MOUNT RAWDON ENVIRONMENTAL EVALUATION

Stage 3: Final Report

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Executive Summary

Mount Rawdon Operations has performed an investigation, in the form of three reports, that explains groundwater concentrations of Chemicals of Potential Concern (COPC) in its mining lease (the Mine). This is the third and final report, which summarizes the earlier reports and provides a strategy for mitigating risks of groundwater contamination that may occur at the Mine.

A conceptual model was developed from existing and well-established models of dryland salinity in the Central Burnett, which identifies landscape processes that naturally concentrate solutes as a function of groundwater movement through catena soils accompanied by evapo-concentration. The spur and valley landscape of the Mine dissects a variably weathered and structurally complex volcanic sequence and straddles the headwaters of two sub-catchments. Groundwater is contained in weakly connected compartments that are recharged by rainfall, and dewatered by evapotranspiration, which influences the concentrations of solutes in groundwater that follows topographically controlled flow-paths.

The conceptual model identified six chemical elements and compounds that structured the observed variations in groundwater solutes at the Mine. These were cyanide (a compound used in ore processing, which provides a tracer for seepage of process water from the TSF), nitrate-N (an element associated with blast residues, which provides a tracer for seepage from blasted rock noting that there are other landscape sources for nitrate-N), sulfate and arsenic (chemicals associated with ore mineralisation that may or may not be mined), sodium and bicarbonate (chemicals associated with rock weathering and landscape processes). Principal Components Analysis and Classification was used to identify groundwater associations with mining source terms representing seepage from waste rock and tailings, and hypotheses were developed that were tested by statistical analysis of groundwater data, hydrogeological modelling, groundwater transport modelling and surface water modelling.

The result of the statistical tests was a decision tree to distinguish groundwater data that was possibly mine-affected from groundwater with naturally elevated COPC concentrations, which required further testing using:

- Hydrogeological modelling: to identify whether the exceedences were found along potential flow paths from mining sources.
- Geochemical Transport Modelling: to confirm that the solute concentrations observed in receiving groundwater bores can be justified as being derived, at least in part, from mining sources
- Surface Water Modelling: To confirm that solutes concentrations predicted to exfiltrate into ephemeral creeks, and which are subsequently mobilised as flow pulses along the creeks during rainfall spates, are of environmental concern.

The outcome of the Environmental Evaluation is a modelling platform that has been used to develop a strategy that mitigates the risk of solutes generated by mining activity causing present and future environmental harm.

1 Introduction

1.1 Purpose

Mount Rawdon Operations (MRO) is required to control and prevent seepage from mine operations to groundwater and surface waters under Environmental Authority EPML00712113 (EA) at Schedule E, Conditions E18 – E27. In 2018 a Notice to commission an Environmental Evaluation STAT 1264 EPML00712113 (the EE) was issued to MRO by the Queensland Department of Environment and Science (DES). The Notice discusses exceedences of trigger values listed in the EA that had been observed in some groundwater monitoring bores, and seeks to better understand the natural and potential mine influences leading to the results. To date Mount Rawdon Operations has provided two reports to DES to address the Notice:

- Stage 1 submitted in November 2019 characterised groundwater resources, and identified potential contaminant sources relevant to the Mine. This report showed that the dominant underground hydrogeological system is in variably weathered, structurally complex, volcanic sequence, and described the aquifer features relevant to groundwater storage and transmission. This information was synthesised into a conceptual hydrogeological flow model for the Mine. The spatial location of environmental values that depend on the hydrogeological system, and the sources of all potential contaminants that are contacting, or historically have contacted, receiving waters were identified.
- Stage 2 submitted in October 2020 mapped the potential for contaminant migration at the Mine. This report discussed the pathways and mechanisms for the release and transport of potential contaminants from mine operations to groundwater, the aquifer dimensions that potentially facilitate groundwater movement (inferred from geophysical images that were reported in ASST, 2020), and the current and future extent of solute breakthrough in groundwater towards receptors. Solute transport modelling was used to assess options to prevent further contaminant releases from the premises to groundwater, and reduce contaminant concentrations in groundwater.

Stage 3 of the EE requires the following:

- *Item 17. Provide to the department a final report on the environmental investigation that must include:*
    - *a) The methodology applied in conducting this investigation, including any assumptions relied upon and their justification;*
    - *b) A description of the conceptual groundwater flow model, including a visual representation;*
    - *c) A description of the solute transport model, including sources of uncertainty, calibration and sensitivity analysis, and a visual representation;*
    - *d) Graphical presentation of data analysis conducted in Requirement 12*
    - *e) The results of any studies, assessments or analyses that were carried out during this investigation;*
    - *f) Outcomes, findings and conclusions which are scientifically supported by the investigation results and are provided with clear explanations and justifications for any assumptions made;*
    - *g) The electronic raw data used for all data analysis, modelling, and graphical summaries with a clear indication of where the data was used.*

The Stage 3 report (this report) addresses this deliverable by consolidating the methodology and outcomes of the earlier reports, and providing a summary report of the EE.

1.2 Site description and history

The groundwater bore field in the Mount Rawdon Operations Mining Lease (the Mine) surrounds a Tailings Storage Facility (TSF), a processing area, an open pit and waste rock dumps. Groundwater monitoring bores were constructed downslope and along drainage lines leading from mine operations, and have recorded groundwater quality during the history of the Mine. The Mine is thought to contain a number of semi-confined groundwater compartments. An overview of the Mine is shown in Figure 1.

TSF construction started at the end of December 2000, with tailings discharge into the TSF commencing in January 2001. The TSF was originally built as a valley type storage in a small tributary of the Perry River (Rawdon Creek) located north-west of the processing area. It was formed by a primary embankment (Northern Embankment), an embankment on the southern end that extends around the eastern perimeter (Southern Embankment) and a saddle embankment to the west (Western Saddle Embankment). The TSF has been built in successive stages (stages 1 to 5) to maintain the freeboard requirements stated in the EA, and has evolved into what is now a 150 Ha turkeys nest structure at the headwater of Rawdon Creek.

Mining generates about 7Mt of waste rock and sub-economic ore each year that is stockpiled permanently on two waste rock dumps near the mine, with waste rock disposal at the Northern Waste Rock Dump (NWRD) commencing in March 2002. The NWRD is located south-east of the TSF and adjacent to the pit, and the West Waste Rock Dump (WWRD) that commenced construction in June 2014 is located south of the TSF and west of the Pit. The waste rock handling procedure originally proposed for Equigold (then owners of the Mount Rawdon Mine) was reviewed in 2003 by RSG (2003) after low pH water (3.45 – 6.9) and elevated Cd, Cu and Zn concentrations relative to the EA requirements were observed in the Goldings Sediment Dam overflow that was located upgradient of the current WD1 interception dam, and is now buried under the NWRD. A revised classification and waste handling procedure that accommodates for the heterogenous mineralogy of waste rock and applies the precautionary principle (i.e. the procedure is conservatively biased towards minimising environmental harm) was developed after this environmental incident, which has proved to be successful and is the basis of the waste rock handling procedure currently used by Mount Rawdon Operations. Features that underpin the efficacy of revised waste rock handling procedure are:

- An acid mine drainage model generated from a block model that predicted tonnages of waste rock in 5 categories (A-E) based on modelled acid producing potential. Most Net Acid Producing Potential (NAPP) is associated with categories A-C (i.e. S% = 2.6±0.5 %C) that represents about 80.2MT of rock. Dacite and breccia are expected to be potentially acid forming rock (PAF) whereas unweathered tuff is expected to be non-acid forming rock (NAF). Most waste rock (about 70%) and ore (about 80%) are expected to be NAF.
- Ongoing monitoring and review of the acid potential of mined rock for C% and S%.
- In-pit mapping and sampling to ensure the adequacy of the acid potential model.
- Collection of further data in model regions where significant divergence was noted between the in-pit data and model coding.

