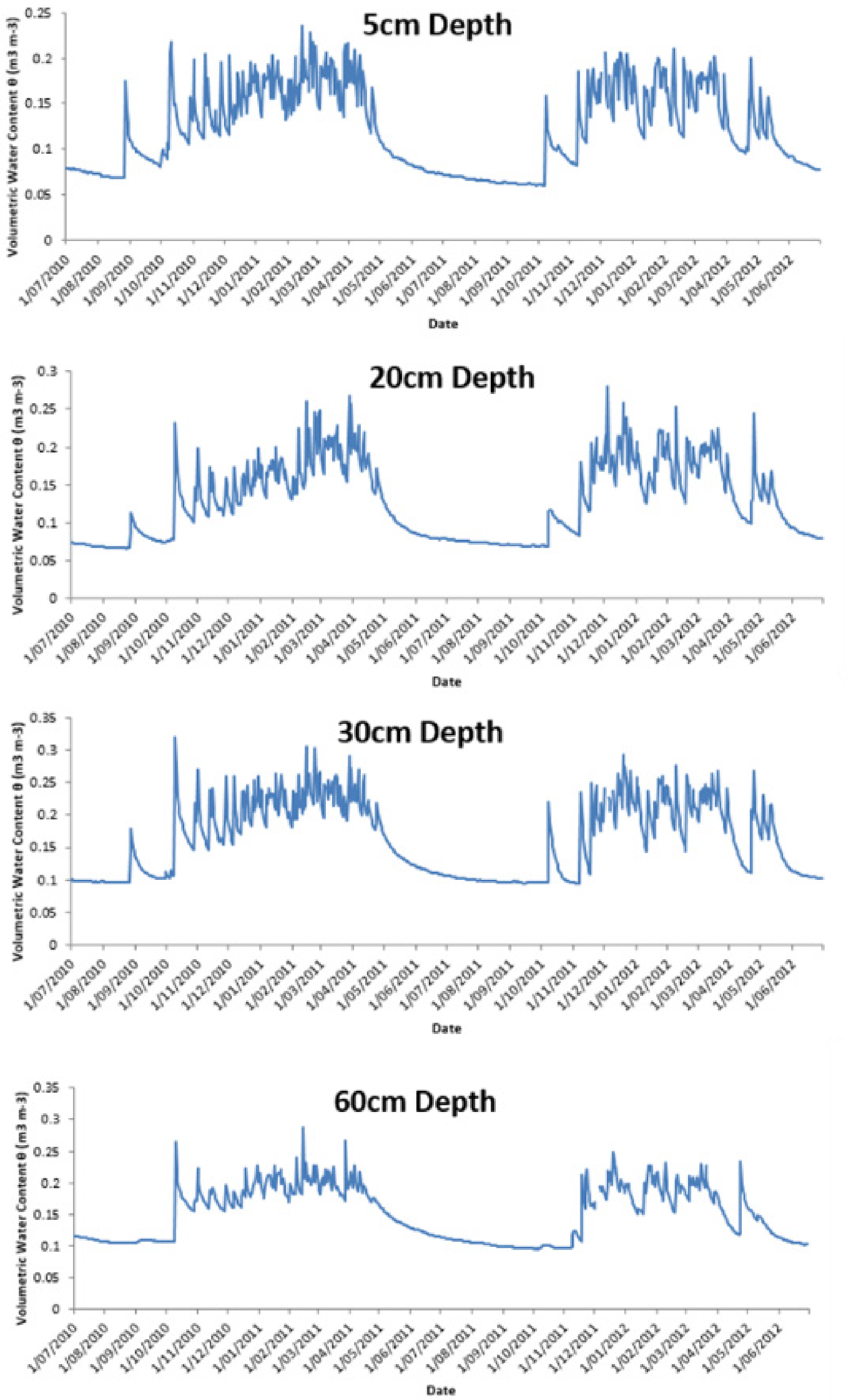


Figure 4-12: Seasonal dynamics in soil volumetric water content at depths 5 to 60 cm in the waste rock only substrate in the TLF 1A section

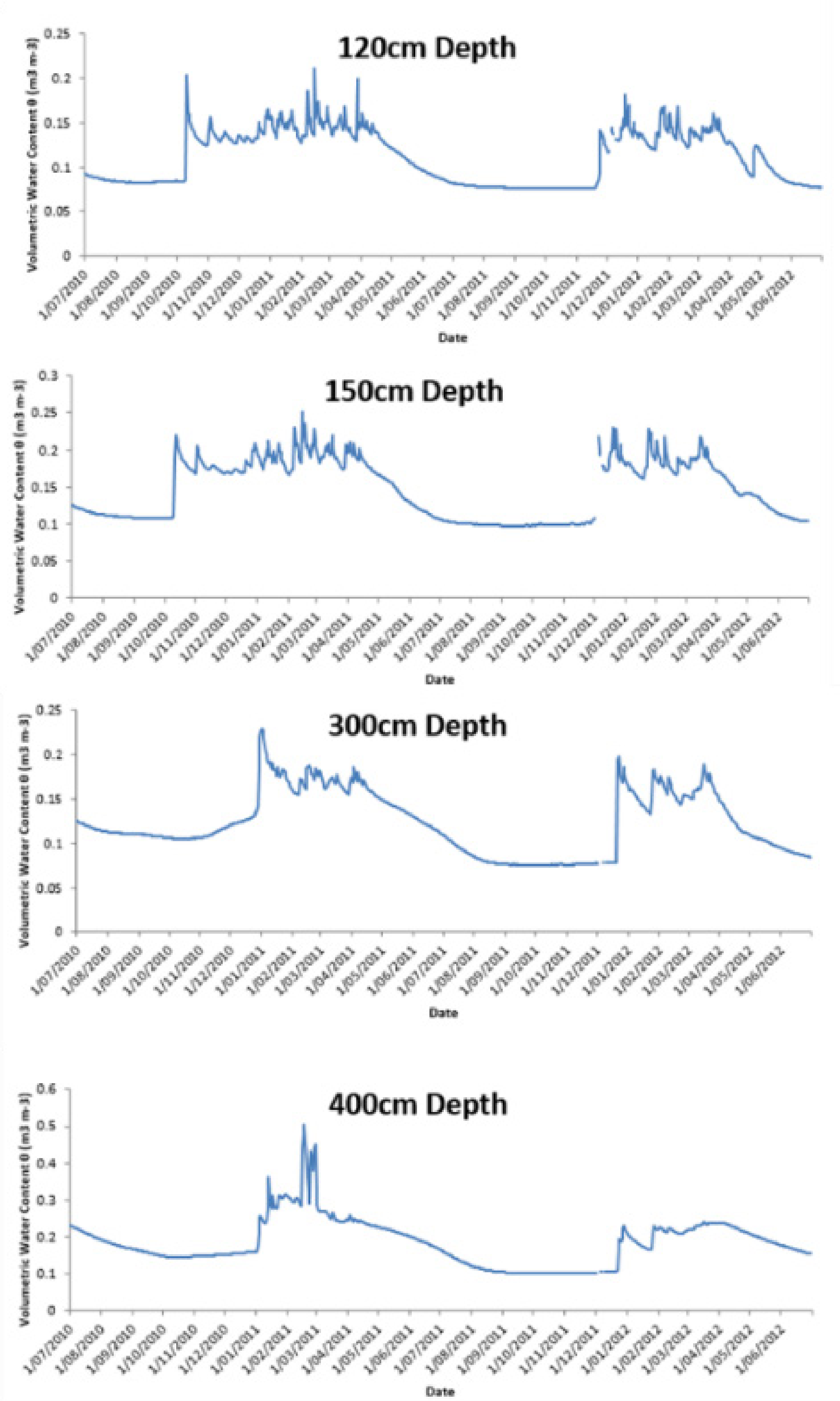


Figure 4-13: Seasonal dynamics in soil volumetric water content at depths 120 to 400 cm in the waste rock only substrate in the TLF 1A section

Figure 4-14 shows long-term dynamics of the soil water contents in the above soil profile from immediately after landform construction until 5 years after. Presumably as a result of the consolidation and improved sensor/substrate contact over time the peaks during the wet season became substantially reduced.

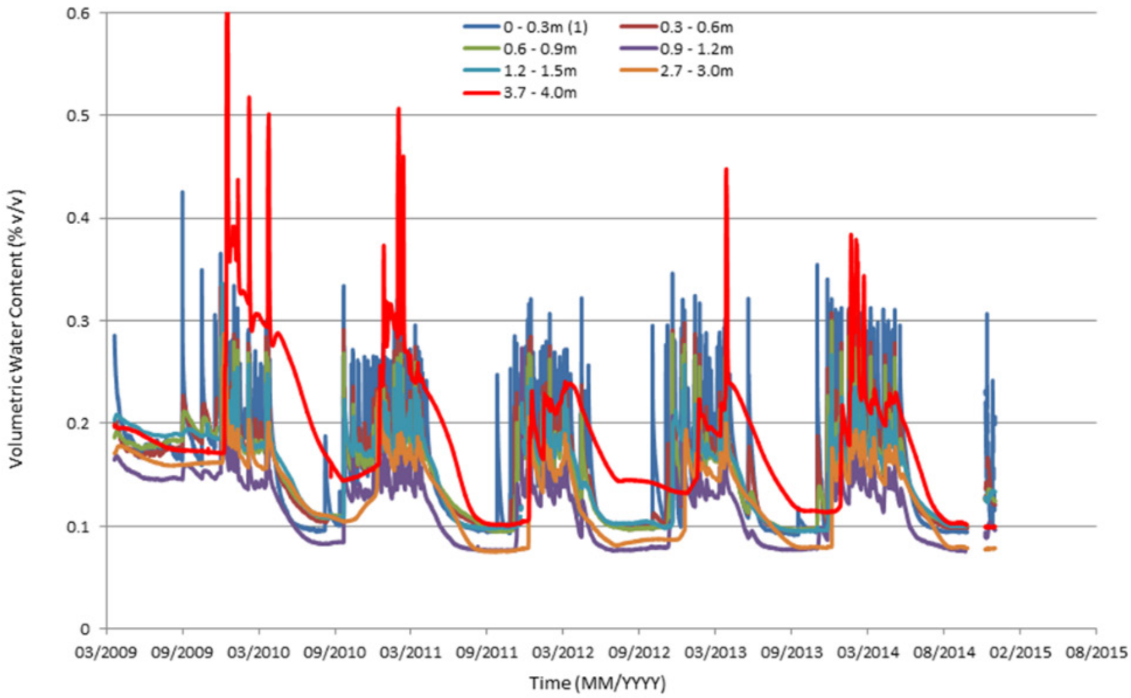


Figure 4-14: Long-term dynamics of the soil water contents in the TLF 1A soil profile from immediately after landform construction until 5 years later

Average volumetric water content for depths 0.3 m to 1.5 m and 2.7 m to 4.0 m are shown in Figure 4-15. Maximum water contents are about 0.25 (25 %) indicating that the saturated void-space is about 25%. Estimated field capacity (green-coloured line) and wilting point (mauve-coloured line) by Croton (2017) are also plotted on the graph in Figure 4-15. The average estimated field capacity is in good agreement with the troughs of the wet-season curve, and the dry-season minima are aligned well with the wilting point.

![ERA logo]

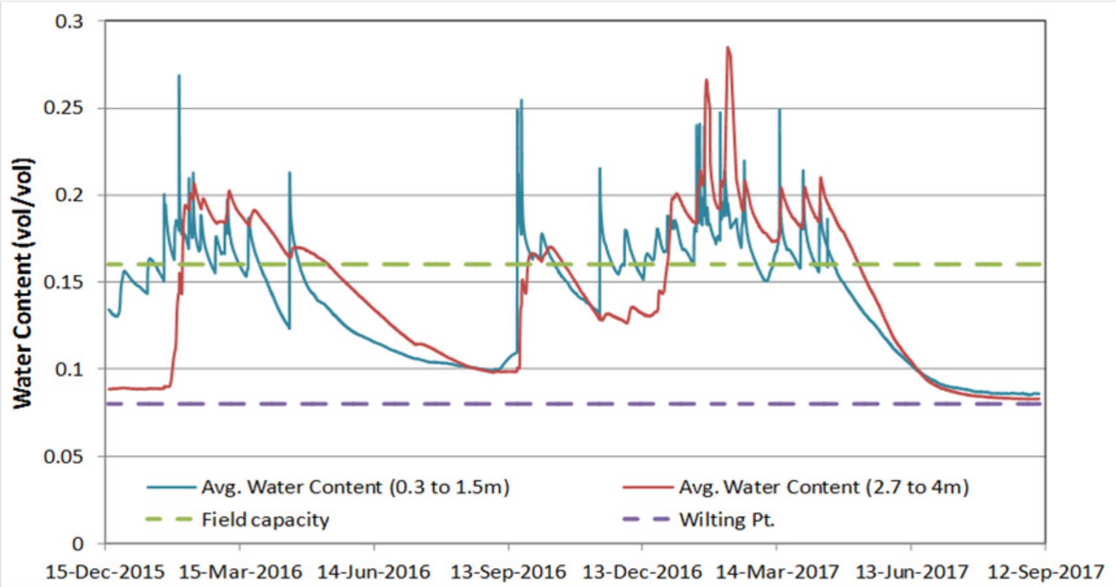


Figure 4-15: Measured volumetric water content and estimated average field capacity and permanent wilting point for TLF growth substrate

The wetting front progression (as shown by soil volumetric water content dynamics) in the in the TLF 1A section is shown in Figure 4-16. The behaviour of wetting front progress after a significant rainfall (47.8 mm) on 24 January 2016 demonstrates a steady downward progression of the wetting front without abrupt peak at lower positions. This suggests that preferential pathways are not a major issue in the TLF 1A section.

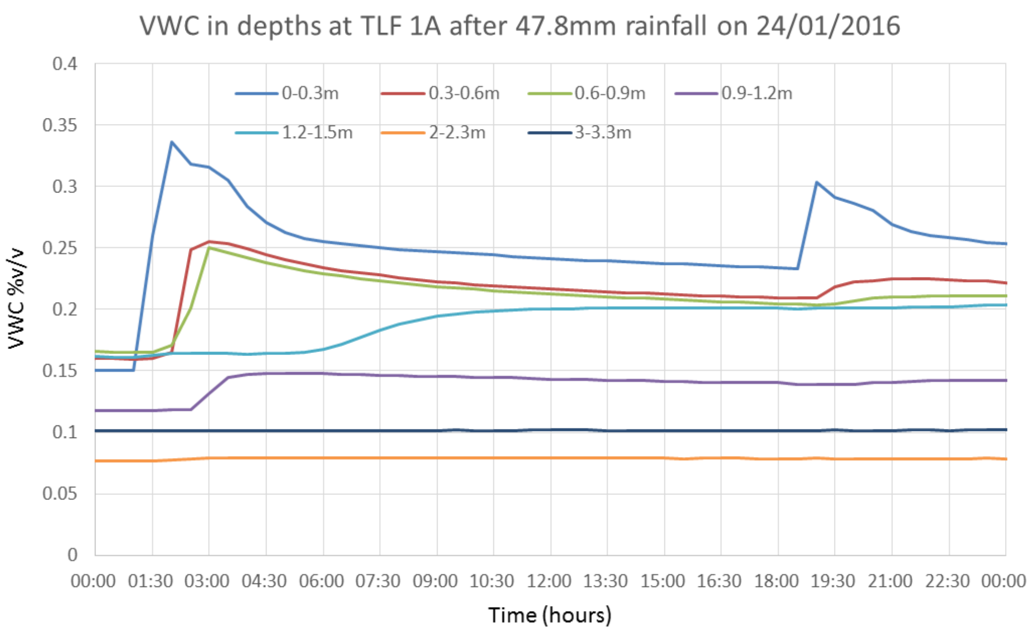
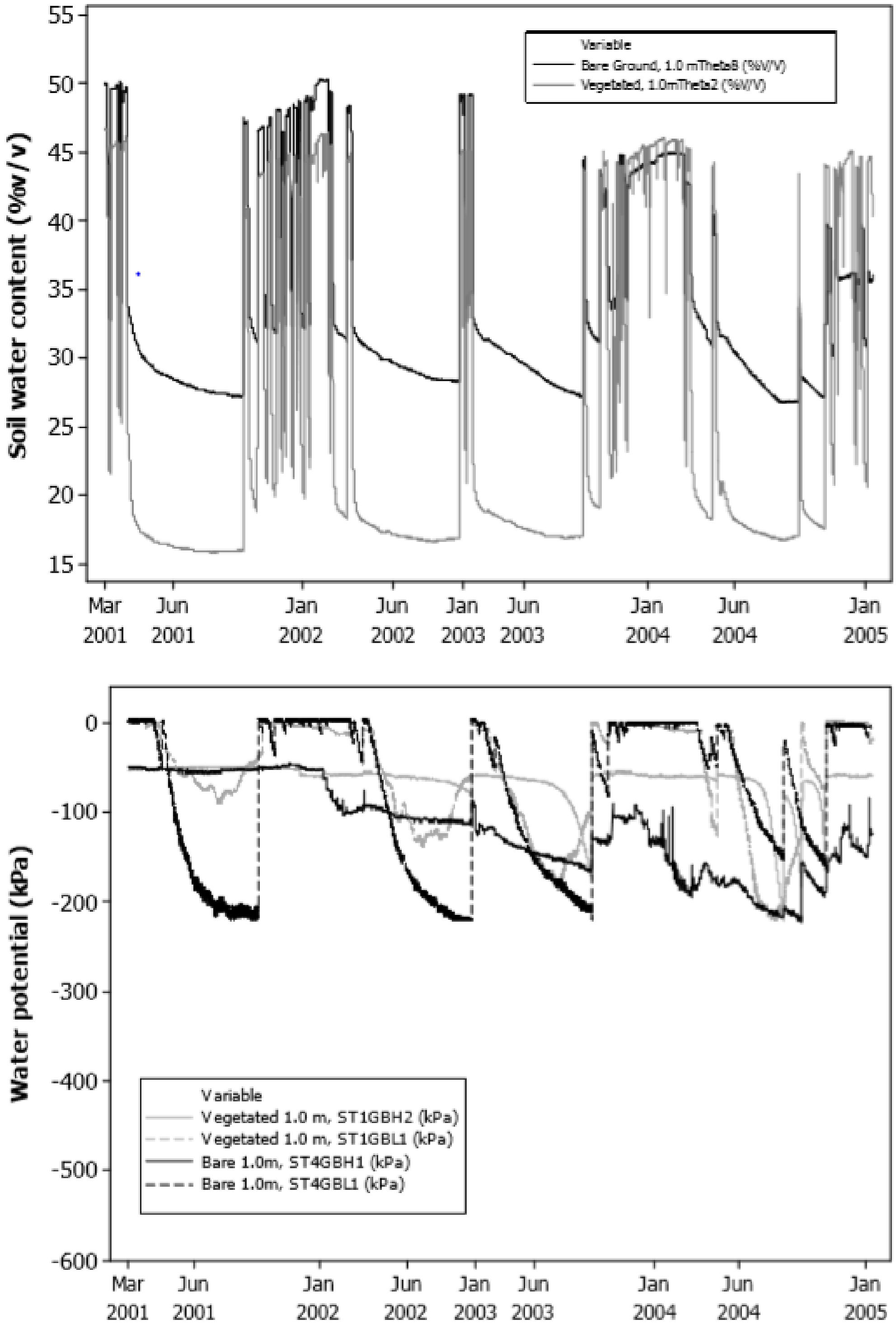


Figure 4-16: Wetting front progression as shown by soil volumetric water content dynamics in the waste rock only substrate in the TLF 1A section

In the study on an experimental waste rock cover established on the Ranger Mine waste rock pile, Hollingsworth (2010) monitored the soil water content and water potential in the cover (0.5 m – 1.0 m) (Figure 4-17). The measured soil water content and water potential in the waste rock cover were higher or similar to that observed on the TLF (Figure 4-13). Hollingsworth (2010) reported the saturated water contents range from 29.5 to 46.0 % v/v and from the above curves, the field capacity is at least 18 %, and the residual water content was 3.2 to 3.5 %. This suggests that the PAW is at least 15 % which is comparable with, albeit higher than the PAW of about 10 % found at the TLF.