As such the revised waste rock handling procedure is based on acid base accounting of total sulfur and carbon, and uses the block model to identify the different rock types across the mine to inform selective handling and placement of excavated rock. The application of this procedure is critical for preventing contaminant releases from the Mine to groundwater, because it addresses seepage generation of mining affected groundwater at the source. Excavated PAF waste rock is either placed below the water blanket in the TSF where it is encapsulated by acid neutralising tailings, or placed in the NWRD under NAF waste rock and a compacted clay cap. The NWRD has been designed to produce neutral mine drainage (NMD), with any potential acid runoff formed in the centre of the dump passing through the cover of neutralising rock. The outer batter walls of the TSF as well as the WWRD are used to store NAF, which weathers more slowly than PAF, and hence by design the WWRD and the outer batter walls of the TSF generate less neutral mine drainage then the NWRD.

An investigative program was implemented to fully understand acid mine drainage (AMD) risks associated with the Mine (RGS 2009, RGS, 2017). RGS (2009) considered that the revised waste rock strategy is reasonably successful and recommended that Evolution maintains the current conservative approach because of uncertainty associated with the geochemical nature and classification of some materials. RGS (2009) reported that low risk waste rock materials can be used

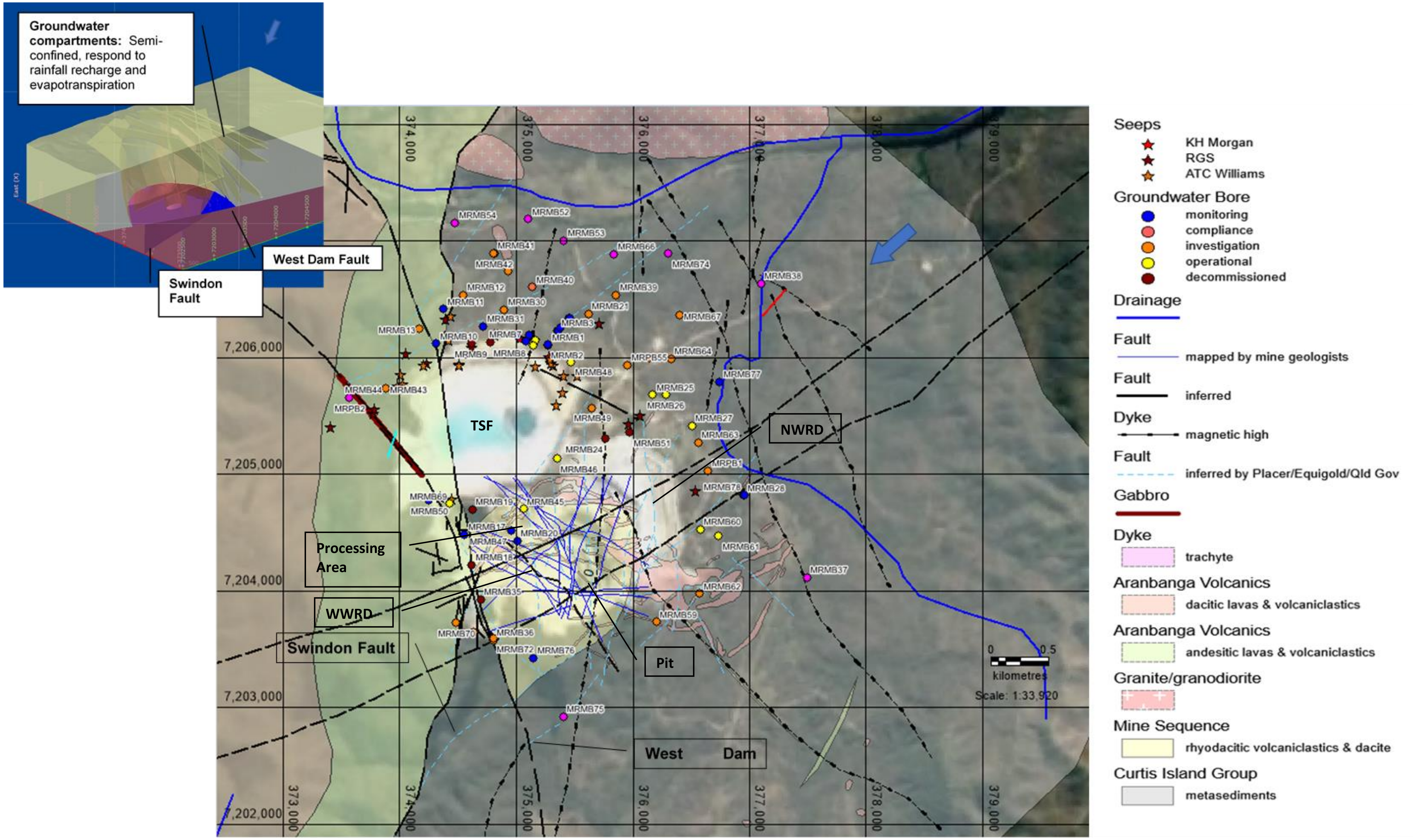

Figure 1: Sample location map showing groundwater monitoring bores in the context of lithology, and structures that contribute barriers to groundwater flow [1]

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[1] Plan view and an angled view from below ground level (insert) showing inferred fault structures (e.g. West Dam Fault) that contribute to groundwater compartmentalization. The blue arrow references the below-ground direction of view of the insert.

in the outer batter wall of the TSF and as part of the final cover over tailings in the TSF (i.e. as armoring against erosion). They showed that NAF waste rock has negligable sulfur content (0.5% of which 0.4% is potentially acid generating), has elevated acid neutralising capacity relative to the maximum potential acidity generated during oxidative weathering (ANC:MPA ratio of 8.3) and has a high factor of safety with respect to AMD, NMD or Saline Drainage (SD). The Cation Exchange Capacity (CEC) and exchangeable sodium percentage (ESP) results for waste rock and low grade ore indicated that low risk waste rock materials should support revegetation as part of rehabilitation activities.

Stormwater and seepage water egressing from the TSF and the NWRD consistently report elevated concentrations of sulfur and nitrate, and occasionally report elevated concentrations of dissolved heavy metals. Seepage water from the TSF also reports elevated concentrations of cyanide. Mine affected water is captured and is either reused in the processing circuit or removed by enhanced evaporation, and has not been released to downstream receptors except for the few uncontrolled releases following high rainfall events that were reported to regulators at the time, and are described below. Seepage management at the Mine is described by the Seepage Management Plan (Evolution, 2019).

The seepage collection system beneath the Northern Embankment of the TSF began receiving water some six months after initial tailings deposition, and has historically been managed by the following actions:

1. A seepage drain that collected and returned water to the TSF.
2. Sediment Dam 1 (SD1) that provides secondary containment and returns water to the TSF. Bore MRMB1 was equipped with a pump to intercept deep seepage pathways in fractured rock below the Northern Embankment.
3. A second seepage drain that flows into SD1, with subsequent addition of 3 sumps to the drain to enhance recovery.
4. A dewatering bore installed on the TSF, immediately upstream and adjacent to the clay cut off wall of the Northern Embankment on the Stage 2C (RL 150.0m) crest to transfer water to SD1. The dewatering bore extended some 7m into the upstream rock fill shell of the Stage 1/Stage 2 (starter embankment). The purpose of this bore was to: (a) Remove excess water in the rock fill of the Starter Embankment, which is likely to be applying hydraulic head potential against the Northern Embankment core and cut-off key; (b) Allow greater dewatering of the tailings beach adjacent to the Northern Embankment, thereby enhancing tailings consolidation and associated beach strength. Siphoning was halted in late 2016, and the system was revaluated in light of alternative seepage and embankment stabilisation measures.
5. A downstream trench (Marty's trench) that transfers seepage to SD1.
6. A cutoff wall of compacted clay between the second seepage drain (item 3) and Marty's Trench (item 5).
7. Bore MRMB73 was equipped with a pump to intercept deep seepage pathways in fractured rock below the Northern Embankment towards the Northern Seepage Zone.
8. In 2017 the second seepage drain (item 3) was reconditioned. The sumps (item 3) and dewatering bores (items 2 and 7) stopped being used because of the Stage 5 TSF lift, and were replaced by the Downstream Seepage Interception Drain, which was commissioned along the downstream toe of the Northern Embankment at the Stage 5 TSF lift (the final life of mine toe of the Northern Embankment). The Downstream Seepage Interception Drain penetrates weathered basement rock by up to 4m to enhance the overall seepage recovery. This drain extends from the north west flank of the TSF and directs recovered seepage to SD1. Marty's trench (item 5) and the second seepage drain (item 3) still intercept shallow seepage below the Stage 5 buttress wall, which is directed to SD1. The relative level of the base of the Downstream Seepage Interception Drain is designed to limit the effective hydraulic head potential of seepage through the Northern Seepage Zone to about 115m AHD.
9. Containment and diversion of seepage along a backfilled gully near MRMB10. This seep is now diverted into the Seepage Interception Drain (item 8).

TSF seepage under the Western Embankment is being managed by:

- A seepage intersection sump (Well 5) installed downstream of the embankment that returns water to the TSF.
- A seepage collection trench installed downstream of the embankment, which is connected to the sump.
- Dewatering bore MRDB1 that intercepts deep seepage below Well 5.
- Diversion of seepage from Seep 26 to SD1 via the Seepage Interception Drain.
- A second seepage intersection sump (Well 1) installed downgradient of Seep 10 and Seep 26 that transfers seepage to SD1 via the Seepage Interception Drain.
- Dewatering bore MRDB2 installed downgradient of Seep 10 and Seep 26, and dewatering bore MRDB3 installed downgradient of Seep 20, which transfer seepage to SD1 via the Seepage Interception Drain.

TSF seepage under the Southern Embankment/South-Eastern Embankment is being managed by:

- The Southern Embankment seepage collection system, which drains into the South East Dewatering Well (Stan’s Well) and returns water to the Process Dam.
- Relocating the decant, and the decant-access-causeway. The existing decant (current up to 2020) is being decommissioned and will be filled with tailings to seal the hydraulic connection between it and the Southern Embankment via the existing decant-access-causeway.

The South Dam collects runoff of seepage expression that has flowed through the Southern Embankment and stormwater that has flowed through the outer batter wall of the Southern Embankment.

Solutes generated in the NWRD are carried in stormwater that flows through the rock pile and hydrologically connects with four existing drainages on the eastern flank of the NWRD. These seeps are managed by:

- Four water retention dams (WD1, WD2, WD3, WD4) that intercept stormwater. Captured water is transferred to WD1 from WD2, WD3 and WD4, and then returned to the Raw Water Dam for use as process water or removal by mechanical evaporation.
- Seepage interception systems downstream of WD1 and WD2, which drain into sumps that return seepage water to the retention dams.
- Dewatering bores (MRMB26 and MRMB27) downstream of the seepage interception systems that return intercepted seepage in groundwater to WD1 and WD2 respectively.

Neutral mine drainage generated in the WWRD flows south into West Dam. A seepage intersection sump below the downstream toe of West Dam returns intercepted seepage in groundwater to the dam. Seepage water collected by West Dam is removed by enhanced mechanical evaporation.

As such over the 19 year history of Mount Rawdon Operations, there has been reasonably effective minimisation and containment of seepage from mine operations to groundwater and surface waters. However there have been a few uncontrolled releases into drainage lines from the Mine, which were reported to the relevant government agencies at the time of the incidents and were mitigated. These historical environmental incidents and corrective actions are listed as follows:

- On the 4-5/2/2003 after intense rainfall (227 mm in the first week of February and 336.75 mm for the whole of February) there was leaching of solutes from the NWRD and overtopping of Goldings Sediment Dam into Twelve Mile Creek. In addition low pH water reported to Rawdon Creek. Corrective actions by Equigold involved revising the waste rock handling procedure to avoid AMD generation (as described above), and improving the capture of drainage from the NWRD by installing the four Waste Rock Dams, and liming the catch drains at the NWRD toe.
- On the 2/10/2011 desludging of SD1 is thought to have damaged the seal of this dam. Between 1-11/12/2011 SD2 water became more like SD1 water, due to increased hydraulic connectivity between the dams following damage to the SD1 clay lining. Between the 12-13/12/2011 there was overtopping of SD2 (which was originally a farm dam and not designed as an interception dam), which released the mine affected water in SD2 into Rawdon Creek.

SD2 was dewatered between 6-10/01/2012 to lessen the risk of overtopping, but nevertheless SD2 still overtopped between 24/01/2012 and 09/02/2012. The corrective action was to restore the lining of the damaged clay bund in May 2012 after the wet season had finished.

- In January 2013 the Waste Rock Dams overtopped. In 2014 remedial works on WD1 and WD2 were performed by raising the crests and spillways of these dams to increase their storage capacities. Additional seepage works were installed at the downstream toes of WD1 and WD2, to intercept groundwater seepage through alluvium and fractured rock. At WD1 the clay core key was deepened, and a 5 meter deep seepage interception trench was installed. Pump back sumps at WD1 and WD2 were commissioned before the 2014/15 wet season commenced.
- On the 27/11/2015 SD2 overflowed into Rawdon Creek, carrying a recorded sulfate concentration in excess of the EA requirement of 100 mg/L.
- Between 31/3/2017 – 3/04/2017 SD1 and SD2 overtopped into Rawdon Creek. Corrective action commenced in July 2017 with construction of the Downstream Seepage Interception Drain (described above) as part of the TSF Sediment Dam 3 development.
- On the 31/10/2018 Seep 26 was noticed near Well 1, with seepage briefly entering Swindon Creek. Corrective actions involved diverting seepage into the Downstream Seepage Interception Drain and installing dewatering bores (MRDB1, MRDB2, MRDB3).
- On the 19/03/2019 stormwater mixed with seepage and bypassed Well 1 again briefly entering Swindon Creek. Corrective action included improving the reliability and automation of pump operation associated with Well 1, as well as the intention to extend the Downstream Toe Interception Drain to capture seepage and stormwater (one of the mitigation actions proposed by this report).

Groundwater and surface water monitoring programs at Mount Rawdon Operations allow early detection of mining influence, and guide the capture and mitigation of mining affected water. The Mount Rawdon Groundwater Management and Monitoring Plan (Evolution, 2018) describes the groundwater monitoring program for the Mine. Groundwater monitoring bores have historically been constructed along drainage lines leading from mine operations. If and when seepage migrates downgradient and is intercepted by a monitoring bore, another bore is commissioned further down the flow path to track the efficacy of the seepage recovery. Bores furthest from the mine that are not mine affected are used for compliance purposes (called “Compliance Bores”), and most are located near the mine lease boundary adjacent to Perry River or Mingam Creek (Figure 1).

In addition to the bores described above, which were deliberately placed along flow paths leading from mine operations, groundwater bores have been developed from geological or geotechnical bores drilled for other purposes. These screen in fractured rock, and some are extremely low yielding without fully refilling after being purged (called “dry” bores). Dry bores have poor water quality that trigger resampling, but have insufficient yield to allow proper sampling involving purging.

The Receiving Environment Monitoring Program (REMP) assesses the ecological condition of receiving surface water environments, and tracks the performance of the seepage management program described above. Seven years of surface water monitoring between 2013 and 2020 has consistently indicated a low probability that mining activities have influenced stream macroinvertebrate assemblages. In April 2020 in investigation for the presence of subterranean fauna living in groundwaters of the Mine was performed, and while physico-chemical groundwater conditions suitable for stygofauna were recorded, there was low occurrence and limited taxa for subterranean fauna found in groundwaters of the Mine.

2 Methods

2.1 Overview

This EE has investigated the natural background variation of groundwater quality and mining influences that may have led to exceedences of the guideline limits stated in the EA. To achieve this a literature review was performed to consolidate the current understanding of the history of the Mine (discussed in section 1.2), as well as groundwater quality at the Mine. The literature review was supported by statistical analysis of all groundwater data obtained during mining operations, using multivariate statistics to parse out possible mining-related contributions of solutes to groundwater from the natural variation in solutes in groundwater at the Mine. This preliminary analysis was followed by univariate statistical analyses describing the relative concentrations of solutes, and the temporal trends of solute concentrations, in groundwater across the Mine. From these analyses it was possible to develop hypotheses about potential mining or non-mining influences that caused the exceedences of guideline limits, and a conceptual model was developed.