(Source: Hollingsworth 2010)

Figure 4-17: Soil water content and water potential monitoring for Horizon 2 (0.5 to 1.0 m) in an experimental waste rock cover established on the Ranger Mine waste rock pile

4.5.2 Rooting depth in the waste rock landform

To estimate the total PAW in the landform's growth media, it is necessary to know the rooting depth of the revegetation in the waste rock substrate.

In March 2019, ten years after the initial revegetation of the TLF, one observation pit in each of the section 1A and 1B of the TLF (Figure 3-3, the two 2019 pits located furthest from the erosion plots) were excavated to assess root distribution throughout the waste rock soil profile. Pit 1A was excavated less than 0.5 m away from a large *Eucalyptus tetrodonta* tree (9 m high) to approximately 3.5 m deep (which was 0.5 m from the bottom of the landform); Pit 1B was less than 0.5 m away from a large *Eucalyptus phoenicea* tree (8 m high) and excavated to approximately 4 m deep (about 0.5m from the bottom of the landform). Bulk samples (each of ca. 4kg) were collected both at surface and different depths (Table 4-7).

Roots were separated from the waste rock by dry picking and wet sieving. The waste rock was also separated during the process into large, medium and fine fragments (>5 mm, 2 – 5mm, and <2mm, respectively). The separated materials were then oven-dried at 105°C. Surface roots were mostly observed in the top 1 m of the soil for both pits, whilst the tap roots were still visible at approximately 2.5 m depth in pit 1A and 2.0 m depth in pit 1B (Figure 4-18).

Table 4-7: Dry weight percentage of waste rock and roots in pit 1A and pit 1B of the TLF

		Dry Weight Percentage (%)			
Area	**Depth (m)**	*Large WR*	*Medium WR*	*Fine WR*	*Roots*
1A	Surface	37.180	19.312	43.465	0.043
1A	0.5	52.751	20.581	26.633	0.034
1A	1	66.359	18.555	14.586	0.500
1A	2	55.910	18.845	25.222	0.023
1A	3.5	64.316	17.632	18.051	0.001
1B	Surface	26.779	24.935	48.190	0.095
1B	0.5	67.342	13.134	19.400	0.124
1B	1.5	48.826	21.016	30.035	0.123
1B	2.5	60.838	17.899	21.259	0.004
1B	4	66.087	15.345	18.563	0.005

WR = waste rock

Pit observation and root mass measurements demonstrated that root matter was present in all samples at all depths (Figure 4-18), which indicates that large trees can root down to at least 3.5 m depth in pit 1A and down to 4.0 m in pit 1B of the TLF. This is consistent with the visual observation of the pit walls (Figure 4-19). This was the first time that roots were excavated to the 4 m depth in the waste rock landform at the Ranger Mine and provides direct evidence that local native woodland tree species roots can reach depth beyond 1 to 2 m. Although the root biomass significantly decreased with depth, those small amounts of fines roots are critical for the survival of the trees through the late dry season.

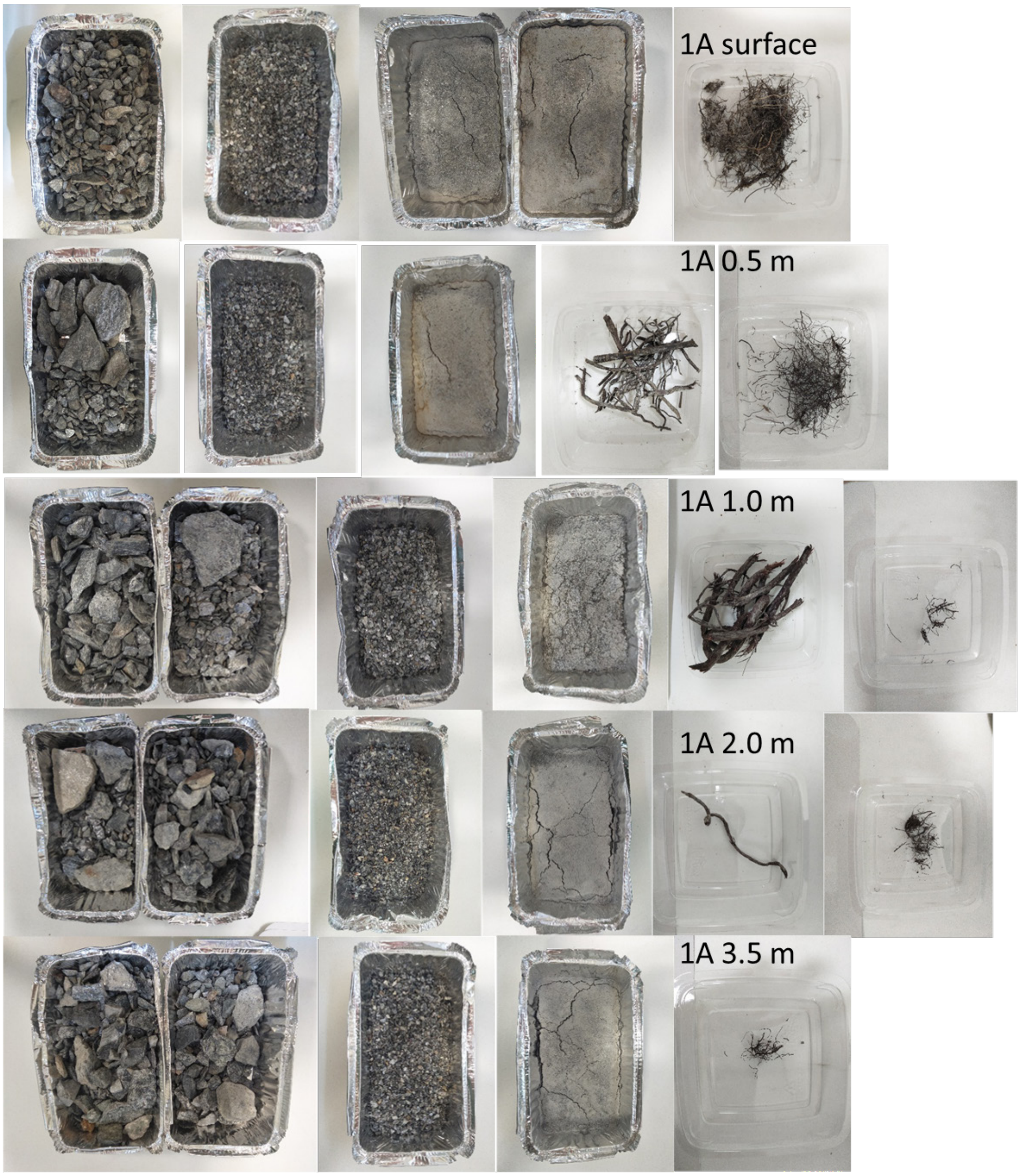


Figure 4-18: A visual comparison of the root mass and waste rock materials from a bulk sample taken at different depths in the pit 1A of the TLF

Figure 4-19: Presence of roots of 1-2 mm in diameter visible at 3.0 m deep in the Pit 1A and at 3.7 m deep in the pit 1B of the TLF

In addition, Humphrey *et al*. (2009) reported that ERA and ERISS have opportunistically examined the depth of penetration of roots of excavated trees, growing in media that includes waste rock, waste rock and fines, or various laterite/waste rock mixes, and in the natural bush soil. They stated that while some roots were observed at depths of 2.1 and 2.5 m in mine-derived and natural soils, respectively, the main root ball of trees, comprising an estimated >95% of the root biomass, is invariably contained in the top 0.7 m of the soil profile. Figure 4-20 is a photograph showing the root ball of 7.7 m high, 12 year old *Eucalyptus glomericassis* excavated from a trial rehabilitation site on the eastern edge of the Ranger tailings dam (from the so-called 'Heritage' site). While shallow lateral roots have been broken off, the main primary (tap) root is relatively intact.

A trench cut in a revegetated Ranger waste rock pile with 4 year-old trees grown from seed showed obvious roots to 0.8m with some evidence of roots at the bottom of the trench (1.6 m) (Emerson & Hignett 1986).

(Source: Humphrey *et al*. 2009)

Figure 4-20: Root ball of a *Eucalyptus glomericassis* excavated from a trial rehabilitation site

4.5.3 Estimated and actual potential plant available water

The SPAW Hydraulic Properties Calculator (a pedotransfer calculator) (Saxton & Rawls 2006) was used to develop the estimated soil water retention curve (volumetric water content vs. matric potential) and the volumetric water content at permanent wilting point and field capacity for the waste rock substrate of the TLF (Segura 2017). The potential PAW for the <2 mm fraction of the profile (PAWp) was calculated by subtracting volumetric water content at permanent wilting point from the field capacity (θ_{fc}- θ_{pwp}) and then multiplying this by the matric fraction of the soil that has the ability to store water. The >2mm fraction of the media is considered to be rock and deemed to be unable to store PAW. The proportion of the rock in the TLF 1A profile is about 67 %v/v, so the fraction of soil is 33 % v/v. (Table 4-8).

Approximately 400 mm of PAW can be potentially held in the substrate of the TLF of a thickness of four metres, corresponding to a 10 % v/v water content. This is the same as the 400 mm identified by Hollingsworth (2016) for 'plant available soil water content between 0 to

8 m depth' (Table 4-8). Whilst the 90 mm of PAW in the top one metre of the waste rock plant growth substrate is more than the required 60 mm identified by Hollingsworth (2016).

The estimated 400 mm PAW is the potential PAW, i.e. assuming the soil profile is filled with that amount of water at the end of the wet season. However, the actual PAW might be less than that amount, depending on the rainfall distribution, especially the last rainfalls in the wet season.

Table 4-8: Potential PAW in the layers of the growth substrate of the TLF section 1A 4 m profile

Nominal Depth (actual depth) (mm) of the layer	Layer Thickness (mm)	Gravel Content %	Potential PAW (mm)
0 (0-500)	500	66	51.90 ± 0.10
1000 (500-1500)	1000	68	97.92 ± 0.32
2000 (1500-2500)	1000	64	109.91 ± 0.21
3000 (2500-3500)	1000	73	82.35 ± 0.00
4000 (3500-4000)	500	62	57.95 ± 00
Total	4000		400 ± 0.13

(Source: Segura 2017)

Observed plant available water (PAW_{obs}) at the TLF section 1A during seven consecutive dry seasons, over the period 2010 to 2016, is presented in Table 4-9. Additional PAW from dry season rainfall has been added to the end of wet season maximum PAW_{obs}. Wet season rain is the total rainfall prior to the studied dry season (based on Segura (2017)). The average actual total PAW stored in the four-metre thick TLF section 1A is 261 mm which is significantly less than the 400 mm which is potentially storable in the four-metre thick waste rock landform (Table 4-8).

Table 4-9: Observed plant available water at the TLF for seven consecutive dry seasons

Dry Season Start	Dry Season End	Duration (d)	Wet Season Rainfall (mm)	Maximum PAWobs (mm)	Additional PAW (mm)	Total PAW (mm)
23/04/10	9/10/10	169	1490	247	13	260
19/04/11	8/11/11	203	2275	231	20	251
27/03/12	10/11/12	228	1318	236	88	324
13/04/13	02/11/13	203	1087	224	30	254
10/05/14	05/11/14*	179	1857	228	17	245
14/04/15	28/11/15**	228	988	217	24	241
21/04/16	19/09/16	151	856	225	30	255
					Average	261
				Average PAW per metre		65.4

*Date based on rainfall, no θ data available for PAW calculation on that day, data missing

**Last PAW value before a data gap, supported by rainfall

(Source: Segura 2017)

Six years of PAW_{obs} is a very limited period for assessing whether the actual PAW will be sufficient to meet the evapotranspiration requirements of the reference vegetation, given the natural variability in weather conditions that the natural vegetation experiences historically. Therefore, a risk assessment was undertaken to simulate the historical actual PAW and evapotranspiration using the past 117 years weather data (1900 to 2016). A modelling approach (WAVES model) was employed to achieve this objective as detailed below.

4.5.4 Modelled actual plant available water

4.5.4.1 WAVES Model

In collaboration with Charles Darwin University, ERA engaged a PhD candidate to undertake PAW studies utilising the 'WAVES Model' on the Ranger Mine TLF. WAVES (Water Atmosphere Vegetation Energy and Solutes) is a coupled water and carbon ecohydrological model that predicts dynamic interactions within the soil-vegetation-atmosphere system at a daily time step (Dawes & Hatton 1993, Zhang & Dawes 1998). In WAVES, soil water movement in both the unsaturated and saturated zones is simulated using a fully finite difference numerical solution of the Richards equation (Berry *et al*. 2005). Modelling of the unsaturated zone using the Richards equation allows water movement in the soil profile to be modelled under dry conditions. For each soil type, an analytical soil model proposed by Broadbridge and White (1988) was employed to describe the relationships between water potential, volumetric water content and hydraulic conductivity. Evapotranspiration was estimated by the Penman-Monteith approach (Monteith & Unsworth 2008). Leaf stomatal conductance was calculated by the equation developed by Ball and Leuning (Ball *et al*. 1987, Leuning 1995), which was scaled to canopy scales using the method proposed by Sellers *et al*. (1992). The micrometeorological feedback of the sensitivity of transpiration to a marginal change in stomatal conductance at the stand level is regulated by a dimensionless decoupling coefficient proposed by McNaughton and Jarvis (1991). The rate of plant growth in the presence of different availabilities of light, water and nutrients was estimated by the integrated rated methodology (IRM) of Wu *et al*. (1994), which is an empirical model without resolving the details of chemical and mechanical controls on photosynthesis. Water is extracted for transpiration by roots, which is distributed along the root profile according to root density distribution and water availability in each soil node (Ritchie *et al*. 1986). The WAVES model is able to simulate plant physiology, which allows changes in environmental factors (temperature, solar radiation, rainfall) to impact water use by vegetation and recharge (Chen *et al*. 2014).

WAVES predicts the dynamic interactions and feedbacks between these processes. Thus, the model is well suited to investigations of hydrological and ecological responses to changes in land management and climatic variation. WAVES emphasises the physical aspects of soil water fluxes and physiological control of water loss through transpiration. It can be used to simulate the hydrological and ecological effects of scenario vegetation management options (e.g. for recharge control), or the water balance implications of changed climatic conditions. A more detailed modelling strategy and description of WAVES is provided in Dawes *et al*. (1998), and Zhang and Dawes (1998).

4.5.4.2 Modelled scenarios and modelling approach

The WAVES model was used to assess the water balance of the TLF and the proposed Ranger Mine final landform under the supervision of Professor Lindsay Hutley of CDU. The focus of this investigation was to determine whether the waste rock substrate of the TLF would be suitable for supporting tropical savanna similar to that of the Georgetown Creek Reference Area, specifically Site 21 (conservative, high ET scenario) and Site 30 (low ET scenario).

Site 21 is typical of one of the vegetation types found in the region and represents high evapotranspiration (ET) and high leaf area index (LAI) and thus is a useful **conservative case** for estimating vegetation water demand (Baumgartl *et al*. 2018).

Site 30 is a less dense woodland with a lower ET and lower LAI, representing the variation in vegetation types that occur in landscapes similar to that predicted for the final landform. At Site 30 the estimated average dry season overstorey transpiration is 0.25 mm/day compared to 0.50 mm/day at Site 21.

Soil water balance inputs include rainfall and run-on, and the outputs are evapotranspiration (soil evaporation and vegetation transpiration), runoff and drainage. Evapotranspiration and drainage are linked to the water holding capacity of the soil (which is a product of its texture/composition) whilst runoff is linked to landform slope, soil saturation and surface conditions.

This study used a variety of methods to measure, calculate or predict the different components of the soil water balance on the TLF (with the exception of run-on which was not applicable in the situation studied).

This study first calibrated the soil component of the WAVES model using the measured soil water contents in the four-metre soil profile at the TLF (Figure 4-21). Potential plant available water (PAW_p) in each layer of the substrate was calculated. Then annual dry-season PAW in the landform was simulated ($PAW_{Predicted}$) using the actual weather-rainfall data of the historical weather records over 117 years. Observed PAW (PAW_{Obs}) scaled from observations of TLF water content dynamics (running monthly mean) and predicted PAW ($PAW_{Predicted}$) dynamics on the TLF over multiple wet-dry cycles using century-scale rainfall are presented in Figure 4-21 and show good agreement.

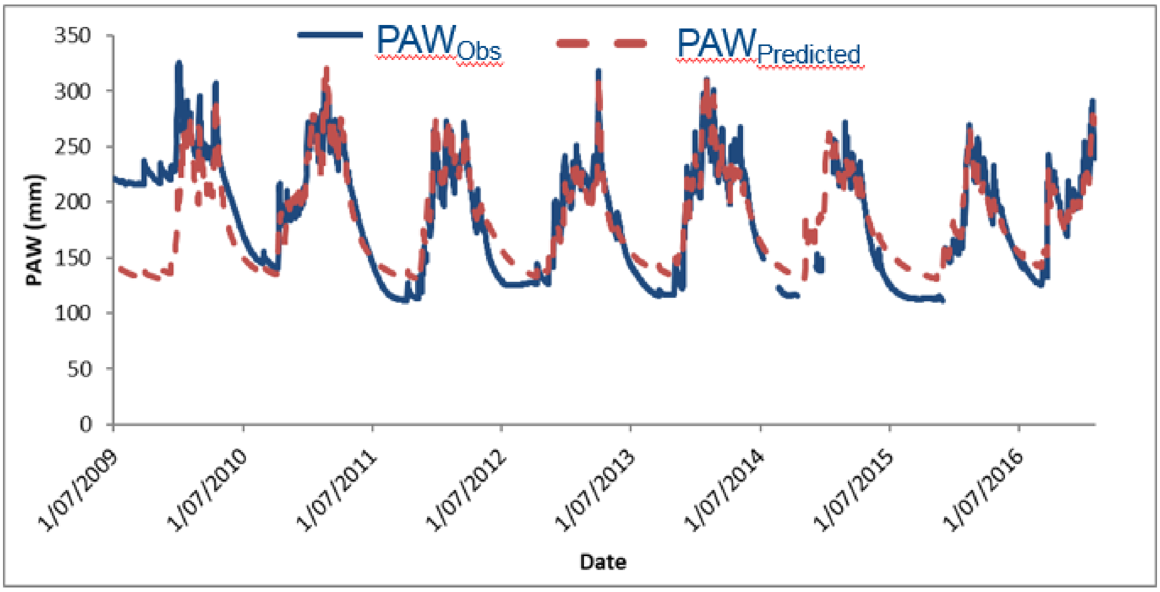
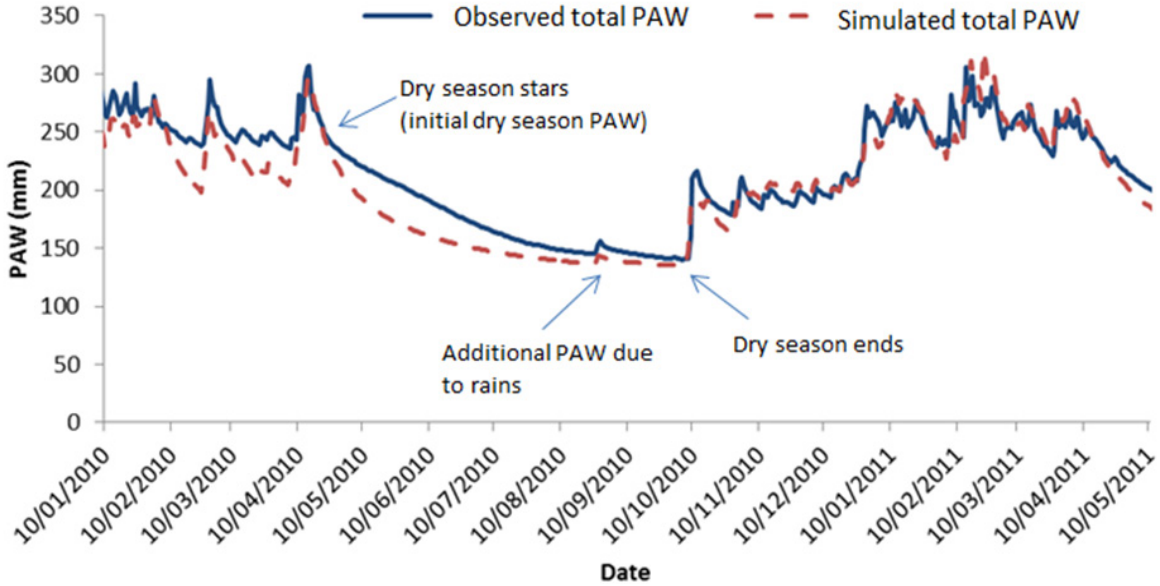


Figure 4-21: PAW_{obs} scaled from observations of TLF water content dynamics and predicted PAW dynamics on the TLF

The soil water balance is assessed by comparing the simulated annual dry-season landform PAWs to the simulated dry season evapotranspiration of the reference sites. Dry season PAW and dry season length were determined from the simulated landform PAW (Figure 4-22). Dry season length was determined based on the assumption that the start of dry season is when consistent decline in soil water starts, and the end of dry season is when PAW is consistently increasing. The simulated and observed PAW values (Figure 4-22) were noted to be well aligned.