Knowledge gaps were identified from the literature and data review, and targeted investigations, were performed to address these gaps. The targeted investigations included (1) a geophysical survey to better understand seepage pathways from the TSF and the NWRD (ASST, 2020), (2) shallow groundwater collection of seepage from the NWRD to better define the waste rock source term, and (3) geochemical tests to better understand weathering of waste rock, and how solutes react with solid mineral phases in fractures and pore spaces of the aquifers at the Mine. These targeted investigations were used to inform the solute transport model.

The conceptual model was used as the basis for subsequent models of groundwater flow, solute transport and surface water quality. These models were used to test the concept of seepage from the TSF and the Waste Rock Dumps towards ephemeral streams, and explain the observed exceedences of COPC reporting to monitoring bores along five flow pathways at the Mine.

The existing guideline limits of the EA were reviewed by identifying the 80th percentile concentrations for COPC's in groundwater bores unaffected by mining, and then applying a decision tree based on the temporal trends of solutes identified from the conceptual model to be tracers of mining activity. The revised guidelines proposed by this EE were developed to ascertain whether the exceedences are likely to be associated with mining operations or with natural variations of groundwater chemistry.

If an exceedence was identified as likely to be mining related, mitigation actions were proposed. The efficacy of the mitigation measures were evaluated using the solute transport model and the surface water model.

To address item 17a of the EE, the methods and assumptions used to better understand the natural background variation of COPC in groundwater, and the non-mining and mining influences that led to the exceedences of EA guideline limits, are discussed below. Specifically addressed are the approaches used to develop the conceptual model, the hydrogeological model, the geochemical transport model, the surface water model, and the water quality statistics used to revise the guideline values.

2.1 Conceptual Model Development

Metadata, reports and archives of MRO and National Heritage Trust reports were reviewed to develop the conceptual model as follows:

2.1.1 Literature Review

National Heritage Trust reports on landscape salinity, a known feature of the sloping granitic terrain in the central Burnett region of South East Queensland (Douglas and Cox, 1994, Queensland Government, 2003), describe landscape processes that are likely contributing to some of the high electrical conductivity and sulfate measurements observed at the Mine. Soil maps (Queensland Government) indicate that sodic soils occur in the mine lease, which is consistent with the concept of salt generation in this landscape.

Reports identifying the nitrogen fixing properties of acacias help to explain high nitrate concentrations in groundwater from some parts of the mine lease (Brockwell et al., 2005, Esslemont et al., 2007).

In the mine lease geochemical exploration data involving soil assays and exploration bores were reviewed (Equigold, 2008; Evolution, 2018a; Evolution, 2019), to identify mineralisation in unmined parts of the lease that may explain exceedences of trigger values in some groundwater bores. Also in the mine lease, geochemical investigations involving isotopes have identified the age of groundwater near to and downslope of the TSF (Leaney, 2006), and the provenance of nitrate and sulfate in some groundwater bores (Northern Resources Consultants, 2015). Finally, annual groundwater reports provided useful records of groundwater condition throughout the life of the mine, and of mining-related incidents that resulted in exceedences of trigger values (discussed in section 1.2).

2.1.2 Characterising Aquifers in the Mine Lease

To understand the stratigraphy, fault and fracture propensity in the mine lease, soil maps and geological information were viewed spatially using 3D-GIS (Geology Leapfrog Viewer).

Three aquifers are recognised in the mine lease (Klohn Crippen Berger, 2010): soil and small pockets of alluvium in drainage lines (of limited extent and thickness), weathered rock (saprolite and oxidation transition zones) of varying thickness and subject to drainage on sloping land after rainfall events that recharge the aquifer, and the fractured rock aquifer. The thickness of the aquifer profiles were deduced from (a) the base of weathering, and base of chemical weathering recorded in logs of exploration bores and hydrogeological bores, and (b) geophysical (electrical resistivity imaging) profiles. Because groundwater bores screen in both the weathered rock aquifer and/or fractured rock aquifer, the groundwater bores screen variably weathered, fractured rock. In practical terms groundwater bores that screen 5 – 15m below the surface were grouped as regolith, and bores that screen 16 - 50m below the surface were grouped as fractured rock aquifer.

Primary minerals that potentially weather in contact with groundwater, and their derivative secondary minerals, were identified from (a) waste rock and (b) the fractured rock aquifer (Ward, 2019). These minerals contribute solutes to groundwater, or remove solutes from groundwater, during groundwater flow through the aquifer.

To understand the hydraulic conductivies of the regolith and fractured rock aquifers, slug tests were performed on bores using the Hvorslev method. Some hydraulic conductivities measured by Northern Resources Consulting were also used. From these measurements, in combination with known aquifer thicknesses obtained from bore logs, the transmissivities of the aquifers were described. In combination with identification of dynamic viscosity and density parameters of brackish water in the expected temperature range (10 - 30^{o}C) the permeabilities of the aquifers were described.

To understand the hydraulic conductivity of the soil aquifer, field descriptions of the exposed soil profile north of the TSF were performed, and soil types were identified by hand texturing. Soil Water Characteristics software developed by the USDA Agricultural Research Service (Saxton & Rawls, 2006) was used to estimate the hydraulic conductivities, bulk densities and porosities of soil horizons down the soil profile from hand-textured samples.

2.1.3 Statistical Analysis of Groundwater Data

Geochemical source terms for (a) tailings, (b) mine waste-rock (used to build the TSF batter wall) and (c) non mine-affected groundwater, were established from historical records and from direct measurements of seepage from the NWRD that were collected for the EE.

To understand how concentrations of Chemicals of Potential Concern (COPC) in groundwater vary at the Mine, multivariate statistics (ordination and classification) using PCORD software (McCune and Mefford, 2018) were used to discriminate groundwater data from mine affected water, and identify mixing of these water types. This was done initially with median solute concentrations from each groundwater bore and each mining source term (waste rock, decant water), to identify the main variation in groundwater chemistry at the mine lease scale during the history of mine operation. Subsequent multivariate analysis was done using all data for solutes in groundwater, from which detailed interpretation of data variation in subcatchments at finer time scales was obtained.

Ordination was performed using Principal Components Analyses (PCA) of the total variance in groundwater data, which transformed COPC concentrations in groundwater into a smaller set of linear combinations represented graphically on two or three axes to show the principal components underpinning the data variation. The PCA assumed:

- **Adequate Sample Size:** For the median dataset there were 78 bores/mining source terms represented by 6 chemical elements (13 samples per element) and for the complete database there were 3190 bores/mining source terms represented by 6 chemical elements and compounds (chemicals).
- **Correlations:** The initial group of 21 chemicals was reduced to 6 representative chemicals, by systematically removing weakly correlated chemicals to achieve a minimal subset of chemicals that were strongly correlated and meaningful. The 6 chemicals were cyanide (a compound used in ore processing, which provides a tracer for seepage of process water from the TSF), nitrate-N (an element associated with blast residues, which provides a tracer for seepage from blasted rock noting that there are other landscape sources for nitrate-N),

sulfate and arsenic (chemicals associated with ore mineralisation that may or may not be mined), sodium and bicarbonate (chemicals associated with rock weathering and landscape processes).

- **Linearity and lack of outliers:** Data representing each chemical were normalised as well as could be achieved, then range standardised to a value between 0 and 1. As such the data were as normally distributed as could be achieved.

The ordinations at the mine lease scale as well as at the subcatchment scale explained at least 80% of the data variation with eigenvalues less than 1, indicating that the analysis was sound.

Classification was performed on the normalised and range standardised data by polytheic heirarchical agglomerative cluster analysis involving Euclidean distance, with flexible beta group linkage to limit the low amount of chainage evident in the classification (<2%). Euclidean distance was chosen to be consistent with the distance metric used for the Principal Components Analysis.

Using this approach, groundwater groups within the Mine were identified and shown spatially on a map to indicate the proximity of the groundwater groups to mining operations and to inferred groundwater flow paths (Figure 4). Hypotheses were set to address whether exceedences of trigger values were potentially associated with mining-related source terms, or with naturally varying solute concentrations in groundwater from the mine lease.

Following the multivariate analysis further statistical testing using univariate and graphical approaches (section 2.4), geochemical transport modelling (section 2.2) and surface water modelling (section 2.3) were used to develop and test the hypotheses.

2.1.4 Hydrogeological Model Development

Groundwater flow rates along flow paths A-A', … E-E' accross the mine lease were described using hydrogeological software developed by the USGS (MODFLOW-SURFACT). Northern Resource Consultants (2019) calibrated the hydrogeological model against 31 water level targets with a scaled root mean square error of 7.8%, indicating that the calibration was acceptable. The model development and calibration is described in Northern Resource Consultants (2019).

2.2 Geochemical Transport Model Development

2.2.1 Speciation Modelling

The groundwater chemistry of the aquifer was evaluated using chemical speciation modelling by VISUAL MINTEQ verson 3.1 software developed by the USEPA (Gustafsson, 2018), informed by the solute chemistries measured in the groundwater. This allowed identification of weathering minerals that are stable in the aquifer.

The speciation model referenced historic records of groundwater compositions, including pH and, where available Eh or dissolved oxygen. Both redox potential (Eh) and dissolved oxygen (DO) have been reported since 2018. Groundwater records of these field parameters can be used as approximate indicators of the prevailing redox condition, and on this basis were considered in the aqueous speciation calculations.

Groundwater constituents and parameters considered in the Visual MINTEQ simulations were major and minor ions (Na, Ca, Mg, K, Cl, F, SO_4, HCO_3), field filtered trace elements (Al, As, B, Cd, Cu, Pb, Zn, Cr, Fe, Mn, Mo), CN, NO_3_N and field measured pH. A temperature of 25^{o}C was assumed for all simulations.

Silica (SiO_2) and phosphorus (P) concentrations in groundwater are useful for understanding the weathering of host rock along the flow-paths. Because SiO_2 and P concentrations have not been historically reported at MRO, SiO_2 and P groundwater concentrations measured in 2018 from across the Mine were used where feasible. The 2018 investigation used low flow pumping technology to obtain minimally disturbed groundwater.

Chemical speciation is an important control of the activities of CPOC such as copper, lead, cadmium and zinc, and oxides of arsenic and chromium, which in turn influences their chemical interactions between groundwater and the host aquifer and potentially their toxicity. As such the chemical species and activities identified by the speciation calculations describe the relative speciation of element concentrations measured in groundwater, which is key to the definition of "critical concentration" used in the EE.

2.2.2 Inverse Modelling

Inverse modelling of groundwater mixing pathways was performed using hydrogeochemical software developed by the USGS (NETPATH for Windows). Model inputs were (a) geochemical source terms for tailings or waste rock seepage (listed in sections 7.5.2 and 7.5.3 of the Stage 2 report), (b) groundwater concentrations reported bores along groundwater flow paths reported by Northern Resources Consultants (2019) and (c) weathering minerals.

NETPATH (Plummer, Prestemon and Parkhurst, 1994) was used to reconcile the stoichiometry of solutes measured in groundwater bores along flow-paths towards ephemeral creeks in the context of landscape processes associated with dryland salinity as identified by the conceptual model (refer section 4 of the Stage 1 report), where a source term of mining-affected water (fluids from the TSF or NWRD) mixes with native groundwater. The model accommodates for interactions between groundwater and the host aquifer, by enabling dissolution and precipitation of minerals associated with rock weathering and allowing evaporation and dilution. The model input for NETPATH simulations included:

- **Components:** major ions (Na, Ca, Mg, K, Cl, F, SO_4, HCO_3), a limited range of field filtered (nominally dissolved) trace elements (Al, B, Fe, Mn), NO_3_N, field measured pH, and where available field measured Eh and DO. The source terms for TSF fluid, Waste Rock fluid and native groundwater from the monitoring bore before mine influence (described in Section 3.3 of the Stage 2 report) were entered for a sequence of monitoring bores along flow-paths A-A', B-B' … E-E'. A temperature of 25^{o}C was assumed.
- **Mineral phases relevant to the weathering sequence** (described in Section 3.3 and Appendices 1 – 4 of the Stage 2 report) were used in the model. The dominant weathering process (refer Appendix 1 of the Stage 2 report) relevant to the flow-path was considered, because the model is limited by the number of phases and components. For groundwater bores near the toe of the TSF (e.g. MRMB9, MRMB10), weathering processes inside the TSF that generate neutral mine drainage (pyrite oxidation and dissolution of fresh calcite, feldspar and chlorite) were represented in the model. For groundwater bores further from the TSF (e.g. MRMB12), weathering processes expected to occur in saprolite (e.g. tropical weathering of feldspars and chlorite to clay, iron oxide and gibbsite) were represented in the model. Processes of evaporation, dilution and ion exchange of Ca/Na between charged mineral surfaces and solution, were also represented in the model.

As such NETPATH was used to represent transport of mining - affected groundwater. Models that converged were investigated in more detail by geochemical transport modelling, as described in Section 2.2.3.

2.2.3 Geochemical Transport Modelling

NETPATH model outputs that reconciled were used as a starting point for more detailed solute transport modeling using USGS software (PHREEQC). Because the NETPATH models indicated that rock weathering, surface reactions and evaporation processes were important, these processes were also represented in the PHREEQC models.

Source terms for the TSF and waste rock were mixed in proportions predicted by the NETPATH model. Aquifer transfer of these combined "mining" source terms towards surface water receptors in Swindon Creek, Rawdon Creek, Twelve Mile Creek and Mingham Creek were modelled. Refer to section 3.3.1 of the Stage 2 report for descriptions of how the source terms were combined using PHREEQC. Refer to section 3.4.1 of the Stage 2 report for how the mixed source term was compared with the observed seepage water quality in Seep 26 and the Northern Seep Zone.

Calculation of the Darcian movement of groundwater along flow-paths A-A' … E-E' was based on the steady state hydraulic gradients calculated by the MODFLOW groundwater flow model (Northern Resource Consultants, 2019). The groundwater discharge (m^3/d), the annual seepage velocities (m/y) and the number of years for groundwater flow to reach the creeks as represented in the PHREEQC model were in line with the aquifer hydraulic conductivities and porosities described in section 3.1.2 of the Stage 1 report. Tables of these flow parameters and the cell inputs used for the PHREEQC model are listed in Appendix 5 of the Stage 2 report.

For aqueous solutions migrating across the shallow aquifer, geochemical equilibrium with quartz (SiO_2), goethite (FeOOH), pyrolusite (MnO_2) and atmospheric oxygen was assumed. This assumption was based on field observations that showed the presence of goethite, pyrolusite, silicification and seasonally variable standing water levels in groundwater bores that could add oxygen to the unsaturated aquifer in fresh rainfall recharge. In the saturated zone, however, there is no mechanism to replenish oxygen if groundwater becomes oxygen depleted by any redox reactions. Further to the field observations, speciation modeling indicated that silica precipitation would be commonly expected to occur in aquifers. Modelling predictions of iron and manganese mineral precipitation based on saturation indices in groundwater were reported where relevant.

The possibility of groundwater flowing through the deep, fractured rock aquifer was also investigated in some model scenarios. However, simulated concentrations could not be reconciled with the observed rates and directions of mining-affected groundwater movement, which indicated that groundwater movement was likely to be predominantly through the shallow aquifer.

Surface complexation reactions with hydrous-ferric oxide were modelled, and were equilibrated against the number of moles of iron extracted by Tamm's Reagent, an assumed porosity of 5.5% in saprolite, and mineral particle density of 2.7 kg/L. In short, the PHREEQC model assumed surface adsorption to reactive iron sites according to Dzombak and Morel, 1990.

Cation exchange reactions associated with clays were also included in the model. This allowed for the adsorption of cations from groundwater to clay surfaces and the release of cations from the clay surfaces to solution. The cation exchange capacity assumed in the model was based on measured K, Mg, Na and Ca concentrations adsorbed to aquifer surfaces (refer to the laboratory tests performed on aquifer samples, as listed in section 3.1.1 of the Stage 2 report).