(Source: Modified from Segura 2016)

Figure 4-22: Simulated landform PAW versus observed PAW in the trial landform

4.5.4.3 Annual water deficit risk assessment over historic 117 years

Annual dry season PAW over 117 years was simulated using the calibrated WAVES model and compared to the estimated dry season evapotranspiration of Site 21 and Site 30 to derive a PAW balance (deficit or surplus). The PAW balance (within the four-metre waste TLF growth substrate layer) at the end of each dry season was calculated as follows:

> PAW balance = PAW - (measured dry season overstorey transpiration + simulated understorey transpiration and soil evaporation).

The net PAW balance within the four-metre waste TLF growth substrate layer at the end of each dry season, over 117 years, is shown in Figure 4-23. Site 30 (represented by blue bars in Figure 4-23) has a low canopy density and Site 21 (represented by red bars in Figure 4-23) represents the “conservative scenario” with a higher canopy density.

The data in Figure 4-23 (red bars, Site 21) show that the four-metre thick TLF growth substrate layer would have held sufficient PAW for each of the 117 years, except for the year 1915 with a deficit of 8 mm. There is a simulated mean net positive PAW balance of 54.4mm for the 117 years (Lu *et al.* 2019) which suggests that a four-metre thick waste rock cover similar to that of the TLF would be able to supply sufficient water to sustain mature native woodland that is similar to that at Site 21.

It might seem to be concerning that the simulation has shown the TLF PAW status to be close to a deficit in a number of years, and in one year recorded an 8mm deficit, and also considering the level of uncertainty for these predictions. However, one shall remember that this deficit situation has only been predicted for when the simulation used a ‘conservative’ scenario, i.e. by using Site 21 with a high ET. When the model uses a vegetation water use or ET of a site dominated by deciduous species (Site 30) the net PAW balance is much more favourable (Figure 4-23, blue bars; Table 4-10). Meanwhile, it must be considered that a slight deficit over a couple of years may not necessarily result in a vegetation collapse, rather vegetation would most likely increase deciduousness and under more severe and long-term drought, decrease stem density via, for example, self-thinning. The data in Figure 4-23 also shows a general trend of increased surplus over the last century, which is mainly due to increased rainfall in the region.

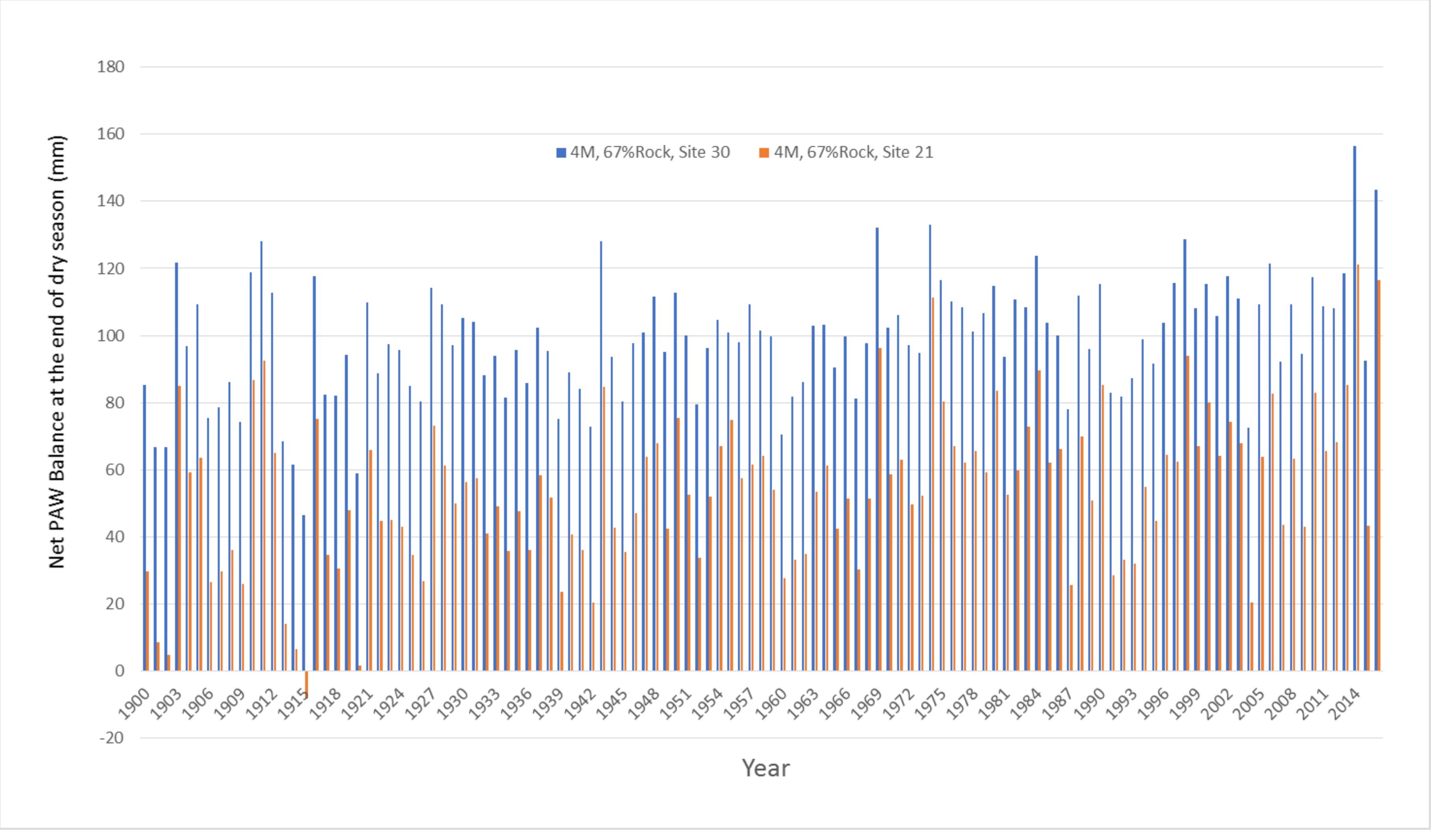


Figure 4-23: Net PAW balance within four metre waste TLF growth substrate at the end of each dry season over 117 years

4.5.4.4 Model uncertainty analysis

Uncertainty of the modelled outcome with regard to occurrence of water deficit depends on mainly three factors:

- Fines % of the growth medium (ie. Potential water holding capacity);
- Growth media thickness (assuming it is also accessible by root system);
- Type of vegetation supported by the growth media; and
- Weather conditions.

The above 117 years PAW balance simulation was based on the worst case scenario where a given area of land of four-meter waste rock growth substrate layer sits on top of a crest where it does not receive run-on (Figure 4-24), and it was assumed that beneath the 4 m depth the root could not access due to either an impermeable layer exist or that roots biologically could not extend below 4 m. In the final landform design, at the crest and over the pits there is actually more than 15 m thick waste rock material (Figure 2-2). Therefore, if necessary roots shall be able to access a depth of 6 m as demonstrated in the natural woodland (Section 4.5.2). It also evident that if below 4 M layer there is natural soil (much better water holding capacity), then the PAW status will be improved. Similarly, if part of the 4 m growth media is natural soil, the PAW status shall be more favourable than discussed in the Figure 4-24.

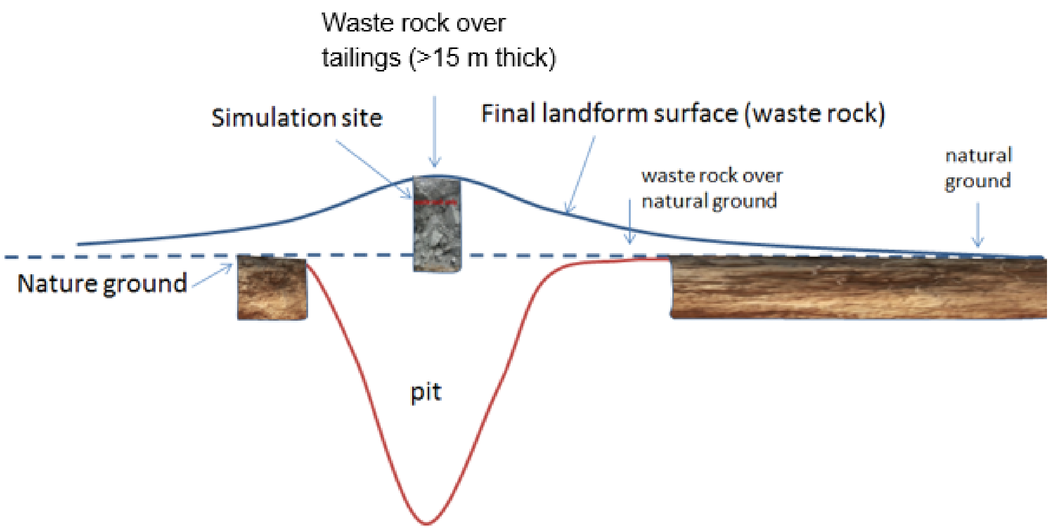


Figure 4-24: Illustration of the waste rock cover in relation to natural ground and backfilled pit

From the TLF monitoring, it is understood the four-metre waste rock growth substrate layer contains an average actual PAW of 261.4 mm, giving an average 65.4 mm of PAW for each metre. If the TLF growth substrate layer thickness was increased from four to five metres, PAW would increase from 261.4mm to 326.76mm, even further reducing the chance of a PAW deficit in any given year (Table 4-10).

Revegetation community type on the final landform shall be matched to the best to the site condition, for the crest site, it may well be a vegetation type that is similar to that at Site 30. Therefore it will use less water than that of the vegetation type that is similar to Site 21.

Fines % of the growth medium can be expected to be quite variable (Section 4.1), and it can range from 40% to 15% of fines. To assess the risk of an annual PAW deficit when using the scenario of the historic 117 years climate conditions, percentage of 117 years with a net negative PAW balance were commutated by varying the above three factors (Table 4-10).

The WAVES model outputs have been used to assess the probability of a constructed landform being able to sustain a mature reference vegetation (Site 21, a ‘conservative’ scenario).

For construction of the Ranger Mine final landform, there will be some areas where plant roots are entirely in the waste rock landform cover and also some areas where materials of a higher rock proportion may have to be used. Following on from the TLF WAVES modelling, WAVES simulations were run to derive the predicted PAW balance for materials higher than that in the TLF section 1A, which contains 33 % v/v of fines. Simulations were run with proportion of rock ranging from 30 to 15 % v/v fines, and for increased waste rock substrate layer thicknesses (i.e. five or six metres). Full simulation results are presented in Lu *et al.* 2019. A summary of percentage of years experiencing a net PAW deficit over 117 years for Site 21 and Site 30, for different modelled rock proportions and substrate thicknesses are presented in Table 4-10.

Table 4-10: Percentage of 117 years with a net negative PAW balance

Substrate thickness (m)	**ET from site**	**% Fines <=,2mm)**							
		33	**30**	**27.5**	**25**	**22.5**	**20**	**17.5**	**15**
4	Site 30	0%	0%	0%	1%	2%	9%	30%	53%
	Site 21	1%	5%	17%	38%	58%	83%	91%	98%
5	Site 30	0%	0%	0%	0%	0%	1%	3%	19%
	Site 21	0%	0%	1%	3%	13%	36%	64%	88%
6	Site 30	0%	0%	0%	0%	0%	0%	1%	2%
	Site 21	0%	0%	0%	0%	2%	6%	30%	57%

The WAVES model outputs have been used to assess the probability of a constructed landform being able to sustain a mature reference vegetation (Site 21, a ‘conservative’ scenario).

For construction of the Ranger Mine final landform, there will be some areas where plant roots are entirely in the waste rock landform cover and also some areas where materials of a higher rock proportion may have to be used. Following on from the TLF WAVES modelling, WAVES simulations were run to derive the predicted PAW balance for materials higher than that in the TLF section 1A, which contains 33 % v/v of fines. Simulations were run with proportion of rock ranging from 30 to 15 % v/v fines, and for increased waste rock substrate layer thicknesses

(i.e. five or six metres). Full simulation results are presented in Lu *et al.* 2019. A summary of percentage of years experiencing a net PAW deficit over 117 years for Site 21 and Site 30, for different modelled rock proportions and substrate thicknesses are presented in Table 4-10.

For the scenario of a four-metre waste rock cover and the plant water demand of Site 30, the waste rock cover would not experience any PAW deficit with fines percentage as low as 27.5%. With access to additional substrate (total thickness of 5 and 6 m), the waste rock cover would continue to have no net PAW deficit even when the percentage of fines dropped to 22.5 % and 20 % respectively. This is consistent with the general observation that some vegetation can still survive and even thrive on rocky ridges on the Jabiluka lease and close to Ranger Project Area (RPA) in the Kakadu NP.

For the scenario of a four-metre waste rock cover and the plant water demand of Site 21, decreasing the proportion of fines to 30 or 27.5 % results in a PAW deficit for 5 % and 17 % (respectively) for the 117 years modelled (Table 4-10). However, these deficits can be offset by an increase in substrate thickness (presumably up to the 6 m). This analysis has demonstrated that a five-metre thick growth substrate containing 30% fines (particles <=2 mm) would never experience a net PAW deficit (based on the 117-year rainfall record), although a substrate containing 27.5% fines would experience a net PAW deficit for about 1 % of the years of the modelled scenario and would require an increased substrate (total thickness six metres) to avoid any PAW deficit. A six-metre-thick cover would continue to have no net PAW deficit even when the proportion of fines decreases to 25 %.

This adjustment of the fines proportion by 2.5 % reductions also demonstrates how the PAW status of the landform should improve over time as the proportion of fines improves (and thus rock proportion decreases) due to weathering and soil formation processes.

Previous studies have reported that modifying waste rock cover thicknesses can provide greater (potential) PAW and thus aid in the establishment of self-sustaining plant communities. Mature trees were considered to be able to access water down to four to six metres below ground surface (Lamoureux *et al.* 2016). Further details are available in Lu *et al.* (2018 & 2019) and Lu (2017).

4.5.5 Soil water retention as affected by landform construction method (including ripping)

In addition to designing planting to optimise vegetation sustainability (i.e. the right species and density for the right locations), the final landform cover will also be designed and constructed to optimise the ability of the final landform to sustain the target vegetation. Choice of construction design and method can have a positive (and negative) impact on the ability of the final landform cover to store/release water and sustain the target vegetation. Final landform cover construction methods and their impact on plant growth substrate properties are discussed in the following sub-sections with full details provided in Section 9.

4.5.5.1 Sub-surface consolidated horizon

The final landform cover over mined out pits will be constructed in lifts (MCP Section 9.4.5). The material at the surface of waste rock dump lifts (or layers) is often consolidated due to heavy machinery activities, such as dump trucks positioning and dumping material in accordance with the spacing plan, or dozers pushing material off tip heads or flattening paddock dumps as shown in Figure 4-25 (e.g. Martin *et al*. 2004 and Diodato & Parizek 1994). This mechanical disturbance can also cause larger particles to break-down, increasing the proportion of fines in the compacted zone. This sub-surface consolidated horizon can be up to one-metre thick and shows a sharp transition back to uncompacted material (Martin *et al*. 2004).

This sub-surface consolidated horizon can be important in reducing macropores and increasing water retention capacity of the plant growth substrate and is discussed in the following sections.

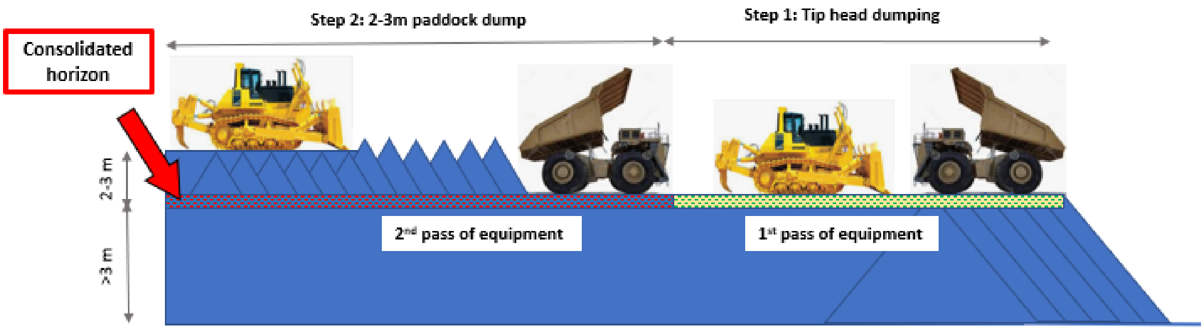


Figure 4-25: Example of a combination of construction methods improving density of sub-surface horizon

4.5.5.2 Macropores and preferential flow

A number of authors have observed that water flow in waste rock dumps occurs preferentially through channels and voids/macropores (e.g. Harries & Ritchie 1983). Water may flow in the channels and macropores somewhat independently. The hydraulic conductivities of the macropore region can be up to several orders of magnitude higher than the micropore hydraulic conductivity. In a waste rock dump composed of coarse fragments, with limited fines content, flow is expected to occur predominantly through partially-saturated channels (Smith *et al*. 1995). These preferential flow paths effectively bypass the desired even percolation of rainfall throughout the profile and prevent the wetting up of the material and thus development of positive PAW.

Good agreement between WAVES modelling results and the measured PAW dynamics in the TLF (Figure 4-22 and Figure 4-23) suggests that the substrate in the TLF 1A is behaving like a soil. The wetting front progression (assessed by soil volumetric water content dynamics) in the TLF 1A section is shown in Figure 4-16. The behaviour of wetting front progress after a significant rainfall (47.8mm) on 24 January 2016 demonstrates a steady downward progression of the wetting front without abrupt peak at lower positions. This suggest that preferential pathways are not a major issue in the TLF as it was constructed.

The sub-surface consolidated horizons within waste rock dumps act to intercept preferential flow paths provided by large or consecutive macropores, or air voids, formed during material placement. Cutting any preferential flow paths off at the higher density horizon ensures that the percolating water is redistributed laterally before continuing through the profile. Additional preferential flow paths may occur below the higher density horizon however these should reduce over time with gravitational compaction and the generation and movement of fines into the voids.