In summary PHREEQC was used to (1) assess and understand the historic migration of mine-affected solutes in groundwater due to activities conducted at the premises, and (2) predict the likely future extent of contaminant migration and impact.

2.3 Surface Water Model Development

Surface water modelling was performed using PCSWMM (Computational Hydraulics International, Guelph, Canada), which is a development of the USEPA surface model SWMM with additional GIS capacity to facilitate rapid and efficient delineation of subcatchments and river channels from digital elevation (DEM) data. Because the Mine is at the headwater of adjacent catchments (the Perry River catchment and the Mingam Creek catchment), the model was built to represent the Perry River Catchment upstream of stream gauging station 136019A, and the ungauged Mingham Creek catchment upstream of Paradise Dam in the Burnett River.

The model is uncalibrated at this stage, therefore predictions of COPC in surface water should be regarded as indicative. In the future the model will be be calibrated against stream gauge data (depth and flow) from the Perry River, as well as against measured solute concentrations in surface water at the Mine.

Model inputs include:

- **Fluvial Geomorphology:**
    - Topography: A Digital Elevation Model clipped to a bounding polygon of the Perry River and Mingham Catchments upstream of stream gauging station 136019A, and the ungauged Mingham Creek catchment upstream of Paradise Dam was developed. The clipped DEM included detailed topography of the mine operations area (from recent drone survey data), on top of detailed topography of the wider mining lease area (ELVIS Ground Layer data), on top of a less detailed DEM clip encompassing the rest of the catchments. The output was a single DEM layer that represented the model area.
    - Subcatchments: The model area has a total drainage of 14,119.6 Ha consisting of 1223 subcatchments, with an average area of 11.5 Ha per subcatchment. There are 755 node links in the stream network, with 754 links that have an average length of 298m and an average slope of 5.25%. The model includes the headwaters of Perry River and Mingham Creek, and excludes the mine operations area from which stormwater and seepage is captured and returned to the processing circuit (refer section 1.2).
    - Infiltration: Horton infiltration was assumed with a decay constant of 4 hr^{-1}, maximum and minimum infiltration rates of 150 and 2 mm/hr respectively, a 7 day drying time (for soil to completely dry) and a maximum infiltration volume of 0.9 mm. 25% of the catchment areas were assumed to be imperveous.

- Channels: Irregular cross sectional areas were calculated using the PCSWMM Transect Creator, which used the DEM model layer to create multiple transect lines along defined channel centrelines from which an average transect was assumed for the link. The transect lines extended 200m on either side of the channel centreline, with vertical measurements taken 1 meter apart. The transect spacings were 100 meters apart. Mannings coefficients ranging between 0.7 and 0.01, and more typically 0.21 – 0.25, were used to describe the roughness of river channels in different parts of the model area.

- **Climate and Rainfall:** Climate and rainfall data entered into the model database are:
  - Site specific data collected onsite at the MRO weather gauge between 17/7/2013 14:00 and 7/9/202017:30. Rainfall (mm volume in 10 minute intervals), daily maximum air temperature (^{o}C) and minimum air temperature (^{o}C) measured at 10 minute intervals, daily evaporation (mm) measured onsite from a pan evaporator at weekly to subweekly intervals and wind speed (km/hr).
  - Pasture Model (P51) data from the SILO database (DES) between 01/01/1889 and 22/03/2018 for coordinates -25.20 151.75. Data included daily rainfall (mm), daily maximum and minimum air temperature (^{o}C) and daily evaporation (mm).
  - Pasture Model (P51) data with drought projection, which was done by adding climate and rainfall data for the driest 15 years on record (01/01/1932 – 31/12/2047) after 31/12/2017. These data were used to support the dry weather scenario between 01/01/2020 and 26/12/2033 described below.
  - Pasture Model (P51) data with high rainfall projection which was done by adding climate and rainfall data for the wettest 15 years on record (01//01/2002 – 31/12/2017) after 31/12/2017. These data were used to support the wet weather scenario between 01/01/2020 and 26/12/2033 described below.

- **Perry River Weir:** The construction plan indicated that it is a trapezoidal weir with the following dimensions: 3.5 m height, 98 meter length, 3.6 sideslope (i.e. right angle triangle with 3.6m base:1m height), inlet offset from the river bed (9.897 m). Assumptions included discharge coefficient (1.45 m^3/s), end coefficient (0.3 m^3/s) and no flap gate. Surcharge was allowed.

- **Groundwater ingress:** Flow inputs representing groundwater exfiltration into the stream links, were entered into nodes of the stream network where hyporheic flow is expected to occur within 3 meters of the creek bed (refer Figure 7). The groundwater exfiltration volumes calculated for water budget zones by the MODFLOW-SURFACT model (Figure 9) were entered into the stream network based on the relative stream length of each link (i.e. stream length in the link/stream length in the water budget zone). The baseline groundwater concentrations for sulfate and arsenic were identified from a groundwater bore near to the relevant stream link as follows:
  - Swindon Creek. Water budget zone 1 received the median background groundwater concentrations for sulfate (250 mg/L) and arsenic (0.003 mg/L) measured in reference bore MRMB13, and no measurable cyanide concentration (0 mg/L). Water budget zone 2 was assumed to receive the median background concentrations measured in MRPB2 (220 mg/L SO4, 0.002 mg/L As, 0 mg/L CN).
  - Twelve Mile Creek: Water budget zones 5 and 6 received the median background groundwater concentrations for sulfate (317 mg/L) and arsenic (0.002 mg/L) measured in reference bore MRMB62, and no measurable cyanide concentration (0 mg/L).
  - Mingham Creek: The downstream section of water budget zone 9 received the median background groundwater concentrations for sulfate (764 mg/L) and arsenic (0.004 mg/L) measured in reference bore MRMB75, and no measurable cyanide concentration (0 mg/L). The upstream section of water budget zone 9 received the median background groundwater concentrations for sulfate (23 mg/L) and for arsenic (0.002 mg/L) measured in reference bore MRMB70, and no measurable cyanide concentration (0 mg/L). A flow weighted concentration of sulfate (380 mg/L) and arsenic (0.002 mg/L) from flows entering from the Southern Drainage and mixing with the upstream Mingham Creek contribution, were added to recognise the elevated sulfate concentration (1530 mg/L) measured in groundwater bore MRMB36

that appears to be high background (refer SO_4/Cl concentration ratio and EC time series graph in Appendix 8.4.6 of the Stage 1 report).

The ingress of mining affected groundwater water from the riparian zones near MRMB44 and MRMB12, were entered into Swindon Creek via the nodes closest to these monitoring bores. Time series data of unmitigated mine influences on groundwater predicted by the groundwater transport model for the A-A' and B-B' flowpaths (refer section 4.2.2) were added to the Swindon Creek water budget through the nodes nearest MRMB44 (the node was titled A-A_seepage_ingress) and MRMB12 (the node was titled B-B_seepage_ingress). These data represent the risk of unmitigated groundwater influence under drought and high rainfall scenarios.

The ingress of mining affected groundwater after mitigation was also modelled. Time series data of mitigated mine influences on groundwater predicted for the A-A' and B-B' flowpaths by the groundwater transport model (discussed in section 6.1) was entered into Swindon Creek through the respecttive nodes (A-A_seepage_ingress near MRMB44, B-B _seepage_ingress near MRMB12). These data represent mitigation of groundwater during drought and high rainfall conditions, and the results were compared with the unmitigated scenarios described above to indicate the efficacy of the proposed management actions.

Four scenarios were run to represent seepage management between 01/01/2020 and 26/12/2033, assuming extreme wet or dry conditions to allow for a range of possible climates. The scenarios were:

- **Wet Unmitigated** – Predictions for sulfate, arsenic and cyanide in Swindon Creek immediately downstream of MRMB44 (C231_2) and MRMB12 (C166), as well as predictions for the Perry River immediately downstream of the Swindon Creek confluence (C14). These results foreshadow the current seepage management during a relatively wet period between 2020 and 2033
- **Wet Mitigated** – Predictions for surface water quality as described above, foreshadowing the proposed mitigation actions during a relatively wet period between 2020 and 2033
- **Dry Unmitigated** - Predictions for surface water quality as described above, foreshadowing current seepage management during a relatively dry period between 2020 and 2033
- **Dry Mitigated** - Predictions for surface water quality as described above, foreshadowing the proposed mitigation actions during a relatively dry period between 2020 and 2033

It is expected that additional mitigations associated with Mine Closure will occur between 2020 and 2033, and these can be modelled as future scenarios.

2.4 Water Quality Assessment

2.4.1 Descriptive Water Quality Statistics

Graphical representations of data required for Item 17d were presented in the appendix of the Stage 2 report. For ease of interpretation and discussion, data were partitioned by sub-catchment. Data for each sub-catchment are represented as tables of summary statistics, time series graphs (Excel), boxplots (PASW Statistics 17.0) and Piper diagrams (GW_Chart, USGS). Due to overlapping data the time series graphs were further partitioned within each sub-catchment into upstream, mid and downstream sections to allow for better visualization. The time series graphs were arranged to allow for visual representation of trends of groundwater quality alongside standing water levels and rainfall patterns, so that antecedent landscape influences upon groundwater quality were evident. These representations provide a visual overview of the variation of solutes in groundwater at the Mine.

Time series graphs of COPC from the groundwater monitoring records at Mount Rawdon were plotted. The trends of electrical conductivity and sulfate/chloride concentration ratios were used as lead indicators of mine influence, with electrical conductivity representing the combined influence of cations in groundwater and an increasing trend of sulfate concentration possibly representing mine-influenced water resulting from dissolution of sulfides in tailings or waste rock. Normalising solutes with chloride adjusts for the evaporative effects in catena soils identified by the literature review to be a landscape process that concentrates dissolved ions in groundwater. The trend of total cyanide was used to represent the possible presence of tailings seepage in groundwater. The trend of nitrate-N was used to represent the possible presence of blast residues in groundwater (or natural causes). An increasing trend of these three tracers in combination, particularly if the trends were in close proximity to a groundwater flow pathway leading from a mining area, was interpreted as indicating a possibly mine influence requiring confirmation by geochemical transport modelling (refer section 2.2).

2.4.2 Identification of Groundwater Limits

Groundwater limits suitable for verifying the current and predicted extents of COPC's released to groundwater by mining activity were developed according to DSITI (2017), by using monitoring data to assess groundwater quality for potential environmental impacts. Two other references were used to support the development of groundwater limits:

- *The Queensland Water Quality Guidelines 2009 (DEHP, 2009). Refer Section 4.3.5 Deriving sub-regional water quality guidelines (SMD waters), pp. 92 – 96; and*
- *Statistical guidance for determining background groundwater quality and degradation (Idaho DEQ, 2014).*

The *Statistical guidance for determining background groundwater quality and degradation* developed by the Idaho DEQ states three assumptions that underpin the statistical theory and practice supporting background determination:

- **Random, unbiased samples:** The collection of measurements represents a random sample of the underlying population that is free of bias imposed by the measurement process (i.e. the individual who is conducting the sampling, or the analytical method used to make the measurements).
- **Representativeness:** The statistics estimated for a population depend on the measurements used to make the estimate but bracket the true population statistics (i.e. the sample is representative), allowing the population to be inferred from any sample.
- **Independence:** Most importantly, the data are independent. That is, each measurement is randomly representative of the target population and its value is not influenced by any other measurement (i.e. each measurement is independent of every other). Dependent variables exhibit less variability (i.e. samples taken at five-minute intervals will be near identical), which leads to an underestimation of the population variance that in turn affects the prediction limit and tolerance limit. Every measurement in the physical world is somewhat autocorrelated with the previous measurement, and from a practical standpoint the Idaho DEQ recommend that samples at six monthly intervals can be assumed to be independent except for slowly flowing aquifers.

Over the last twenty years there has been substantial groundwater data collected at the Mine using a consistent sampling procedure (similar pumps, same purging times, same field protocols and the same field technician for the last 10 years). From a practical standpoint, twelve data sampled over a minimum of three years and four seasons were assumed to meet the minimum sampling requirements from a monitoring bore (Idaho DEQ, 2014 page 44). This philosophy achieves the Australian standard (DSITI, 2017 page 11) that indicates a minimum of eight samples collected over one year for surface waters, and two years for groundwater, as the minimum standard.

A preliminary guide to interpreting groundwater with high concentrations of Chemicals of Potential Concern in the Mount Rawdon mining lease was developed (refer Appendix 12 of the Stage 2 report), and used to identify fifteen groundwater bores with sufficient data that are free of mine influence, from which groundwater limits (background) were identified using guideline procedures.

The Queensland Water Quality Guidelines (2009) have an interim procedure for developing sub-regional guidelines where the 20th and 80th percentiles from the reference sites are naturally heterogenous. This approach involves obtaining data from as many reference sites as possible, then calculating confidence intervals around the average 80th and 20th percentiles. That is two standard errors above the 80th percentile (i.e. +2xSE) and two standard errors below the 20th percentile (i.e. -2xSE) around the average 80th and 20th percentiles respectively. This government-standard procedure was adopted for the Stage 2 EE.

The procedure recommended by the Idaho DEQ (2014) was applied as a supplementary approach for the Stage 2 EE, primarily because the procedure had already been committed to by the Operational Groundwater Management Plan (Evolution, 2018). This procedure tests data for normality (p-p plots as well as Kolmogrov-Smirnov tests) and independence (Ljung Box Test). Some data were transformed to make the sampled population more normally distributed (log 100 [Ca], log [K], square root [Mg]), non-independent data were removed and trigger values were identified that accommodated for non-parametric population data distributions (if normality was not achieved or if a significant portion of data were non-detects). The trigger values were then back transformed to concentration in mg/L.

Both the DSITI (2017) and Idaho DEQ (2014) procedures were used to identify 80th percentile trigger values, and some COPC concentrations (aluminum, arsenic, boron, total cyanide, lead and selenium) were below the default ANZECC toxicant guideline values. For toxicants, DSITI (2017) indicate that when the local groundwater quality is less than the ANZECC toxicant guideline, the ANZECC toxicant guideline value is a suitable groundwater limit for initiating an investigation into the possibility of environmental harm. On this basis, the ANZECC toxicant guideline is proposed as the groundwater limit for aluminum, arsenic, boron, total cyanide, lead and selenium.

In summary, groundwater data collected at the Mine were used to identify the sources of all mining affected groundwater, and the likely horizontal and vertical extents of mining affected groundwater observed at the Mine.

3 The Groundwater Model

3.1 Overview

A salinity hazard map of the Central Burnett shows the possibility of low-moderate to moderate salinity hazard at the Mine, particularly on lower slopes (Figure 2).