In a more typical 'soil', even incidental mechanical compaction can result in negative impacts to vegetation establishment, including reduced infiltration and root penetration. In fact, in their literature review of over 200 references, the Supervising Scientist were unable to locate any studies that directly investigated a positive outcome of compaction on post-mining rehabilitation (Supervising Scientist 2019). The review concluded, however, that there was also no evidence that the creation of higher density horizons will be "unequivocally detrimental to ecosystem restoration".

The TLF was constructed with two lifts of two to three metres each, thus including a central consolidated horizon. The MCP states that the final landform will be constructed using a similar method; therefore, the degree of consolidation shall not significantly differ from that of the TLF. The proposed paddock dump method for the final landform surface layer is unlikely to create a sub-surface consolidated horizon that is impermeable. Observations at the TLF where the same dumping method was used, does not suggest there were such an impermeable layer (Figure 3-5 and Figure 4-18). The sub-surface consolidated horizons proposed for the Ranger final landform, being only incidental due to heavy equipment traffic, will not create an impermeable layer but will break the preferential pathway and slow down the rate of water movement through the profile, and should not impact the ability of roots or water to penetrate to deeper levels.

4.5.5.3 Water retention characteristics

Compaction changes the pore size distribution of a soil. Specifically, it reduces the volume and continuity of the larger pores (voids) in the soil, which slows the movement of water through the soil (Hillel 1980). Dawson and Morgenstern (1995) found that hydraulic conductivity of waste rock material decreases with decreasing void ratio. Archer and Smith (1972) investigated the relation between bulk density, available water capacity and air capacity of soils. They found that for four soils of different textures studied, the volumetric water content increased linearly with bulk density until, depending on texture, a maximum bulk density was reached above which continued compaction decreased the water content. It was concluded that available water (and air) capacity could be optimised using cultivation techniques to adjust the bulk density. The available water capacity of coarse-textured *droughty* soils may be increased by increasing the bulk density, provided the air capacity remains above acceptable lower limits (Archer & Smith 1972).

Knoche (2006) investigated the structural dynamics of a vegetative soil cover for waste rock dumps and found that, six years after placement, self-compaction increased the soil dry bulk density and, as a consequence, decreased the air filled macropores and increased the water storing medium pores. This resulted in a "significant increase in plant available water-holding capacity".

Due to their higher bulk density and proportion of fines, sub-surface consolidated horizons could have a greater ability to retain water than uncompacted waste rock.

4.5.5.4 Particle size segregation due to dumping method

Final landform backfill methods used at Ranger mine are consistent with those found elsewhere (e.g. McLemore *et al.* 2009; Wilson 2011; Nichols 1986), and include the following:

- Tip head or end dumping (dumping rock over dump face resulting in some particle size segregation down slope towards the toe of the rock pile, with particle size generally increasing).
- Short or push dumping (dumping from trucks then levelling by pushing by dozers resulting in particle size segregation; finer at the top, coarser at the toe of the rock pile).
- Paddock dumping (dumping in small piles on the surface of the rock dump, grading the material, and compacting in layers or lifts resulting in dense layers with no real particle size segregation).

End dumping and push dumping are known to result in some particle size segregation down slope, with coarse material occurring further down the profile. There is likely a height threshold below which segregation is insignificant, which would need to be determined at a site-specific level (Wilson 2011).

Particle size segregation was observed during the TLF construction process in 2008. The photograph in Figure 4-26 shows dump trucks end dumping waste rock to form the lower layers of the landform and larger rocks and boulders accumulating at the toe of the dump face.

Figure 4-26: Tip head dumping of the lower layer(s) during TLF construction

The observed particle size segregation means that in the upper levels of the TLF layer there is a higher proportion of fines, which increases the potential PAW in these locations. Positive PAW is more valuable to a vegetation community at the upper-levels of the substrate profile, where more species have access, than further down the profile. Thus, use of a construction method that results in increased fines in the upper level of the (sub-surface) layer is likely to have a beneficial ecological impact.

5 THE ERA REVEGETATION STRATEGY

The Ranger Revegetation Strategy was first endorsed by stakeholders and an independent scientific advisory panel (the Alligator Rivers Region Technical Committee) in 2004 (Reddell & Meek 2004) and more recently updated, refined and published in the Ranger Mine Closure Plan (this document). The strategy is developed based on the learnings from extensive revegetation trials at Ranger and the revegetation of ERA's Jabiluka mineral lease, over the last three decades undertaken by ERA, as well as other research agencies (e.g. CSIRO, ERISS and CDU). Most significantly, recent learnings and experience from a large-scale landform trial of revegetation and monitoring methods, has enabled ERA to further refine its revegetation strategy as reviewed in this report.

A key aspect of the strategy is that the final landform growth medium will be predominately waste rock, setting a not insignificant challenge for the establishment of self-sustaining native eucalypt-dominated woodland. Experience and research outcomes have shown that this objective is achievable, and ongoing efforts are focussed on optimising establishment practices to maximise success, including harnessing and manipulating natural ecological processes such as reproductive phenology and the structural and functional importance of framework species.

5.1 Fourteen key elements

The strategy is comprised of fourteen elements that address: setting objectives and targets; understanding site physical and chemical constraints; species selection and target densities; site preparation and soil amendments including microbial inoculants; plant establishment methods including fertiliser use and irrigation; seed management; weed and fire management; and ongoing monitoring.

It is believed that the strategy will continue to be improved based on long term monitoring of the past revegetation, feedback from stakeholders and forthcoming learnings from the progressive revegetation on site – especially the revegetation of the Pit 1 landform. The fourteen elements of the revegetation strategy are outlined below:

1. Develop different revegetation strategies for different land surface: waste rock covered landform vs disturbed natural land with a 'soil' layer (e.g. land application areas).
2. Identify the likely physical and chemical constraints of the final landform that will influence both the initial establishment and the long-term growth, development and functioning of revegetated plant communities.
3. Maximise surface roughness and "patchiness" during site preparation.
4. Identify and describe vegetation types that are ecologically and technically realistic target endpoints (or 'habitats'), for different facets of the final landform, based on the likely physical and chemical environments that will be created.
5. Use of seed collected within KNP for all species.
6. Introduce a range of local mycorrhizal fungi to aid in the establishment of the framework species.

7. Include non-aggressive local native acacias but avoid the use of high densities of aggressive Acacia species.
8. Avoid actively introducing overly competitive grasses and herbaceous species, or sensitive species, until framework species are established and conditions are suitable.
9. Use nursery-grown planting stock to establish the framework species.
10. Apply fertilisers in a strategic manner using formulations and delivery methods that maximise their effectiveness.
11. Provide irrigation to new planted or sown plants.
12. Rigorously control potentially threatening weed species, both on and in proximity to the final landform.
13. Exclude fire from the revegetation areas during the first 5 – 8 years after establishment.
14. Design and implement a rigorous and scientifically-based strategy for on-going evaluation of the performance of the revegetation.

Element 1: Develop different revegetation strategies for different land surface types

The physical, chemical and biological characteristics of the waste rock landform and disturbed natural areas with a 'soil' layer are fundamentally different to each other and also from the natural ecosystems of the region. Despite this, they share a broad objective of re-establishing vegetation that is similar to the natural eucalypt-dominated woodlands, or other suitable vegetation communities of the surrounding area. To achieve this from such different starting points requires specially tailored revegetation strategies and the revegetation will develop along different pathways, or trajectories, to become the mature target ecosystem/s.

The waste rock landform presents unique ground conditions which are not present in the natural environment and subsequent elements of this revegetation strategy are largely focused on addressing unique challenges such as limited plant available water (PAW; in unit volume but similar in total root extractable volume), high levels of sunlight, thermal stress and open space, threat of weeds and fire, and an absence of any plants, propagules, organic matter, nutrient cycling, or natural fauna or microbial communities.

While areas of disturbed natural land with soil, such as the Land Application Areas (LAAs), have more suitable physical and chemical characteristics for vegetation establishment compared to bare waste rock, it still requires a revegetation strategy that will overcome its own unique challenges. These include the threats of 'weeds' (including local native aggressive acacias and spear grasses), fire, herbivores and competition for resources from surrounding vegetation, which necessities adjusted strategies such as spray of pre-emergence herbicides, more frequent weed and fire management and revegetation maintenance interventions (e.g. thinning of aggressive acacias).

ERA will use revegetation domains to identify and describe the different post-mining conditions of the final landform and surrounding disturbed areas requiring rehabilitation (Section 7.2.1.1).

Element 2: Identify the likely physical and chemical constraints of the final landform that will influence both the initial establishment and the long-term growth, development and functioning of revegetated plant communities

This element concentrates on characterising geomorphic and hydrological features, in different facets of the rehabilitation, that will determine (a) seasonal water availability for vegetation (e.g. infiltration and PAW), (b) chemical fertility and nutrition in the varying substrates, and (c) any other features that will impact revegetation (Section 7.2.1.1).

ERA's water balance study of the Ranger trial landform indicates that a waste rock cover layer of 4 – 6 metres thick would provide sufficient plant available water for most overstorey revegetation (Section 1 and Lu *et al*. 2019). Although framework tree and some shrub roots are capable of accessing deeper rock substrates (up to 6 metres), low net PAW in the near surface section (e.g. 0 – 1 metres) may affect the establishment and success of some shallower rooting species. Evidence from the trial landform indicates that surface and subsurface preparation methods such as rip lines and consolidation of sections of the subsurface as a result of material placement methods will improve the water holding capacity of the waste rock substrate.

Many soils typical of the tropical north of Australia are very old and highly leached, and have inherently low fertility, including a particularly low phosphorus and nitrogen content (Langkamp & Dalling 1979). Ranger Mine waste rock has, compared to the natural undisturbed soils of the area, higher pH, higher content of labile minerals, but lower organic carbon content, and nitrogen (Fitzpatrick 1989). Huang and You (2018) found that nutritional and microbial components of the TLF waste rock 'soil' was developing, however they observed relatively low rates of mineralisation that may be due to heat stress, rapid evaporation and water deficit at the surface. As vegetation establishes, and overstorey canopy and shade from other plants increase, these conditions should improve. The chemical characteristics and nutritional processes of the rehabilitated waste rock landform is presented in Section 4.3.

There is no concern of phytotoxity limiting revegetation outcomes. As part of a 2018 cumulative ecological risk assessment, Bayliss (2018) determined that risks to revegetation from mine-derived chemicals is assumed to be zero. This is supported by observations and studies of natural vegetation irrigated with water (mostly waste rock solutes) for over a decade, which indicate there are no observed negative effects on vegetation from waste rock contaminants (e.g. Addison 2011).

Element 3: Maximise surface roughness and 'patchiness' during site preparation

The aim is to establish a heterogeneous land surface that has (a) localised run-on/ runoff zones for control and capture of sediment, water and nutrients, and (b) microhabitats for seedling establishment and litter accumulation/decomposition and nutrient cycling, to support plant development, and to encourage natural flora recruitment and ground dwelling fauna. Experience and modelling have shown that rip lines installed at across the entire surface of the waste rock landform will mitigate soil loss and sediment transport (Saynor *et al*. 2019), particularly where slopes are less than 4% (i.e. the majority of the final landform).

Site preparation, including surface treatments, are presented in MCP Section 9.4.5.

Element 4: Identify and describe vegetation types that are ecologically, culturally and technically realistic target endpoints, for different facets of the final landform, based on the likely physical and chemical environments that will be created

The identification of suitable reference vegetation types has mainly been based on surveys in the surrounding natural landscapes that are potential geomorphic analogues of those formed on the final landform (based on the reasonable assumption that many of the environmental determinants of vegetation distribution will be similar in these settings). The majority of the landform will be revegetated to open eucalypt-dominated woodland vegetation typical of the surrounding area. Reference sites are discussed in Section 2.1.

The revegetation strategy is to initially establish framework overstorey species along with a subset of important and predictable midstorey and understorey species (MCP Section 5.3.3). Framework species control much of a site's nutrient and water resources, providing many of the core habitat values for other plants and animals, and contributing substantially to both the overall functioning and long-term stability of the plant communities (Reddell & Hopkins 1994). They typically include eucalypts, corymbia, xanthostemons, ironwoods, kakadu plum, quinine bush and other long-lived shrubs. Ecologically, these species are characterised by:

- High resistance to (tolerance of) fire.
- Reliance primarily on vegetative regeneration strategies (through root suckers, lignotubers and rhizomes) in response to stresses and disturbance.
- Seeds which are relatively short-lived and do not accumulate as a canopy (serotinous) or soil seed bank.
- A population structure dominated by even-age cohorts from one or a small number of discrete regeneration/recruitment events (usually from vegetative sprouts), resulting in highly discontinuous size class distribution.
- High predictability of growth performance and development.

Element 5: Use of seed collected within KNP for all species

The use of seed collected only from within KNP ensures that the genetic make-up of the revegetation is consistent with locally adapted populations of each species and provides a buffer for adapting to future global change (Zimmermann 2013b). To this end, a 'conservative provenance zone' has been adopted based on assessment of environmental factors, species distributions, taxonomy, present and past gene flow and species traits known to influence genetic variation in plants (Zimmermann & Lu 2015).

In 2011 to 2013, ERA conducted an extensive study investigating the provenance boundaries of the Ranger Mine revegetation in order to possibly extend the 30 km seed collection zone (Zimmermann 2013b, Zimmermann & Lu 2015). The usefulness of genetic and non-genetic methods was assessed, and a non-genetic approach, based on the methods developed by FloraBank, Greening Australia and other experts in the field, was adopted. The method assessed environmental factors, gene flow and species traits known to influence genetic variation in plants and identified zones of least likely genetic variation. The resulting zones match the eco-geography of the Ranger Mine area and hence maintain the 'home site' advantage of local plants. Some genetic diversity that may be present in more distant seeds is welcomed, as it may allow plant populations to respond to environmental changes such as climate change (e.g. Prober *et al*. 2015). This 'composite provenancing' approach ensures increased genetic diversity whilst reducing the risk of genetic pollution and outbreeding depression.

In identifying the environmental factors, the provenance assessment took into account the unique growing conditions on the constructed final landform, which are unlike those found in the natural surrounding ecosystems. Earlier studies identified an analogue site the nearby Georgetown area on rocky substrates.

The Atlas of Living Australia was identified as the most suitable and accurate environmental modelling tool, in the absence of fine-scale regional soil, vegetation and climate data. Environmental layers relevant to plant species distribution in the Top End (mean annual evaporation, annual precipitation, mean annual temperature, annual drainage, and topographic wetness index) were combined to predict a zone with a similar environment to the Ranger Mine, representing the Ranger Mine 'environmental provenance zone'. Investigations into revegetation species distributions found that each is well represented within the conservative provenance zone.

An assessment of potential gene flow indicated that there are no major geographic barriers within the Top End that may hinder the exchange of genetic material. As far as is known, there were no historical barriers in the Top End in the more recent geological past and the evolution in climate and vegetation was most likely uniform. Pollination takes place for the large majority of the investigated species not only by insects, but also by birds and bats, with most birds being generalists and hence being able to use other species as stepping stones between populations. Dispersal mostly takes place within 1 km of the source, but birds and bats can carry seeds over longer distances (e.g. 100 km).

Considering the abundance of birds, a continuous vegetation cover and that most revegetation species are common and widespread across the Top End, genetic exchange is likely to happen over large areas, if not the entire region. Any localised environmental variations that could cause genetic variation were eliminated by composite provenancing, which identified the 'environmental provenance zone' eco-geographically similar to the Ranger Mine. This was further narrowed by applying the conservative provenance zone. Seed collection guidelines further define and match the vegetation community and local environmental characteristics with the disturbed and created environments to be revegetated.

The seeds collected within the proposed conservative provenance zone (Figure 5-1) should be well adapted to the current conditions of the Ranger Mine, as well as provide sufficient genetic diversity to reduce inbreeding, promote the plants' adaptive potential and increase the resilience of the revegetation areas against moderate changes in climate. However, larger changes in climate may require seeds to be sourced from environments currently dissimilar to the Ranger Mine area, with the risk that they may not perform well under the current environmental conditions at the mine. The scope of changes in climate and associated risks for revegetation has a high degree of uncertainty at this point in time and should be reassessed in the future.

The outcomes of this study were presented to ARRTC and submitted to the GAC Board for endorsement. The GAC advised that "… after long and careful consideration… [the GAC Board] …are comfortable with seeds being collected for rehabilitation only within the borders of Kakadu" (Melanie Impey 2015, *pers. comm.*, 12 August). This makes provision for harvesting seeds from the southern part of Kakadu NP, where edaphic conditions are closer to the future conditions at the Ranger Mine under global climate change scenarios.

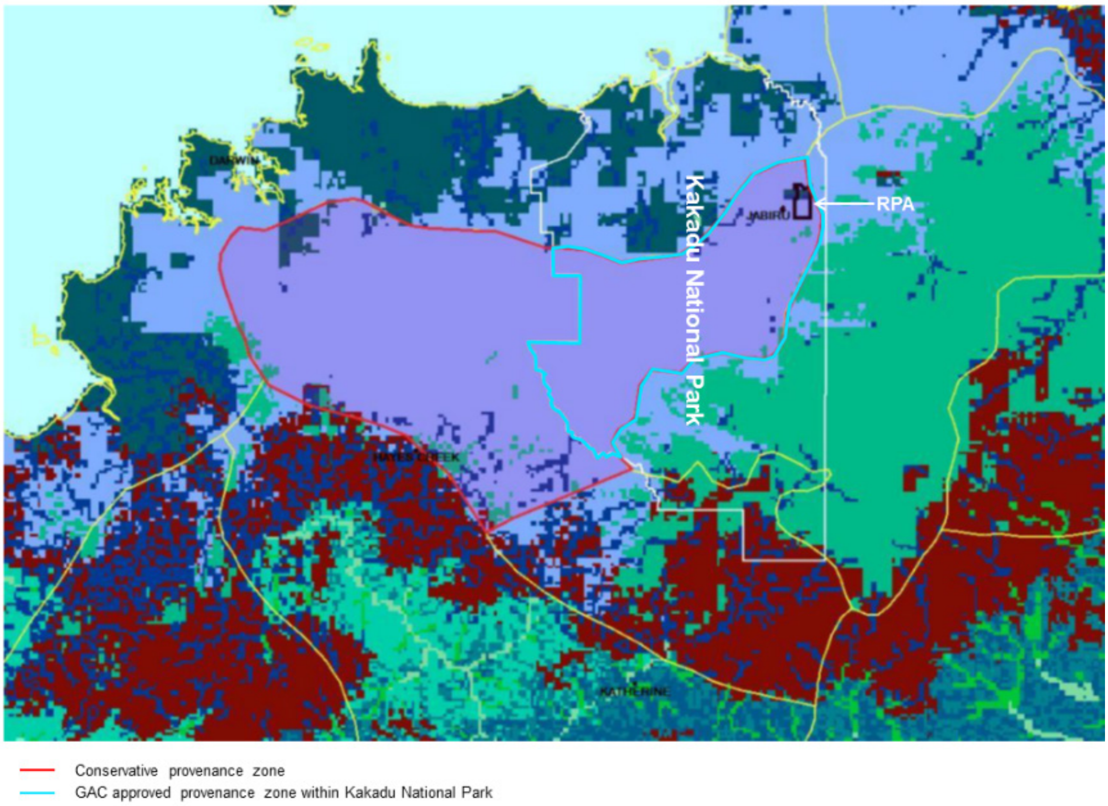


Figure 5-1: Proposed conservative provenance zone (bordered by the red line) and the GAC approved provenance zone within Kakadu NP (bordered by the blue line)

Element 6: Introduce a range of local mycorrhizal fungi to aid in the establishment of the framework species

As discussed in Section 1 above, initial establishment of vegetation into waste rock is a challenge; the substrate lacks nutrients, organic matter and fine particles, and, is also virtually devoid of nutrient-acquisitioning microorganisms (Reddell & Milnes 1992; Milnes 1989). Symbiotic microorganisms, such as mycorrhiza fungi and *Rhizobium* bacteria, play a critical role in nutrient uptake (esp. nitrogen and phosphorus) from soil by native Australian plants (Attiwill & Wilson 2006), and are highly prevalent in the natural soils of the Kakadu region (Brundrett *et al*. 1995; Reddell & Milnes 1992; Reddell & Joyce 1989). The vast majority of flora species in the undisturbed woodlands surrounding Ranger Mine have been found to have positive associations with symbiotic microorganisms (Reddell & Milnes 1992).

The importance of symbiotic microorganisms for the revegetation of post-mining land has been well documented (Johnson & Milnes 2007; Chandrasekaran *et al*. 2000; Corbett, M 1999). Mycorrhizal and *Rhizobium* inoculation of tubestock has been found to alleviate nutritional problems and promote plant growth during early establishment (Reddell & Zimmerman 2002). *Eucalyptus miniata* tubestock had significantly improved establishment on Ranger waste rock when inoculated with *Pisolithus* and *Laccaria*, or when 'locally contaminated' by *Nothocastoreum* (Gordon *et al*. 1997; Reddell *et al*. 1999). Inoculated seedlings had significantly greater shoot growth and leaf phosphorous concentrations than uninoculated seedlings, and seedling dry weight was found to increase consistently with levels of fungi colonisation (Reddell *et al*. 1999). Hinz (1997, as reported in Corbett M 1999) also found that *Nothocastoreum* mycorrhizal associations were also important for *E. tetrodonta* growth and development at Gove mine. Inoculation of *Rhizobium* has also been found to alleviate *Acacia* seedlings' nitrogen deficiencies when growing on Ranger waste rock (Reddell & Milnes 1992)

From their review of revegetation research at Ranger Mine, Reddell and Zimmermann (2002) concluded that "*inoculation of framework species with spores of ectomycorrhizal fungi would seem a very cheap and effective way of partially alleviating nutrient limitations to seedling establishment on the waste rock stockpiles*".

An effective microbial population, including mycorrhizae, is considered essential to establishing a self-sustaining woodland ecosystem on waste rock. A practical method has been refined at Ranger Mine by incorporating mycorrhizal fungal spores in the tubestock potting mix during propagation in the nursery.

Element 7: Include non-aggressive local native acacias but avoid the use of high densities of aggressive acacia species

A number of acacia species are common in the local woodlands, and are generally a positive component of the revegetation because of their ability to fix atmospheric nitrogen and rapidly produce organic matter. However, some acacias can be overly 'aggressive' in young revegetation and outcompete the slower-growing framework species, which are much less competitive until they have established dominant canopy and underground regenerative structures (e.g. Meek 2008; Zimmerman & Reddell 2011). Only natural proportions of short-statured, non-aggressive acacias will be included at initial establishment. Other acacia species are expected to self-colonise over time or can be introduced at the secondary establishment stage, once the framework species are dominating the site (see Element 8 below).

Element 8: Avoid actively introducing overly competitive grasses and herbaceous species, or sensitive species, until framework species are established and conditions are suitable

In young revegetation, vigorous grasses and herbaceous species can outcompete the preferred framework species (as for acacias) and if present in high densities can also increase the risk of fire (e.g. Meek 2008). Only low-risk native grasses and herbs will be introduced at initial establishment.

As the initial plantings of (mostly) framework overstorey and midstorey species establish and develop, a process expected to take five or more years based on trial landform experience (Section 3.2), the soil and litter layer will develop, canopy should increase providing shade and plants will develop attributes resilient to fires (e.g. stem diameter, lignotubers). It is at this stage that introductions of the remaining target understorey (and any midstorey or overstorey) species are planned to complete the diversity of the ecosystem. These species are generally those that are either too high risk or, alternatively, too sensitive to introduce at the earlier (initial) stage.

High risk species, also known as r-strategists (*sensu* MacArthur & Wilson 1967), are those that have, for example, high fecundity and rapid growth and should thrive in the temporary initial conditions of open space and high sunlight. These species might threaten to take advantage of the situation and out-compete the preferred eucalypt and other framework species as they gradually mature. This group includes aggressive acacias (e.g. *Acacia holosericea*), grasses (e.g. Sorghum spp.) and some herbs and will only be introduced during the secondary establishment stage. This will ensure that the preferred species are dominating the ecosystem and the r-strategists can establish in natural densities that will be supportive of a stable, self-sustaining ecosystem.

Sensitive species are those that are not suited to initial conditions however, they should be suited to passive or active introduction as environmental conditions improve. For example, *Xanthostemon paradoxus* is an important midstorey tree species and has shown extremely low survival rates in past revegetation at ERA. Research conducted in 2011–12 investigated the potential reasons for this and tested planting methods that could be used to improve the survival rate of this species in future revegetation (Gellert 2012). This study demonstrated that the use of shade-cloth tree shelters when planting can significantly increase survival, likely because the shade cloth reduced the light stress and heat stress experienced by the plants during planting shock and initial establishment.

More recently, Parry (2018) found that understorey species established from seed at almost twice the rate in the presence of surface litter as compared to other ameliorants (fine sand, fertiliser, ground incorporated organic matter, or combinations) or controls. Relationships between seedling emergence and distance to nearest tree, canopy cover and seed mass were also found. The study concluded that when establishing native understorey on mine waste rock in hot and intermittently dry periods in the wet season, the application of locally-collected surface litter to waste rock with broadcast seed may improve seedling establishment. With understorey species that have poor establishment from seed, tubestock planting has been proved to be a viable method for more efficiently introducing native understorey species into the ecosystem (Parry 2018).

These species will be established through either application of seed or tubestock planting, potentially concentrated in islands or strips across the final landform (particularly for the more infrequent or recalcitrant species). These concentrated areas will act as sources of future propagules which will spread out and self-colonise the rest of the landform over time. The work will be scheduled to utilise wet season rains and will be complemented by application of suitable fertiliser to assist early establishment and also contribute to the overall nutrient status of the developing rehabilitation.

Refining the appropriate introduction strategy for each species is the focus of the ERA species establishment research program (SERP) and is discussed further in Section 3.3.

Element 9: Use nursery-grown planting stock to establish the framework species

Based on current technology this will (a) significantly reduce the risk of planting failure associated with erratic rainfall and extreme temperatures; (b) accelerate the speed of vegetation development; and (c) overcome the poor predictability of establishing a final revegetated landform from direct seeding techniques. This strategy is proven to be the most cost-effective method for the initial establishment of framework species at Ranger and is reasonable given the constraint imposed by greatly limited seed availability within KNP. However, where reliable and predictable direct seeding success can be achieved for some species, such as Pandanus and Kapok *(Cochlospermum spp.)*, this method will be used.

Vegetation establishment techniques are discussed in Section 3.3.3 and MCP Section 9.4.6.

Element 10: Apply fertilisers in a strategic manner using formulations and delivery methods that maximise their effectiveness and environmental outcomes

Slow release fertiliser will be incorporated into the potting media for all planting stock, at rates that provide a significant 'residual' effect on growth after planting out. Some fertiliser will also be applied during the first wet season to facilitate more rapid seedling growth, especially if direct seeding is used; however, this fertiliser will not be of a highly soluble formulation. Additional fertiliser will be applied as required to ensure vegetation structural development is not inhibited and that sufficient site nutrient recapitalisation occurs, and also to support any subsequent infill or understorey planting. Fertilisation particularly favours invasive grassy species colonisation in the Top End and will be carefully managed to minimise this risk.

Use of fertilisers in the Ranger revegetation program is included in MCP Section 9.4.6.

Element 11: Provide irrigation to new planted or sown plants

For the initial planting activities, irrigation shall ensure good plant survival rates across all framework species during dry season and potentially erratic wet season conditions. However, irrigation will only be applied for 6 months or so, to avoid dependence and encourage deep rooting. Where possible, wet season rains will be used as the primary water source, particularly for the replacement ('infill') and secondary planting activities.

Some detail on proposed irrigation is provided in MCP Section 9.4.6.

Element 12: Rigorously control potentially threatening weed species

Weeds are the most critical risk to the reconstruction of the ecosystem. Final landform substrates shall be carefully managed during construction to prevent site contamination with weeds or their seeds. Furthermore, a weed-free buffer zone (approximately 200 metres wide) around the revegetation sites will be established to assist in preventing weed incursion into revegetation zones and areas will be treated with a pre-emergence, residual herbicide prior to planting. Weed monitoring and control will continue during the revegetation and post-closure management phases until closure criteria and relinquishment are achieved.

Weed management is further discussed in MCP Section 9.4.6.

Element 13: Exclude fire from the revegetation areas during early establishment until all plants have developed adequate resilience

Fire will be actively excluded from the developing revegetation through a program of controlled fuel reduction burns in surrounding vegetation and delayed introduction of highly flammable or high-biomass species, such as vigorous grasses. However, fire-resilience is a desirable feature of the mature ecosystem and it is important to introduce it as soon as possible, to ensure that fire-sensitive species do not come to dominate the revegetation. Introduction of low intensity fire to the developing revegetation will be dependent on the stage of development in the revegetation, for example framework species achieving a minimum stem diameter of six centimetres (Gellert 2013) and optimal fuel loads being present. Fire would then be used to maintain 'natural' fuel loads and to prime the framework species composition and structure to future fire regimes.

Surveys of the vegetation response to a controlled burn undertaken on the TLF (Wright 2019a), have shown that, especially when combined with appropriate and thorough herbicide application, may also be a useful method for controlling the spread of weeds and undesirable aggressive species such as *Acacia holosericea*. Fire may also promote germination and recruitment of several species such as *Owenia vernicosa* and *Eucalyptus tetrodonta*, and contribute to the establishment of a functioning and robust ecosystem.

Element 14: Design and implement a rigorous and scientifically based strategy for on-going evaluation of the performance of the revegetation

ERA is committed to a period of monitoring and maintenance, including activities required to manage the rehabilitated site, until all closure criteria can be satisfied (MCP Section 8).

A flora and fauna monitoring program will be developed for rehabilitation and closure, taking into consideration the information provided by the monitoring of natural reference sites. The monitoring program will comprise vegetation plots and fauna observation methods to assess terrestrial flora and fauna development.

The monitoring program will capture relevant information as the revegetation progresses. For example, in the initial stages of revegetation (e.g. years 1–5), the flora monitoring will focus on species survival rates, which will inform remediation works. As plants develop, a more comprehensive suite of parameters addressing ecosystem development and closure criteria will be introduced. The early fauna monitoring (e.g. years 1–3) is likely to focus on incidental observations of vertebrates and invertebrates. As habitat features develop, there will be an increase in monitoring to include trapping and systematic observation-based surveys to determine the presence of major functional groups. The proposed survey frequency of flora and fauna across the final landform is: three, six and 12 months (year 1); annually (years 1–5); and one-off surveys every five years (e.g. at years 10, 15, etc).

6 PLANNED REVEGETATION TRIALS

6.1 Pit 1 revegetation trials

Pit 1 will be available for revegetation in 2021, two years before other sections of the FLF, and provides an opportunity to test and evaluate a range of aspects relating to early revegetation activities. Overall, Pit 1 will allow ERA to:

- Fine tune nursery propagation and planting methods;
- Obtain improved data on predicted species performance and adjust planting strategy (species, density, locations) accordingly;
- Develop efficient monitoring for establishment and long term species-specific performance;
- Inform the FLF Revegetation Application (July 2022 submission); and
- Inform future trials and scaling up for operational planning for FLF (2023 – 2025);

The revegetation activities at Pit 1 will include ‘conceptual reference ecosystem’ (CRE) trial plantings based on reference ecosystem surveys, and targeted revegetation trials.

The information presented in the following sections is subject to change as the Pit 1 trials are yet to be finalised and stakeholder consultation is ongoing. The completed design, details on execution and preliminary results will be provided in the 2021 MCP.

6.1.1 Conceptual reference ecosystem trial planting

ERA has been collaborating with key stakeholders to develop a series of ‘conceptual reference ecosystems’ that represent the locally-occurring natural vegetation communities most likely to be suited to the challenges posed by the rehabilitated Ranger Mine site (Section 2.1.3). Recent focus has been eucalypt-dominated woodland ecosystems, based on vegetation surveys conducted by Supervising Scientist Branch (SSB) and ERA on ecosystems in areas adjacent to the Ranger Mine. Four potential woodland CREs have been identified: the Initial Conceptual Reference Ecosystem (ICRE) based on SSB survey sites, and three versions of draft Agreed Conceptual Reference Ecosystems (ACREs) based on different combinations of SSB and/or ERA survey sites.

Multiple areas of Pit 1 will be planted trialling different CREs. The objective is to revegetate using different CREs so that their suitability for revegetating waste rock landforms can be assessed/determined. The CRE trial planting will also provide an opportunity to visually demonstrate the different ecosystem types to Traditional Owners and external stakeholders.

6.1.2 Targeted revegetation trials

Tubestock and direct seeding trials will be conducted on Pit 1.

6.1.2.1 Tubestock trials

Similarly to Stage 13.1, the overall objective of the tubestock trials is to investigate different potting and propagation techniques with the aim of improving tubestock survival and health during the first two years after planting. The study will provide an opportunity to:

- Gather species-specific data to fine tune nursery propagation methods, such as germination rates, required growing times, irrigation requirements etc.;
- Obtain baseline performance data for species that have not been grown on FLF media previously; and
- Propagate and plant tubestock during different times of the year.

ERA has explored a range of methodologies and techniques for optimising tubestock planting success (most recently at Stage 13.1). Three factors have been identified which warrant further investigation/experience, including:

- *Pot type* - Although plastic nursery tubes are the commercial standard for revegetation, past experience at Ranger suggests biodegradable pots may be a preferable option as they eliminate the need to depot.
- *Plant Size/Age* - Planting smaller tubestock may result in a higher root-shoot ratio, decreasing the initial water demand of the seedling. Planting smaller sized tubestock appeared to improve *Xanthostemon paradoxus* survival on the TLF (*per comms*. Dr Ping Lu).
- *Planting Season* - When revegetation is at its peak in 2024/2025, tubestock will need to be grown and planted all year round. There will be three lots of tubestock planting: during the wet, dry and build-up.

Species will be selected for tubestock trials based on the following four considerations (Table 6-1):

- Which species are most important to optimise establishment? eg. Culturally significant species, species which occur at high densities etc.
- Which species have historically been difficult to establish on waste rock?
- Which species do ERA have limited or no experience establishing on waste rock?
- Which species are not suitable for initial planting, either because the conditions are too harsh or because they may be too aggressive?

All of the trial species will be planted in March; however, due to space and planting restraints approximately half of the species will be included in the dry and build-up trials. The species chosen for the unseasonal planting trials will: generally occur at high densities; will be a range of Families and lifeforms; and a combination of deciduous, evergreen and/or fresh-seeded species.

Table 6-1: Species being considered for trials at Pit 1 (currently being reviewed)

Species	Lifeform	Family
Overstorey and Midstorey		
Acacia lamprocarpa	Tree	Fabaceae
Acacia mimula	Shrub	Fabaceae
Brachychiton megaphyllus	Shrub	Malvaceae
Buchanania obovata	Shrub	Anacardiaceae
Calytrix exstipulata	Shrub	Myrtaceae
Corymbia bleeseri	Tree	Myrtaceae
Corymbia chartacea	Tree	Myrtaceae
Corymbia disjuncta	Tree	Myrtaceae
Corymbia dunlopiana	Tree	Myrtaceae
Corymbia foelscheana	Tree	Myrtaceae
Corymbia polysciada	Tree	Myrtaceae
Corymbia porrecta	Tree	Myrtaceae
Erythrophleum chlorostachys	Tree	Fabaceae
Eucalyptus miniata	Tree	Myrtaceae
Eucalyptus tectifica	Tree	Myrtaceae
Eucalyptus tetrodonta	Tree	Myrtaceae
Gardenia megasperma	Shrub	Rubiaceae
Grevillea mimosoides	Shrub	Rubiaceae
Jacksonia dilatata	Shrub	Fabaceae
Livistona humilis	Palm	Arecaceae
Melaleuca viridiflora	Tree	Myrtaceae
Planchonella arnhemica	Shrub	Sapotaceae
Planchonia careya	Shrub	Lecythidaceae
Stenocarpus acacioides	Tree	Proteaceae
Syzygium eucalyptoides ssp. bleeseri	Shrub	Myrtaceae
Terminalia ferdinandiana	Shrub	Combretaceae
Terminalia pterocarya	Shrub	Combretaceae
Understorey		
Acacia gonocarpa	Shrub	Fabaceae
Alloteropsis semialata	Grass	Poaceae
Ampelocissus acetosa	Vine	Vitaceae
Aristida holathera	Grass	Poaceae
Cartonema spicatum	Herb	Commelinaceae

Species	Lifeform	Family
Eriachne obtusa	Grass	Poaceae
Galactia tenuiflora	Vine	Fabaceae
Larsenaikia suffruticosa	Subshrub	Rubiaceae
Grevillea goodii	Shrub	Proteaceae
Haemodorum coccineum	Herb	Haemodoraceae
Heteropogon triticeus	Grass	Poaceae
Petalostigma quadriloculare	Shrub	Picrodendraceae
Tacca leontopetaloides	Herb	Taccaceae
Uraria lagopodioides	Vine	Fabaceae

6.1.2.2 Direct seeding trials

The overall objective is to determine which species can successfully establish from seed on the FLF during the initial stages of revegetation. In addition, for some species:

- Does time of sowing impact plant establishment from seed?
- Does surface treatment impact establishment from seed?

Species will be selected for direct seeding trials based on the following considerations:

- Which species have seed available in high quantities, and are easy to collect and process?
- Which species occur at high densities in the surrounding bushland, therefore would provide significant savings if able to direct seed?
- Which species have failed to establish in previous direct seeding trials on Ranger waste rock?
- Which species do ERA have limited or no experience direct seeding on waste rock?
- Which species have naturally colonised Ranger waste rock dumps or typically grow in harsh conditions somewhat similar to those found on the initial FLF?
- Which species are not suitable for initial planting, either because the conditions are too harsh or because they may be too aggressive?

The majority of the species selected for direct seeding trials will be understorey species, however a few midstorey species that are deemed to be potentially suitable for direct seeding will also be included.

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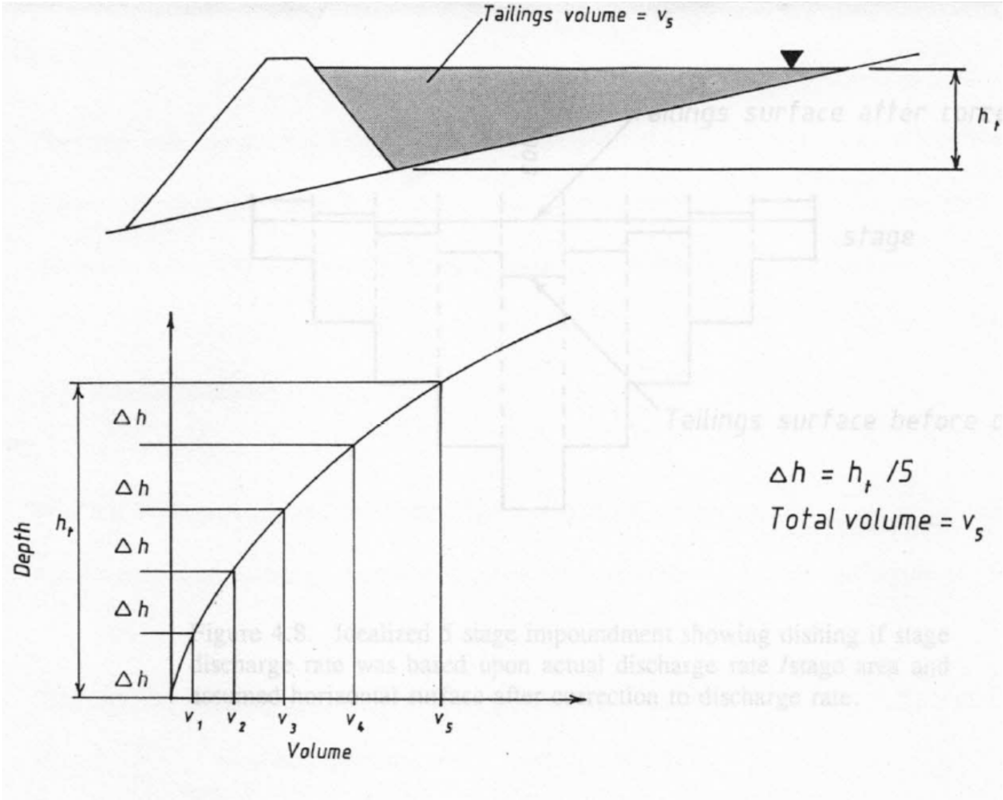
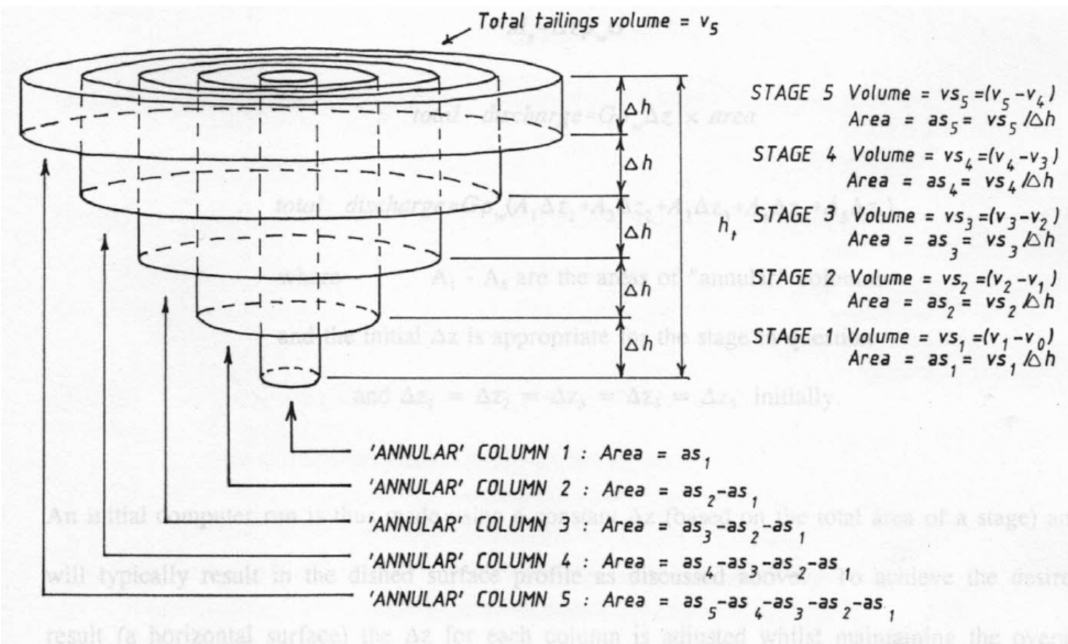
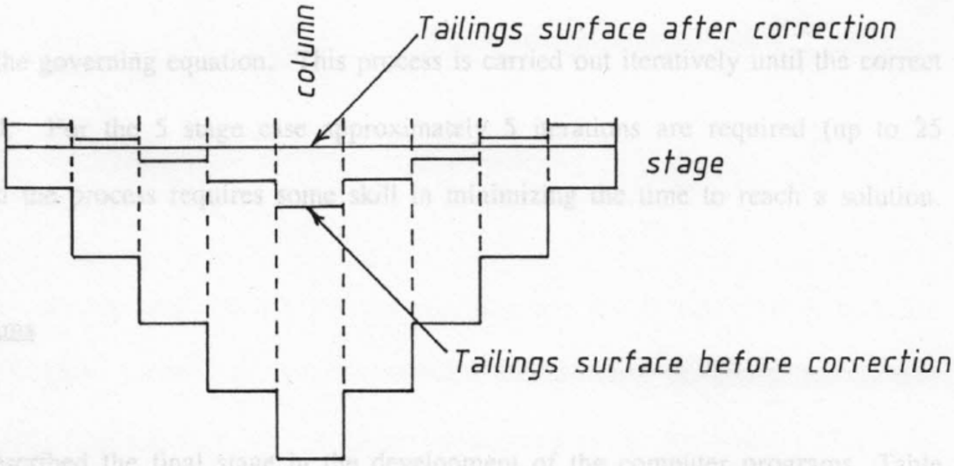
APPENDIX 5.2 CONSOLIDATION MODEL A



ACTUAL IMPOUNDMENT



MODEL IMPOUNDMENT



Idealized 5 stage impoundment showing dishing if stage discnarge rate was based upon actual discharge rate /stage area and assumed horizontal surface after correction to discharge rate.

APPENDIX 5.3 CONSOLIDATION MODEL B

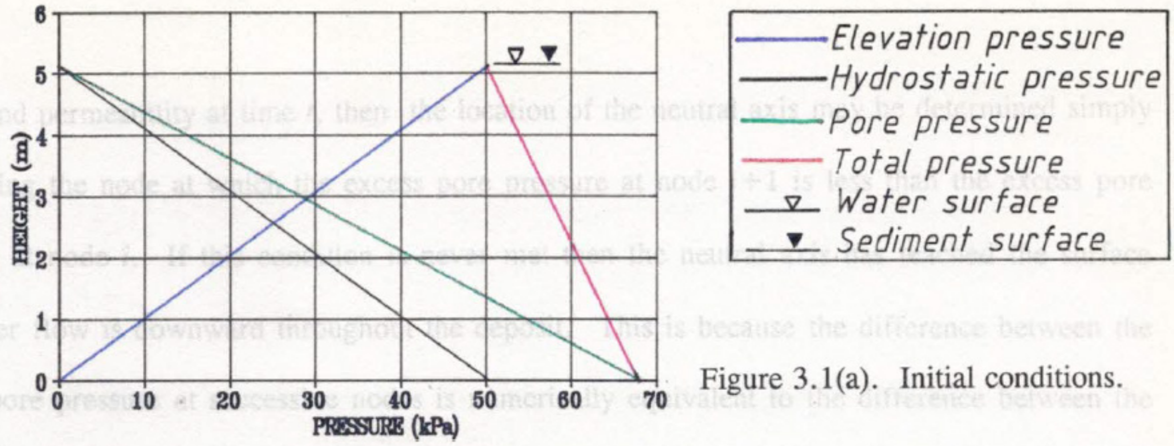


Figure 3.1(a). Initial conditions.

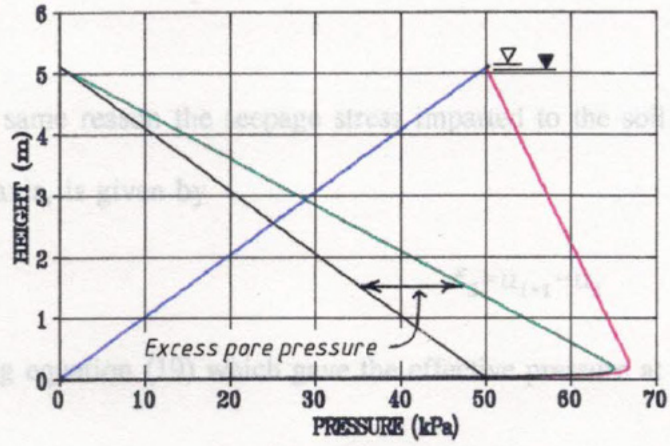


Figure 3.1(b). Conditions immediately after opening base drain.

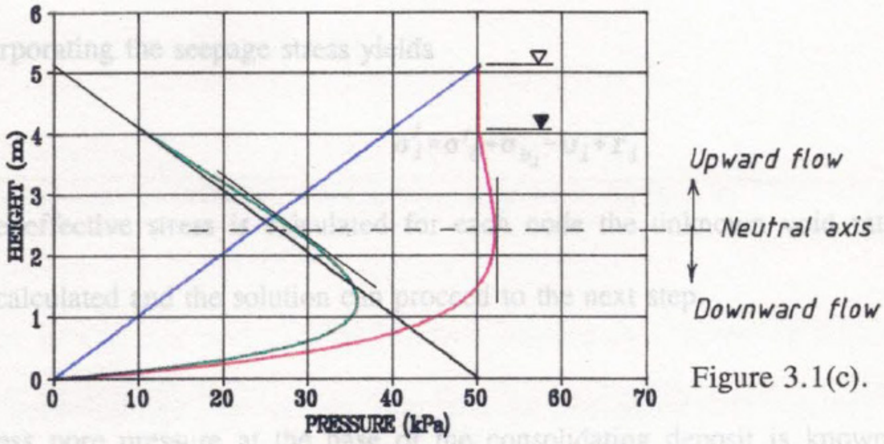


Figure 3.1(c). Intermediate case.

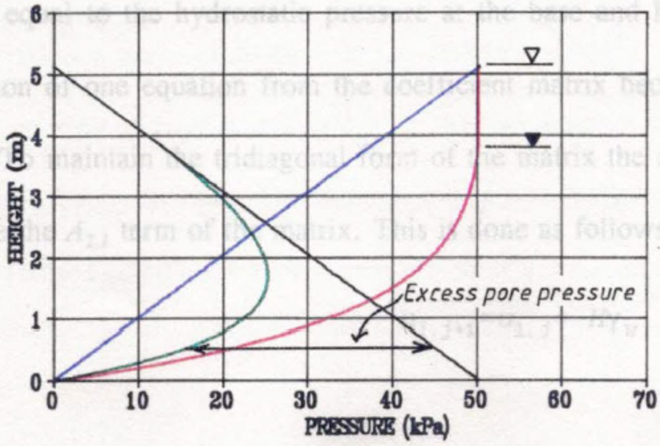


Figure 3.1(d). Approaching equilibrium conditions.

Figure 3.1. Pore pressure conditions in an underconsolidated slurry.

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APPENDIX 5.4 KEY KNOWLEDGE NEEDS

Australian Government
Department of Agriculture, Water and the Environment
Supervising Scientist

KEY KNOWLEDGE NEEDS - May 2020

Note: KKN questions shown in greyed-out text have been closed out (i.e. required information has been attained) or removed (i.e. clearly no longer required, or covered in other KKNs)

LANDFORM REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH)**
LAN1	Erosion	Baseline	LAN1. Determining baseline erosion and sediment transport characteristics in areas surrounding the RPA	LAN1A. What are the baseline rates of gully formation for areas surrounding the RPA?	Baseline information on gully characteristics and formation (e.g. extent/occurrence and distribution of gullies of differing size and complexity, rate of ‘knick-point’ retreat) in natural landforms is needed. This information can be obtained from appropriate imagery and will be used to assess whether the extent, rate and magnitude of gully formation predicted for the rehabilitated site will vary significantly from those observed in comparable non-mine disturbed landforms in adjacent areas.	SSB
				LAN1B. What are the baseline rates of sediment transport and deposition in creeks and billabongs?	The risk of bedload sediment transport from the rehabilitated site is generally considered to be low because of the ability to manage it through appropriate mitigation measures (e.g. sedimentation basins). However, information on natural bedload yields in Magela and Gulungul creeks is needed to distinguish mine-derived bedload from natural yields and monitor the effectiveness of mitigation measures. If the mitigation measures are not effective, this information would also be used to assess potential impacts to aquatic ecosystems.	SSB
LAN2	Erosion	Baseline	LAN2. Understanding the landscape-scale processes and extreme events affecting landform stability	LAN2A. What major landscape-scale processes could impact the stability of the rehabilitated landform (e.g. fire, extreme events, climate)?	Identification of major landscape-scale processes or extreme events that could adversely affect the stability of the rehabilitated landform is needed to assess whether there are any potential risks associated with these processes that could result in mass failure and containment of tailings for at least 10,000 years. This information is likely to be available in existing reports and will be used to assess potential impacts on landform stability (see LAN2B).	SSB
				LAN2B. How will these landscape-scale processes impact the stability of the rehabilitated landform (e.g. mass failure, subsidence)?	Information to assess the degree to which major landscape-scale processes or extreme events could affect the stability of the rehabilitated landform is being addressed and will be further sought from the available literature.	BOTH
LAN3	Erosion	Predicting	LAN3. Predicting erosion of the rehabilitated landform	LAN3A. What is the optimal landform shape and surface (e.g. riplines, substrate characteristics) that will minimise erosion?	The shape (e.g. slope) and surface characteristics (e.g. particle size, roughness, riplines, drainage) of the rehabilitated landform will influence erosion rates. These characteristics and their effect on erosion rates can be assessed through an iterative modelling approach using CAESAR-Lisflood. Information on proposed landform characteristics should be used to optimise landform design. This could include using ‘geomorphic reclamation’ processes, which are the characteristics (e.g. slope curvature/length) of the pre-mining or adjacent landscape. These will be calculated and used to inform the design of the final landform.	BOTH
				LAN3B. Where, when and how much consolidation will occur on the landform?	The degree of subsidence within the rehabilitated landform (e.g. over Pits 1 and 3 associated with tailings consolidation) may influence erosional processes. Determining these rates will require some knowledge of predicted location and extent of consolidation over the pits.	ERA

LANDFORM REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH)**
				LAN3C. How can we optimise the landform evolution model to predict the erosion characteristics of the final landform (e.g. refining parameters, validation using bedload, suspended sediment and erosion measurements, quantification of uncertainty and modelling scenarios)?	Some input parameters for the landform evolution model may be influenced by local conditions and these need to be understood to maximise the accuracy of the model predictions. Examples of parameters include: • sediment settling velocity, • shear stress and roughness, • rate of weathering for waste rock, • effect of vegetation succession and fire on suspended sediment transport, and • impact of extreme rainfall events and scenarios over time on suspended sediment transport. Validation of bedload predictions could be undertaken by comparing measured parameters from the trial landform and the rehabilitated Pit 1 landform (e.g. bedload, suspended sediments) with the model outputs at both plot and catchment scale.	SSB
				LAN3D. What are the erosion characteristics of the final landform under a range of modelling scenarios (e.g. location, extent, timeframe, groundwater expression and effectiveness of mitigations)?	In order to assess the effectiveness of the final landform design (including any integral control structures), it will be necessary to identify and understand the erosion characteristics (extent and magnitude of gully formation; denudation and erosion rate; potential for groundwater expression) that may result under the different model scenarios.	SSB
				LAN3E. How much suspended sediment will be transported from the rehabilitated site (including land application areas) by surface water?	Suspended sediment has the potential to impact on aquatic ecosystems downstream of the rehabilitated site. Turbidity/suspended sediment should be monitored on the constructed Pit 1 final landform to determine what loads are likely to be released from the mine site and to assist with the calibration/validation of model predictions of suspended sediment transport at the catchment scale. The significance of suspended sediment that may be transported from land application areas will also need to be assessed. This assessment is commensurate with the level of soil disturbance associated with remediation of these areas.	BOTH

WATER AND SEDIMENT REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
WS1	Biodiversity and ecosystem health	Source	WS1. Characterising contaminant sources on the RPA	WS1A. What contaminants (including nutrients) are present on the rehabilitated site (e.g. contaminated soils, sediments and groundwater; tailings and waste rock)?	A comparative assessment of contaminants of potential concern (COPCs) and their respective source(s) (e.g. waste rock, tailings/pore water, groundwater, soils) is needed, including consideration of any 'hotspots' that may be present on the rehabilitated site (e.g. groundwater under the plant area, GCT2 area, LAAs, billabong/stream sediments). This information contributes to whole-of-site contaminant transport modelling to predict post-closure water quality and will inform the rehabilitation and risk management of the site.	ERA
				WS1B. What factors are likely to be present that influence the mobilisation of contaminants from their source(s)?	For each contaminant source present on the rehabilitated site, physical, chemical and other factors that affect, or interact to affect, contaminant mobilisation need to be identified and assessed. This information contributes to whole-of-site contaminant transport modelling to predict post-closure water quality and will inform the rehabilitation and risk management of the site.	ERA
WS2	Biodiversity and ecosystem health	Pathway	WS2. Predicting transport of contaminants in groundwater	WS2A. What is the nature and extent of groundwater movement, now and over the long-term?	Knowledge of current and post-closure groundwater movement is required, both within the rehabilitated site and to the off-site environment. This is being achieved through numerical model predictions that consider the implications of changes to the groundwater movement due to the mine closure and recovery, i.e. the return to a stable state of levels, contaminant concentrations, flow paths and the influence of sea-level rise on groundwater flow, after rehabilitation. The most appropriate monitoring locations for calibration and verification of models needs consideration. This information contributes to whole-of-site contaminant transport modelling to predict post-closure water quality and will inform the rehabilitation and risk management of the site.	ERA
				WS2B. What factors are likely to be present that influence contaminant (including nutrients) transport in the groundwater pathway?	There is a need to determine whether conservative modelling or reactive modelling provides a worse-case for contaminant transport within the groundwater pathway. Reactive modelling examines physical and chemical factors that influence contaminant transport within the groundwater pathway (e.g. pH, redox conditions) and interactions amongst these (e.g. COPC mixtures). Identification of these factors (and their significance) informs contaminant transport modelling to predict the downstream concentrations of COPCs.	ERA
				WS2C. What are predicted contaminant (including nutrients) concentrations in groundwater over time?	The contaminant concentration in the groundwater system will vary with time due to the development of geochemical reactions at the source and movement of contaminants through the groundwater. Understanding of the variation of contaminant concentration will be used to determine the timing and amount of contaminant that may reach a receptor affecting the health of the ecosystem. Knowledge of the concentrations of COPCs in groundwater informs contaminant transport modelling used to predict the downstream concentrations of COPCs and inform rehabilitation and risk mitigation strategies.	ERA

WATER AND SEDIMENT REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
WS3	Biodiversity and ecosystem health	Pathway	WS3. Predicting transport of contaminants in surface water	WS3A. What is the nature and extent of surface water movement, now and over the long-term?	Detailed information on current and future hydrological conditions for catchments both within the RPA and adjacent/downstream areas is required. The effect of sea-level rise on the surface waters flow also needs consideration. The timing and magnitude of surface water flows informs contaminant transport modelling used to predict the on-site and downstream concentrations of COPCs.	ERA
				WS3B. What concentrations of contaminants from the rehabilitated site will aquatic (surface and ground-water dependent) ecosystems be exposed to?	Determination of the concentrations of COPCs that aquatic ecosystems (including riparian vegetation) will be exposed to from the rehabilitated site needs to be based on the integration of modelling predictions for both groundwater (WS2) and surface water (WS3). Predicted COPC concentrations in surface and groundwaters can then be compared against water quality guideline values or other locally-derived biological effects information (for ground-water dependant species) in order to assess whether aquatic biodiversity and ecosystem health are exposed to risk following rehabilitation. (To address this KKN, information from WS3D is first required.)	ERA
				WS3C. What factors are likely to be present that influence contaminant (including nutrients) transport in the surface water pathway?	There is a need to determine whether conservative modelling or reactive modelling provides a worse-case for contaminant transport in the surface water pathway. Reactive modelling examines physical and chemical factors that will influence contaminant transport and toxicity (e.g. pH) and interactions amongst these (e.g. COPC mixtures). Identification of these factors (and their significance) informs contaminant transport modelling used to predict the downstream concentrations of COPCs.	ERA
				WS3D. Where and when does groundwater discharge to surface water?	Information on the locations and timing of groundwater discharge to surface water is required to assess the significance of this contaminant transport pathway. Improved understanding of groundwater/surface water interactions informs contaminant transport modelling used to predict the downstream concentrations of COPCs.	BOTH
				WS3E. What factors are likely to be present that influence contaminant transport (including nutrients) between groundwater and surface water?	Factors that could influence movement of contaminants, and limit or increase their concentration from groundwater to surface water, include geology, topography, aquifer geometry and hydraulic characteristics. Identification of these factors (and their significance) informs contaminant transport modelling to predict the downstream concentrations of COPCs.	ERA
				WS3F. What are the predicted concentrations of suspended sediment and contaminants (including nutrients) bound to suspended sediments in surface waters over time?	When suspended sediments are transported from the rehabilitated site, they could affect aquatic ecosystem health directly (e.g. habitats/biota effects) and/or indirectly (e.g. transport of bound contaminants). Knowledge of the concentrations of suspended sediments and associated contaminants informs contaminant transport modelling to predict the downstream concentrations of COPCs.	BOTH

WATER AND SEDIMENT REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				WS3G. To what extent will the interaction of contaminants between sediment and surface water affect their respective qualities?	Contaminants in surface water may accumulate in sediments to concentrations above those at which biological effects could be expected. Conversely, contaminants in sediments may resuspend into the water column and reduce water quality. An understanding of the factors affecting the flux of contaminants between surface waters and sediments is required to determine if closure criteria will protect both environmental compartments.	BOTH
				WS3H. Where and when will suspended sediments and associated contaminants accumulate downstream?	If contaminants from the rehabilitated site accumulate in downstream sediments, it is possible that they could affect aquatic ecosystem health directly and in the short term (e.g. to benthic biota) and/or in future through re-mobilisation of deposited contaminants. Knowledge of locations and likely timing for deposition of suspended sediments and associated contaminants informs the assessment of risk to aquatic ecosystems.	ERA
WS4	Biodiversity and ecosystem health	Receptor	WS4. Characterising baseline aquatic biodiversity and ecosystem health	WS4A. What are the nature and extent of baseline surface water, hyporheic and stygofauna communities, as well as other groundwater dependent ecosystems, and their associated environmental conditions?	Although there is currently substantial knowledge on baseline water quality and biodiversity in surface waters during early dry season (recessional) flow periods, information on water quality and biota for other periods of surface water flow and inundation (i.e. both wet and dry seasons, stream channels and billabongs) is limited. More complete information will allow a more comprehensive assessment of whether predicted (modelled) concentrations of COPCs transported from the rehabilitated site are likely to impact on downstream aquatic ecosystem health. Hyporheic and stygofauna communities in the Magela Creek sand beds are poorly understood and the significance of their contribution to ecological processes to the biodiversity of the ARR is unknown. The environmental conditions sustaining these (e.g. water quality, flow), and other groundwater dependent ecosystems (e.g. dry season water sources for riparian vegetation) are also unknown. If these communities are ecologically important, their potential sensitivity to increased solute loads needs to be assessed (WS7C). This information helps determine if specific closure criteria are needed to protect these communities.	SSB
WS5	Biodiversity and ecosystem health	Receptor	WS5. Determining the impact of contaminated sediments on aquatic biodiversity and ecosystem health	WS5A. Will contaminants in sediments result in biological impacts, including the effects of acid sulfate sediments?	Some COPCs transported from the rehabilitated site, e.g. uranium and sulfate, will bind to organic matter and benthic sediments in downstream ecosystems, in particular, the shallow lowland billabongs. The long-term risk of accumulation of these COPCs in sediment to biodiversity or ecological processes needs to be assessed for both the creek and billabongs. This information will inform management of the rehabilitated site and, in relation to sulfate in particular, any ongoing need to manage this COPC in surface and groundwater. Such a risk assessment would include analyses of the temporal trends in COPC concentrations in the sediments and, for sulfate, the predicted budget for billabongs (i.e. Coonjimba, Georgetown, Gulungul) to assess the risk of acid sulfate sediment formation and associated potential impacts on aquatic biodiversity and ecosystem health.	BOTH

WATER AND SEDIMENT REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				WS5B. What are the factors that influence the bioavailability and toxicity of contaminants in sediment?	Closure criteria for U in sediments were derived using sediments from Gulungul Billabong, as they are representative of the major depositional zones in and outside of the RPA (i.e. shallow backflow billabongs). However, if physico-chemical conditions (e.g. pH, TOC) of sediments differ from those in Gulungul Billabong, this may affect the toxicity of COPCs, and the closure criteria may not be appropriate. Knowledge of the influence of bioavailability and toxicity modifying factors in sediments helps derive closure criteria specific for different sediment conditions.	SSB
				WS5C. What would be the impact of contaminated sediments to surface aquatic ecosystems?	If predicted COPC concentrations in sediments are likely to reach a threshold where there is a risk that they could be mobilised into surface waters, the potential impacts to these aquatic ecosystems need to be assessed.	*Removed November 2019*
WS6	Biodiversity and ecosystem health	Receptor	WS6. Determining the impact of nutrients in surface water on aquatic biodiversity and ecosystem health	WS6A. What is the toxicity of ammonia to local aquatic species, considering varying local conditions (e.g. pH and temperature)?	The effects of ammonia on local species under local conditions need to be quantified. The toxicity of ammonia is highly influenced by pH and temperature, which can vary substantially between billabongs and streams, and seasonally. This research also needs to include assessment of toxicity to freshwater mussels, which have been reported as particularly sensitive to ammonia, an important component of the local aquatic ecosystem and a highly-valued food source for traditional owners. This information assists in deriving site-specific closure criteria for ammonia.	*Closed out May 2020*
				WS6B. Can Annual Additional Load Limits (AALL) be used to inform ammonia closure criteria?	A review of the literature supporting AALLs is needed to understand their continuing relevance. It needs to be determined whether ammonia loads could be considered in the same context as the AALLs.	ERA
				WS6C. Will the total loads of nutrients (N and P) to surface waters cause eutrophication?	Contaminant transport modelling will predict loads of nutrients that downstream surface waters are likely to receive from the rehabilitated site. This information should be used to assess if there is a risk of eutrophication to downstream surface waters.	ERA
WS7	Biodiversity and ecosystem health	Receptor	WS7. Determining the impact of contaminants in surface and ground-water on aquatic biodiversity and ecosystem health	WS7A. Are current guideline values appropriate given the potential for variability in toxicity due to mixtures, modifying factors and different exposure scenarios?	Water quality limits that have been derived for individual toxicants do not incorporate potential interactive (e.g. additive, synergistic, antagonistic) effects of toxicant mixtures or other modifying effects occurring in the field (e.g. pH, temperature, DOC). This knowledge informs the development and application of closure criteria for COPCs.	SSB
				WS7B. What is the risk associated with emerging contaminants?	Contaminant research has been prioritised on a risk basis, but the continued gathering of contaminant knowledge before and during the mine's transition into a rehabilitated site may result in the identification of new or emerging contaminants of potential concern (e.g. contaminated sites studies and where the risk profile of a contaminant changes through increased knowledge of effects or exposure). Where such contaminants are identified, they need to be assessed using a tiered, risk-based approach.	BOTH

WATER AND SEDIMENT REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH)**
				WS7C. Are current guideline values appropriate to protect the key groups of aquatic organisms that have not been represented in laboratory and field toxicity assessments (e.g. flow-dependent insects, hyporheic biota and stygofauna)?	Current guideline values are derived from a limited suite of laboratory tests and, where possible, validated using field-effects data. Some (sandy) stream-dwelling species, which have been reported as sensitive to contaminants, are not represented in these data sets and their sensitivity to COPCs are unknown. This knowledge will indicate if closure criteria are protective of these taxa and identify any phase of the hydrograph of receiving stream environments that represents greater risks to stream biota than other phases.	SSB
				WS7D. How do acidification events impact upon, or influence the toxicity of contaminants to, aquatic biota?	Acidification events, and associated increases in dissolved metal concentrations, have been observed in on-site waterbodies (e.g. Coonjimba Billabong, RP1) as a result of acid sulfate soil formation associated with elevated sulfate concentrations from the mine. These events typically occur during re-wetting events in the early wet season and in most cases are short-lived (days, weeks). In order to fully inform management actions for sulfate in surface and groundwaters (see WS5A), biological-effects studies of the impacts to such receiving waters should be undertaken to examine short (during events) and longer-term (seasonal, interannual) changes to biodiversity and ecological processes.	*Removed November 2019*
				WS7E. How will Mg:Ca ratios influence Mg toxicity?	An understanding of the Mg:Ca ratio of seepage water from various sources and how this affects toxicity is required. The gathering of field (or semi-field) effects data for mine released waters (including groundwater sources) mixed with receiving waters would provide supporting evidence.	*Closed out May 2020*
				WS7F. Can a contaminant plume in creek channels form a barrier that inhibits organism migration and connectivity (e.g. fish migration, invertebrate drift, gene flow)?	Previous studies in Magela Creek have demonstrated avoidance by fish of mine wastewater discharges, indicating potential reduced recruitment to upstream sites. Information on seasonal movement and dispersal of organisms needs to be considered and combined with groundwater contaminant modelling data, in order to assess potential for impaired movement and connectivity in streams.	SSB
				WS7G. What concentrations of contaminants will be detrimental to the health of (non-riparian) aquatic vegetation?	The guideline values for COPCs were derived using a limited species range that included one aquatic macrophyte (*Lemna*) with a relatively short exposure duration (4 days). Apart from their inherent biodiversity and conservation values, the diverse aquatic plant communities in billabongs and along littoral portions of the creeks constitute critical habitat for other biota, and for this reason are deserving of more detailed investigation than just the limited laboratory information available for the single species. Laboratory and field studies under a range of realistic exposure scenarios or across existing contaminant gradients in onsite waterbodies should be undertaken to assess the potential sub-lethal impacts of COPCs on aquatic vegetation in these aquatic ecosystems and thereby determine if healthy aquatic habitats can be maintained following rehabilitation.	BOTH

WATER AND SEDIMENT REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				WS7H. What concentrations of contaminants will be detrimental to the health of riparian vegetation?	Riparian vegetation, particularly that growing along the banks of the major drainage lines (Magela and Gulungul creeks) may be seasonally exposed to elevated concentrations of contaminants in shallow groundwater after minesite rehabilitation. An assessment of the potential sub-lethal impacts of COPCs on germination and early growth of representative species (e.g. through pot trials) will assist in determining if healthy riparian habitats can be maintained following rehabilitation.	SSB
WS8	Biodiversity and ecosystem health	Receptor	WS8. Determining the impact of suspended sediment on aquatic biodiversity and ecosystem health	WS8A. What are the physical effects of suspended sediment on aquatic biodiversity, including impacts from sedimentation and variation in sediment characteristics (e.g. particle size and shape)?	Suspended sediments can have various physical effects on aquatic ecosystems, such as habitat alteration (e.g. deposition), light attenuation and subsequent influence on primary productivity and physiological effects on organisms (e.g. inhibition of reproduction/growth, fish gill function). The magnitude of the effects of suspended sediments can vary according to their characteristics. For example, larger particle sizes are more likely to result in impacts associated with deposition (e.g. smothering of habitat), whereas smaller particle sizes are more likely to result in impacts upon filter feeding organisms. An assessment of potential impacts of suspended sediment on aquatic biodiversity should be based on predicted characteristics of sediments that may be transported from the rehabilitated site.	SSB
				WS8B. To what extent does salinity affect suspended particulates, and what are the ecological impacts of this?	Salinity can affect behaviour of suspended particles by processes such as flocculation and may affect the rate at which the particles settle from the water column. The potential for high-salinity waters associated with the rehabilitated site (e.g. evapo-concentration in billabongs during the dry season) to affect behaviour of suspended particulates (e.g. increased deposition rates) and subsequent ecological impacts (e.g. infilling of billabongs) needs to be assessed.	*Removed May 2020*

HEALTH IMPACTS OF RADIATION AND CONTAMINANTS REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
RAD1	Human and ecosystem health	Source	RAD1. Radionuclides in the rehabilitated site	RAD1A. What are the activity concentrations of uranium and actinium series radionuclides in the rehabilitated site, including waste rock, tailings and land application areas?	Waste rock, buried tailings and contaminated soils on land application areas represent potential sources of radionuclides to the environment from the rehabilitated site. The radionuclides of concern are those of the uranium and actinium decay series because they occur at elevated concentrations in the source materials. Radionuclides of the thorium decay series are not of concern, as they do not occur at elevated levels in the source materials. Knowledge of the activity concentrations of uranium and actinium decay series radionuclides in waste rock, tailings and land application area soils is needed to model activity concentrations in the environment post-rehabilitation, which in turn are needed to estimate radiation doses to the public and wildlife. The knowledge could be acquired through radionuclide measurements on existing waste rock, tailings and land application area soils.	ERA
RAD2	Human and ecosystem health	Pathway	RAD2. Radionuclides in aquatic ecosystems	RAD2A. What are the above-background activity concentrations of uranium and actinium series radionuclides in surface water and sediment?	Increased radionuclide activity concentrations in surface water and sediment due to contaminated water arising from the rehabilitated site could result in radiation doses above natural background to the public and wildlife. Knowledge of the increases in activity concentrations of uranium and actinium decay series radionuclides in surface water and sediment is needed to estimate these doses. The knowledge could be acquired through modelling of: • radionuclide releases to surface water via runoff and groundwater pathways from the rehabilitated site • the mixing of released radionuclides in surface water • radionuclide partitioning between sediment and water. Furthermore, the modelling of radionuclide releases could be based on an element with high solubility to provide conservative estimates of activity concentrations.	ERA
RAD3	Human and ecosystem health	Pathway	RAD3. Radon progeny in air	RAD3A. What is the above-background concentration of radon and radon progeny in air from the rehabilitated site?	Radon (a radioactive gas) will be emitted to the atmosphere from the rehabilitated site due to the decay of radium-226 in surface waste rock. The inhalation of radon progeny radionuclides produced through the decay of emitted radon could result in radiation doses above natural background to the public. Knowledge of radon and/or radon progeny concentrations in air is needed to estimate these doses. This knowledge could be acquired by modelling the atmospheric dispersion of radon from the rehabilitated site, using site-specific data (as necessary) for parameters such as: • radium-226 activity concentrations in surface waste rock (RAD1A) • radon exhalation rates for waste rock • dry and wet season meteorological conditions.	SSB

HEALTH IMPACTS OF RADIATION AND CONTAMINANTS REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				RAD3B. If an assessment using conservative values shows a potential issue with meeting closure criteria (3A and 7A): What is the equilibrium factor between radon progeny and radon in air?	If the modelling under RAD3A gives radon concentrations in air, then knowledge of the equilibrium factor between radon progeny and radon will be needed to obtain radon progeny concentrations for dose modelling. If needed, site-specific knowledge on equilibrium factors could potentially be acquired through simultaneous measurements of radon and radon progeny concentrations in ambient air off-site of the operating mine.	*Removed November 2019*
				RAD3C. If an assessment using conservative values shows a potential issue with meeting closure criteria (3A and 7A): What is the unattached fraction of radon progeny in air?	The dose coefficient for radon progeny depends on the proportion of radon progeny attached and unattached to aerosols. If needed, site-specific knowledge on the unattached fraction could be acquired through simultaneous measurements of radon progeny attached and unattached to aerosols in ambient air at locations off-site of the operating mine.	*Removed November 2019*
RAD4	Human and ecosystem health	Pathway	RAD4. Radionuclides in dust	RAD4A. If an assessment using conservative values shows a potential issue with meeting closure criteria (4B and 7A): What is the resuspension factor (or emission rate) of dust emitted from the final landform?	If the modelling under RAD4B uses a resuspension factor approach to estimate the release of radionuclides in dust from the rehabilitated site to the atmosphere, then site-specific knowledge of dust resuspension factors or emission rates may be needed. If needed, this knowledge could be acquired through measurements of radionuclide activity loadings in dust and activity concentrations in ambient air.	*Removed November 2019*
				RAD4B. What is the above-background activity concentration in air of long-lived alpha-emitting radionuclides in dust emitted from the final landform?	The inhalation of radionuclides in dust emitted to the atmosphere from the rehabilitated site could result in radiation doses above natural background to the public. Knowledge of airborne activity concentrations of radionuclides in dust is needed to estimate these doses. This knowledge could be acquired by modelling the atmospheric dispersion of radionuclides in dust from the rehabilitated site, using site-specific data (as necessary) for parameters such as: • activity concentrations of uranium and actinium decay series radionuclides in surface waste rock (RAD1A) • resuspension factors (or emission rates) of radionuclides in dust from waste rock • dry and wet season meteorological conditions.	*Closed out November 2019*
				RAD4C. If an assessment using conservative values shows a potential issue with meeting closure criteria (4B and 7A): What is the activity median aerodynamic diameter of long-lived alpha-emitting radionuclides in dust emitted from the final landform?	The dose coefficient for radionuclides in dust depends on the activity median aerodynamic diameter (i.e. size) of the aerosol. If needed, site-specific knowledge on activity median aerodynamic diameter could be acquired through radionuclide measurement of size fractionated dust samples collected using cascade impactors.	*Removed November 2019*

HEALTH IMPACTS OF RADIATION AND CONTAMINANTS REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
RAD5	Human and ecosystem health	Pathway	RAD5. Radionuclides in bushfoods	RAD5A. What are the concentration ratios of actinium-227 and protactinium-231 in bush foods?	The ingestion of uranium and actinium decay series radionuclides bioaccumulated in bush foods could result in radiation doses above natural background to the public. Radiation dose assessments for the human food chain use concentration ratios to predict radionuclide activity concentrations in food items from those in the surrounding soil or water. A sizeable body of knowledge exists on concentration ratios for uranium decay series radionuclides. However, there is effectively no knowledge (site-specific or otherwise) on concentration ratios for actinium decay series radionuclides. The actinium decay series radionuclides of potential concern include actinium-227 and protactinium-231, which have relatively high ingestion dose coefficients. Knowledge on concentration ratios for these radionuclides could potentially be acquired through sampling and measurement on bush foods and associated soils and waters after development of radiochemistry separation and measurement techniques for actinium-227 and protactinium-231.	SSB
RAD6	Human and ecosystem health	Receptor	RAD6. Radiation dose to wildlife	RAD6A. What are the representative organism groups that should be used in wildlife dose assessments for the rehabilitated site?	Wildlife dose assessments are generally based on a small number of organism groups representative of the broad variety of species present in the environment. This is because it is not usually practical to sample and perform radionuclide analyses on all species present. Knowledge of representative organism groups could potentially be acquired from reviewing ecological information about the species present in the local environment and generalising them up to a small number of representative organism groups. Alternatively, broad wildlife groups defined by international bodies (e.g. International Atomic Energy Agency) or within wildlife dose assessment tools (e.g. ERICA) could potentially be used. When selecting representative organism groups, consideration should be given to any rare, threatened or culturally significant species that may be present in the local environment.	*Closed out November 2019*

HEALTH IMPACTS OF RADIATION AND CONTAMINANTS REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				RAD6B. What are the whole-organism concentration ratios of uranium and actinium series radionuclides in wildlife represented by the representative organism groups?	The bioaccumulation of uranium and actinium decay series radionuclides in wildlife could result in radiation doses above natural background to those wildlife. Standard dose assessment tools for wildlife use whole organism concentration ratios to predict radionuclide activity concentrations in wildlife from those in the surrounding soil or water. Whole organism concentration ratios of uranium decay series radionuclides have been derived for some (but not all) types of wildlife using site-specific data. There is effectively no data (site-specific or otherwise) for deriving whole organism concentration ratios for actinium decay series radionuclides, specifically actinium-227 and protactinium-231. Knowledge of whole organism concentration ratios for uranium and actinium decay series radionuclides could potentially be acquired by one or more of the following methods: • sampling and radionuclide measurements on organisms and associated soil or water to derive additional site-specific values • review and analysis of international databases (e.g. Wildlife Transfer Database) and publications to fill gaps in site-specific values • use of surrogate organism and analogue element approaches to fill gaps in site-specific values.	SSB
				RAD6C. What are the tissue to whole organism conversion factors for uranium and actinium series radionuclides for wildlife represented by the representative organism groups?	Standard dose assessment tools for wildlife use whole organism concentration ratios to predict radionuclide activity concentrations in wildlife from those in the surrounding soil or water. Most site-specific data on radionuclide activity concentrations in wildlife is tissue-specific, as it was originally collected to support human food chain dose assessments. The data need to be converted to whole organism values to be useful in wildlife dose assessments. Knowledge on tissue to whole organism conversion factors could be acquired by one or more of the following methods: • review and analysis of existing site-specific data to reconstruct whole organisms from individual tissues using a mass balance approach • sampling and radionuclide measurements on the individual tissues comprising whole organisms • review and analysis of international databases and publications • use of surrogate organism and analogue element approaches to fill knowledge gaps.	SSB

HEALTH IMPACTS OF RADIATION AND CONTAMINANTS REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				RAD6D. What are the dose-effect relationships for wildlife represented by the representative organism groups?	The potential radiation risk to wildlife can be evaluated by comparing whole organism dose rates to environmental reference levels, which generally represent the dose rates at which radiation effects in organisms may begin to occur. Environmental reference levels derived by international bodies are currently used within the rehabilitation standard for radiation protection of the environment. If needed, dose-effect relationships for specific organism groups could be derived by one or more of the following methods: • laboratory studies within which aquatic and terrestrial organisms are chronically exposed to known activities of radionuclides and the effects on key biological endpoints (i.e. mortality, morbidity, reproduction and genetic mutations) observed • review of international databases (e.g. FREDERICA) and publications.	*Removed May 2020*
				RAD6E. What is the sensitivity of model parameters on the assessed radiation doses to wildlife?	Radiation dose modelling for wildlife uses a large number of parameters. The potential variability in parameter values used in the modelling can cause variability in the estimate of the dose to wildlife. Sensitivity analysis is a standard method that can be used to identify key parameters causing variability in modelling results. Understanding the variability in dose modelling results due to each input parameter is important so that research to acquire additional site-specific knowledge (if needed) can be appropriately prioritised and targeted.	ERA
RAD7	Human and ecosystem health	Receptor	RAD7. Radiation dose to the public	RAD7A. What is the above-background radiation dose to the public from all exposure pathways traceable to the rehabilitated site?	The pathways through which the public can be exposed to radiation due to the rehabilitated site are: • inhalation of radon progeny and radionuclides in dust • ingestion of bush foods and drinking water • external gamma The statutory limit on radiation dose to the public applies to the dose above natural background from all sources and exposure pathways summed. The assessment of radiation dose to the public due to the rehabilitated site requires an analysis of each exposure pathway for a clearly defined scenario of future land use. Parameterisation of exposure pathways can be made using existing knowledge and that acquired under RAD1A, RAD2A, RAD3A, RAD3B, RAD3C, RAD4A, RAD4B, RAD4C and RAD5A. Knowledge on future land use to develop a quantitative scenario against which radiation doses can be assessed can potentially be acquired by : • consultation with traditional owners • review of the literature or other records for information on historic use of the area	ERA

HEALTH IMPACTS OF RADIATION AND CONTAMINANTS REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
				RAD7B. What is the sensitivity of model parameters on the assessed doses to the public?	Radiation dose modelling uses a large number of parameters to estimate doses to the public. The potential variability in parameter values used in the modelling can cause variability in the estimate of the dose. Sensitivity analysis is a standard method that can be used to identify key parameters causing variability in modelling results. Understanding the potential variability in the estimated dose due to each input parameter is important so that research to acquire additional site-specific knowledge (if needed) can be appropriately prioritised and targeted.	ERA
RAD8	Ecosystem health	Receptor	RAD8. Impacts of contaminants on wildlife	RAD8A. Will contaminant concentrations in surface water (including creeks, billabongs and seeps) pose a risk of chronic or acute impacts to terrestrial wildlife?	Wildlife may drink water from waterbodies affected by the mine but their intake profile from these sources is not aligned with the models of intake on which livestock drinking water guidelines are based (e.g. infrequent, occasional use versus longer-term frequent use). Livestock drinking guidelines are probably not appropriate for small wildlife or taxa such as reptiles. An assessment of the risks associated with both chronic and acute impacts to all large and small terrestrial wildlife needs to take into account how much of an animal's consumption is likely to come from poor quality sources associated with the rehabilitated site. This information will determine if specific water quality closure criteria are required to protect large and small terrestrial wildlife.	ERA
RAD9	Human health	Receptor	RAD9. Impacts of contaminants on human health	RAD9A. What are the contaminants of potential concern to human health from the rehabilitated site?	Identification of the COPCs that may be elevated in soil (e.g. landform and LAAs) or water (e.g. creeks and billabongs) is a key first step in assessing potential risks to human health. A screening approach to identify those COPCs with higher toxicity (from relevant drinking water guidelines) and which may also be present in the environment due to the rehabilitated site should be undertaken. This will inform whether closure criteria for human health are required.	ERA
				RAD9B. What are the concentration factors for contaminants in bush foods?	Human food-chain assessments of COPC exposure use concentration factors to quantify transfer from the environment (e.g. soil and water) to food items. This is particularly the case for prospective assessments, where exposure estimates are made from predicted soil or water COPC concentrations using concentration factors.	SSB
				RAD9C. What are the concentrations of contaminants in drinking water sources?	Dietary exposure to COPCs in drinking water will be proportional to the COPC concentrations in the water and the amount consumed.	ERA
				RAD9D. What is the dietary exposure of, and toxicity risk to, a member of the public associated with all contaminant sources, and is this within relevant Australian and/or international guidelines?	The total dietary intake of each COPC needs to be assessed and compared to relevant guideline values to determine the acceptability of the exposure in a human health context.	ERA

ECOSYSTEM RESTORATION REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
ESR1	Ecosystem similarity	Ecosystem similarity	ESR1. Determining the requirements and characteristics of terrestrial vegetation in natural ecosystems adjacent to the mine site, including Kakadu National Park.	ESR1A. What are the compositional and structural characteristics of the terrestrial vegetation (including seasonally-inundated savanna) in natural ecosystems adjacent to the mine site, how do they vary spatially and temporally, and what are the factors that contribute to this variation?	Baseline information on terrestrial vegetation composition and structure at scales that adequately capture and explain heterogeneity in natural ecosystems is required. This information, historical or new, will be used in the development of closure criteria and to assess whether vegetation growing on the rehabilitated site is similar to reference sites observed in non-mine disturbed ecosystems in adjacent areas. Examples of compositional and structural characteristics of vegetation include species abundance, and density, number of species, size class distribution of trees and shrubs, vegetation strata (e.g. canopy or ground cover) and hollow abundance. Such information would ideally be based on large-scale survey methods (e.g. remote sensing) that will better capture the spatial and temporal variation than the historical smaller scale ground-based surveys. Accompanying environmental measurements are also required in order to identify factors accounting for the variations in vegetation. Identifying factors responsible for observed ecological patterns may assist in revegetation planning and establishment.	SSB
				ESR1B. Which indicators of similarity should be used to assess revegetation success?	The proposed vegetation similarity indicators have been drawn from the National Restoration Standards (Standards Reference Group SERA 2016) and include species composition, number of species, vegetation strata, tree/shrub class size distribution and vegetation distribution (‘naturalness’). Closure criteria will be developed for these indicators and applied for each of these to assess the degree of similarity between vegetation growing on the rehabilitated site and that observed in non-mine disturbed ecosystems in adjacent areas. Indicators will be developed for both understorey and overstorey vegetation.	*Closed out November 2019*
				ESR1C. What values should be prescribed to each indicator of similarity to demonstrate revegetation success?	Once appropriate similarity indicators have been identified, specific value(s) for each need to be established that account for the expected range in natural spatial and temporal variability (i.e. avoidance of single numbers). This information will be used in the development of closure criteria and to assess whether vegetation growing on the rehabilitated site is progressing acceptably towards that observed in non-mine disturbed ecosystems in adjacent areas, the extent of such progress, and whether it has achieved an agreed level of similarity. The indicator values may vary according to the spatial scale at which they are derived and this dependence needs to be understood for future applications.	BOTH

ECOSYSTEM RESTORATION REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
ESR2	Ecosystem similarity	Ecosystem similarity	ESR2. Determining the requirements and characteristics of a terrestrial faunal community similar to natural ecosystems adjacent to the mine site, including Kakadu National Park	ESR2A. What faunal community structure (composition, relative abundance, functional groups) is present in natural ecosystems adjacent to the mine site, and what factors influence variation in these community parameters?	Much baseline information on terrestrial fauna community structure in natural ecosystems adjacent to the mine site is already available, but additional information may be required. This reference information will be used to characterise fauna communities in natural ecosystems adjacent to the mine site, the extent of variation in the fauna and the factors that influence such variation. This context will be used in the development of faunal community closure criteria and to measure and interpret progress of fauna communities in the rehabilitated site towards those in adjacent suitable reference locations. For vertebrates, such information would ideally be based on contemporary fauna survey methods (e.g. camera trapping) that will better capture the spatial and temporal variation than the historical survey techniques.	BOTH
				ESR2B. What habitat, including enhancements, should be provided on the rehabilitated site to ensure or expedite the colonisation of fauna, including threatened species?	The establishment of vegetation does not guarantee that suitable habitats for terrestrial fauna colonisation are available, particularly early in the ecosystem restoration process. Information is needed on the time that it may take before the rehabilitated site can be expected to naturally develop key fauna habitat features (e.g. tree hollows); if this is likely to be many years, options for habitat enhancements will need to be examined (e.g. nesting boxes, rock piles).	BOTH
				ESR2C. What is the risk of introduced animals (e.g. cats and dogs) to faunal colonisation and long-term sustainability?	The risk that introduced animals could impede the re-establishment of fauna and the long-term sustainability of faunal communities needs to be assessed. This is likely to be particularly important early in the ecosystem restoration process, when the rehabilitated landscape could provide optimal habitat for introduced animals (e.g. ideal conditions for predators) and before suitable habitats for native fauna are established (e.g. fallen logs, tree hollows for refuge). This information will inform the need for mitigation measures, such as active management of introduced animals and/or establishment of habitat enhancements that favour native fauna.	BOTH
ESR3	Ecosystem similarity	Ecosystem similarity	ESR3. Understanding how to establish native terrestrial vegetation, including understory species.	ESR3A. How do we successfully establish terrestrial vegetation, including understory (e.g. seed supply, seed treatment and timing of planting)?	The ability to establish the full range (or an appropriate complement) of native vegetation species from the reference ecosystem needs to be demonstrated. While this has been shown in initial trials for over 35 framework species, there is far less available evidence for the successful establishment of a diverse suite of understorey species. This information will be sought from the literature, and from ongoing research including trials on the Ranger Trial Landform and, in future, on the Pit 1 rehabilitated site. The information will provide necessary assurance that it is possible to establish vegetation communities on the rehabilitated site that will be similar to adjacent non-mine disturbed ecosystems.	ERA

ECOSYSTEM RESTORATION REHABILITATION THEME

KKN No.	ER Link	Category	Title	Questions	Description	Responsibility (SSB/ERA/BOTH)
ESR4	Ecosystem similarity	Ecosystem similarity	ESR4. Determine the incidence and abundance of introduced species in natural ecosystems adjacent to the mine site, including Kakadu National Park, and their potential to impact on the successful rehabilitation of Ranger mine	ESR4A. What is the incidence and abundance of introduced animals and weeds in areas adjacent to the mine site, and what are the factors that will inform effective management of introduced species on the rehabilitated mine site?	Information on the composition and abundance of introduced species in areas adjacent to the rehabilitated site is required, both to assess the risk that these ecological stressors may pose to successful ecosystem restoration and to demonstrate that their presence on the site is not higher than in adjacent to areas. This information will be required throughout the restoration process to inform trigger points for implementing mitigation strategies (e.g. early detection of pests or weeds may allow for ready cost-effective eradications). Further research may be required to inform management options that (i) result in control of pests and weeds but (ii) do not prevent the successful restoration of native species and communities.	SSB
ESR5	Long term viability	Ecosystem Sustainability	ESR5. Develop a restoration trajectory for Ranger mine	ESR5A. What are the key sustainability indicators that should be used to measure restoration success?	The proposed indicators of long-term viability and ecosystem function (sustainability) of the restored ecosystem have been drawn from the National Restoration Standards (e.g. Standards Reference Group SERA 2016). These indicators include recruitment of revegetation, nutrient cycling, faunal usage, habitat availability, resilience to fire, extreme weather events, pests and diseases. Other attributes to be considered are external exchanges (e.g. habitat connectivity, physical conditions (e.g. nutrient availability), and absence of threats (e.g. weeds). This information will be used in the development of closure criteria and to assess whether ecosystems established on the rehabilitated site will be similar to those observed in natural non-mine disturbed ecosystems in adjacent areas.	BOTH
				ESR5B. What are possible/agreed restoration trajectories (flora and fauna) across the Ranger mine site; and which would ensure they will move to a sustainable ecosystem similar to those adjacent to the mine site, including Kakadu National Park?	Restoration trajectories will be required to assess the achievement of closure criteria that are expected to be reached after a period of time (e.g. decades) from the initial establishment. The trajectory approach outlined in the National Ecological Restoration Standards is based on modelling of a desired and/or expected trajectory pathway, distinguishing the desired pathway from possible undesired states, and selecting points within the desired trajectory that represent milestones leading to agreed closure. This should be based on previous regional revegetation studies, either at Ranger or elsewhere, and response of the savanna ecosystems to disturbance. The model should also consider scenarios (e.g. fire and weeds) that capture key aspects of revegetation establishment and natural disturbances. This information should also be used to identify and plan for management of risks and should form the basis for design and assessment of monitoring programs and results.	BOTH

ECOSYSTEM RESTORATION REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
ESR6	Long term viability	Ecosystem Sustainability	ESR6. Understanding the impact of contaminants on vegetation establishment and sustainability	ESR6A. What concentrations of contaminants from the rehabilitated site may be available for uptake by terrestrial plants?	Exposure of vegetation (both revegetation and existing native vegetation) to contaminants could occur from a number of sources on the rehabilitated site, such as waste rock, contaminated soils and groundwater. Integrated surface and groundwater modelling should identify areas of the rehabilitated site that may act as potential hotspots for increased concentrations of contaminants (see KKN WS1A), such as magnesium sulfate. The concentrations of contaminants available for uptake by terrestrial plants needs to be understood in order to assess whether there may be a risk to vegetation establishment and long term sustainability. For waste rock, which represents an unnatural substrate and plant medium, the assessment is conducted separately through KKN ESR7D.	BOTH
				ESR6B. Based on the structure and health of vegetation on the Land Application Areas, what species appear tolerant to the cumulative impacts of contaminants and other stressors over time?	Contaminants and/or other stressors associated with the operation of Land Application Areas have altered and impaired the structure and health of vegetation. While the presence of multiple stressors confounds the ability to isolate specific causes of impaired plant health, the identification of plants tolerant to multiple stressors (including contaminants) may assist in revegetation planning and establishment (e.g. selection of species best suited to locations of contaminant build-up and/or water-logging) and in assessing plant health, over the longer-term).	ERA
ESR7	Long term viability	Ecosystem sustainability	ESR7. Understanding the effect of waste rock properties on ecosystem establishment and sustainability	ESR7A. What is the potential for plant available nutrients (e.g. nitrogen and phosphorus) to be a limiting factor for sustainable nutrient cycling in waste rock?	There are likely to be substantial differences between waste rock and natural soils in nutrient concentrations (e.g. P, N, Mg, exchangeable K and S) and rhizobia/mycorrhizal fungi available to plants. Combined with a potential lag in the timing at which effective nutrient cycling processes develop in the waste rock, nutrient deficiency may impair the establishment and sustainability of healthy vegetation communities. Targeted monitoring of processes, including soil available nutrient levels and plant nutrient status in established vegetation, compared to levels in soils and plants in reference sites, can provide evidence (i.e. empirical data) of progression to a self-sustaining nutrient cycle. This information will assist in determining whether an active nutrient maintenance regime may be required for a period of time following rehabilitation.	ERA

ECOSYSTEM RESTORATION REHABILITATION THEME

KKN No.	ER Link	Category	Title	Questions	Description	Responsibility (SSB/ERA/BOTH)
				ESR7B. Will sufficient plant available water be available in the final landform to support a mature vegetation community?	Plant available water in waste rock substrate may be limited. Studies on the trial landform have demonstrated water holding capacity of the landform is comparable to the natural reference system. Despite uncertainties in measurements and modelling, the trial landform studies indicate that the waste rock of 4 m thickness may support mature vegetation similar to adjacent areas over short dry seasons but possibly not during longer dry seasons. Further information is needed to determine the availability of water in the waste rock substrate, such as: • influence of waste rock depth on water holding capacity • water availability at greater depths (e.g. 4-8 m) and ability of plants to access this (e.g. maximum rooting depths) • influence of waste rock particle size and pore spaces • contribution of understorey to evapotranspiration rates • uncertainty associated with water balance models and sensitivity of input parameters. These factors will need to take into account location (e.g. elevation and aspect) on the final landform.	ERA
				ESR7C. Will ecological processes required for vegetation sustainability (e.g. soil formation) occur on the rehabilitated landform and if not, what are the mitigation responses?	There is uncertainty about whether key ecological processes required to support sustainable vegetation communities will occur on the rehabilitated landform. It has also been assumed that rapid weathering of waste rock will occur to form rudimentary soil materials but there is little information to demonstrate that this will be applicable across the rehabilitated site (i.e. all types of waste rock materials). This information can be used to determine whether specific mitigations may be needed (e.g. addition of fines, mulch).	ERA
				ESR7D. Are there any other properties of the rehabilitated site that could be attributed to any observed impairment of ecosystem establishment and sustainability, including vegetation and key functional groups of soil fauna?	Apart from plant available water and nutrients, other factors need to be identified in the event that ecosystem establishment and sustainability are impaired. These factors may include, for example, sub-optimal light conditions for tubestock or water-logging of the landform at initial planting.	ERA

ECOSYSTEM RESTORATION REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	**Responsibility (SSB/ERA/BOTH**
ESR8	Long term viability	Ecosystem Sustainability	ESR8. Understanding fire resilience and management in ecosystem restoration	ESR8A. What is the most appropriate fire management regime to ensure a fire resilient ecosystem on the rehabilitated site?	Fire can present a significant risk to long term sustainability of restored ecosystems. The current strategy is to exclude fire from revegetation areas for the first 5-7 years following initial planting, followed by the gradual introduction of fire to rehabilitated areas. With the large spatial extent of fires in the region, management of fires is a cross-jurisdictional issue and needs to be managed for ecosystem restoration success at multiple scales. More specific information is needed to determine the most appropriate fire management regime over time, from initial introduction to a regime that is similar to surrounding areas, including consideration of sensitive plant and animal species. Recent research in Kakadu National Park that modelled the effects of fire regimes on overstorey population dynamics would be particularly relevant to this knowledge need.	ERA

CROSS-THEME REHABILITATION THEME						
KKN No.	**ER Link**	**Category**	**Title**	**Questions**	**Description**	
CT1	Biodiversity and Ecosystem Health	Risk	CT1. Assessing the cumulative risks to the success of rehabilitation on-site and to the protection of the off-site environment.	CT1A. What are the cumulative risks to the success of rehabilitation on-site and to the off-site environment?	It is important to assess cumulative risk as examining risks individually does not address the interaction between risks and their iterative effects. An integrated conceptual model will capture the interactions between multiple risks (e.g. landform stability, revegetation and contaminant exposure) and assessment endpoints (receptors). The integrated model and assessment will be continually tested and improved as part of best practice and include outputs from all other KKNs.	BOTH
CT2	World Heritage values	Heritage Values	CT2. Characterising World Heritage values of the Ranger Project Area	CT2A. What World Heritage Values are found on the Ranger Project Area, and how might these influence the incorporation of the site into Kakadu National Park and World Heritage Area?	There are areas within the Ranger Project Area that exhibit World Heritage Values for which Kakadu is listed, and documentation of these may assist decision-makers in incorporating the site into Kakadu National Park once closure has been achieved.	BOTH