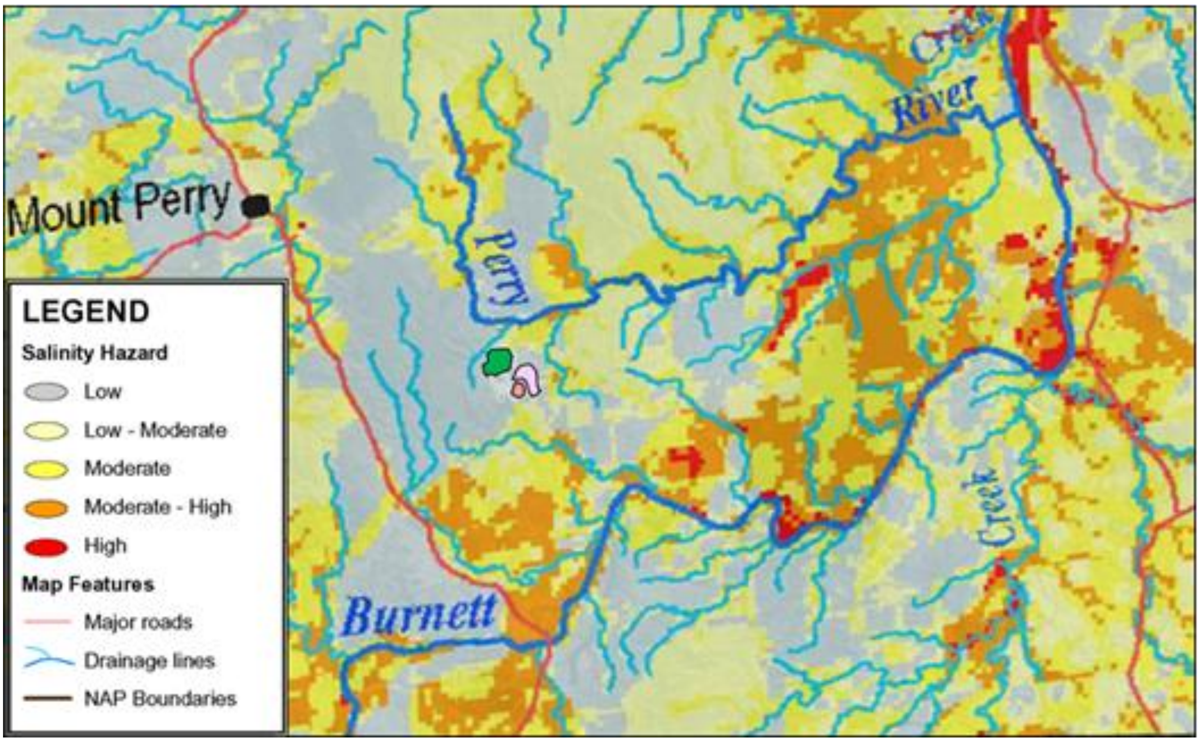

Figure 2: Salinity Hazard (Source: Queensland Government, 2003)[2]

The conceptual model for regional salinity developed by Douglas and Cox (1994) is included (Figure 3). Backflow of groundwater from the river towards the flanking landscape, through alluvium of creek-lines was identified by Douglas and Cox (1994). An overview of the main groundwater flow-paths in the Mine that illustrates the sources of solutes, the groundwater pathways that transfer solutes through the landscape, and barriers / conduits to groundwater flow is provided (Figure 3).

[2] The TSF (green), Northern Waste Rock Dump (pink) and the pit (orange) shown in the regional context of salinity hazard

The overview and conceptual model for regional salinity show that the main aquifers (fractured rock, weathered rock, colluvium and alluvium) facilitate groundwater movement, and flow barriers (dykes, lithological contacts) direct groundwater to surface exfiltration. These images are not to scale.

A geological map (Figure 4) and geological cross sections (Figure 5) show the main rock types and structures in the Mine. The map shows locations of groundwater groups identified by multivariate statistical analysis of water chemistry. Group 1 represents predominantly tailings seepage affected groundwater, Group 2 represents predominantly waste - rock affected seepage and group 4 represents groundwater (some of which has been mining - affected) hosted in Curtis Island Group meta-sediments with dacite or granite intrusive rock (refer section 3.3.1 of the Stage 1 report).

The locations of groundwater monitoring bores among groundwater flow pathways, and the potential for groundwater exfiltration to the surface, are reported in Section 3.3.

3.2 Conceptual Hydrogeological Model

Previously developed and well-established conceptual hydrogeological models of dryland salinity, which describe the generation and movement of solutes in catena soils of the Central Burnett, apply to the Mine (Douglas and Cox 1994, Queensland Government, 2003). These conceptual models describe the natural hydrological and hydrogeological process, as they occur unaffected by mining activities. Following these conceptual models, granite weathering in sloping terrain generates solutes that concentrate on lower slopes, and there is a moderate hazard of salt expression on the lower slopes of Mingham Creek south of the TSF and the WWRD, as well as moderate hazard of salt expression into Twelve Mile Creek east of the TSF and the NWRD, and into Rawdon Creek north east of the TSF (Figure 2).

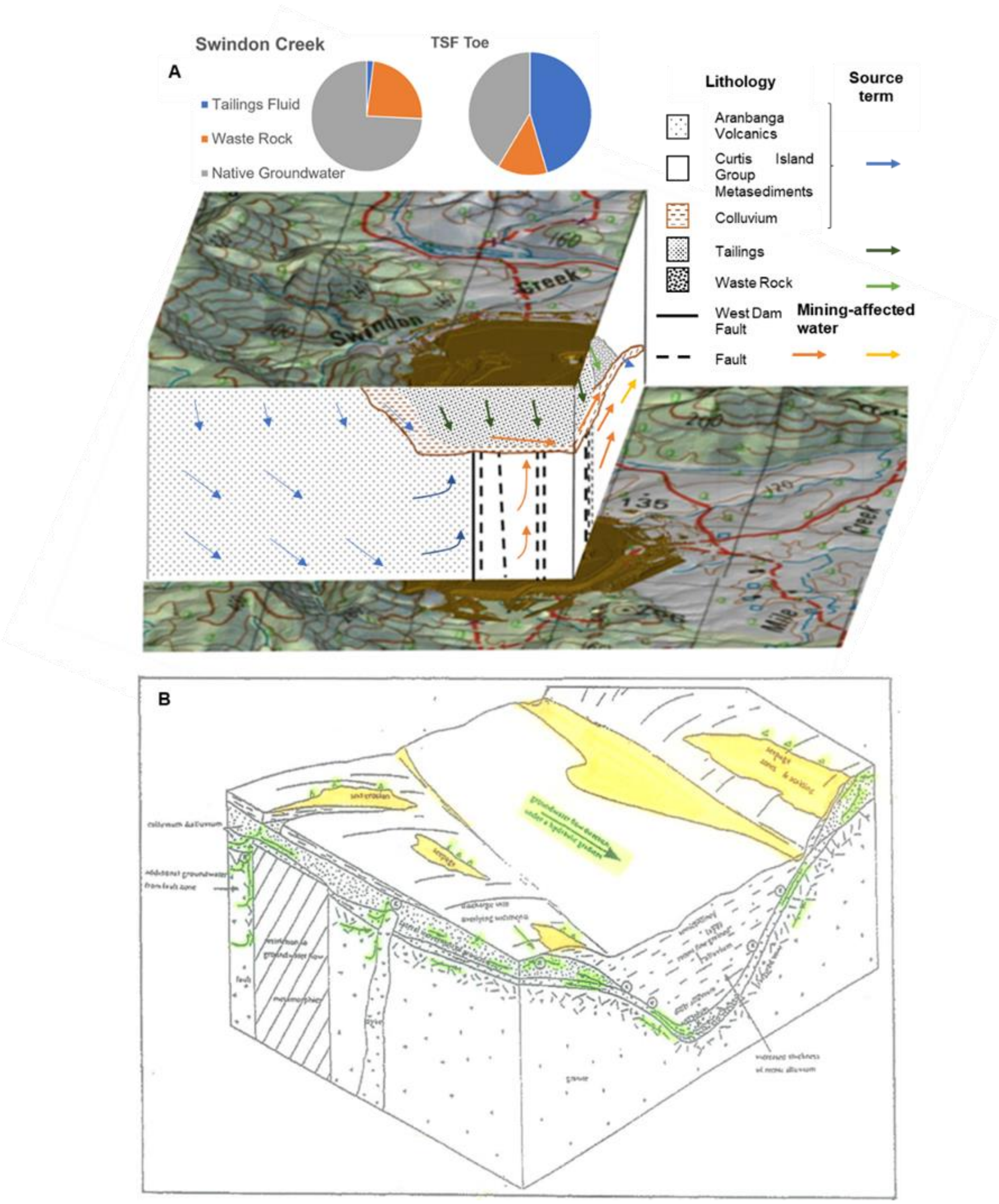

Figure 3: Overview of the main groundwater flow-paths in the Mount Rawdon mine lease [3]

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[3] (A) Schematic showing the main lithologies in the Mine, groundwater flow-paths, and barriers / conduits to groundwater flow. Source terms represented are tailings fluid and waste rock seepage (both mining related) and native groundwater. These source terms form mixtures of mining-influenced groundwater that migrate slowly downslope towards ephemeral creek lines. The Pie charts show mixtures of mining affected water sampled in 2006 from MRMB9 (close to the toe of the TSF) and from MRMB12 (close to Swindon Creek). (B) Conceptual model for regional salinity developed by Douglas and Cox (1994).

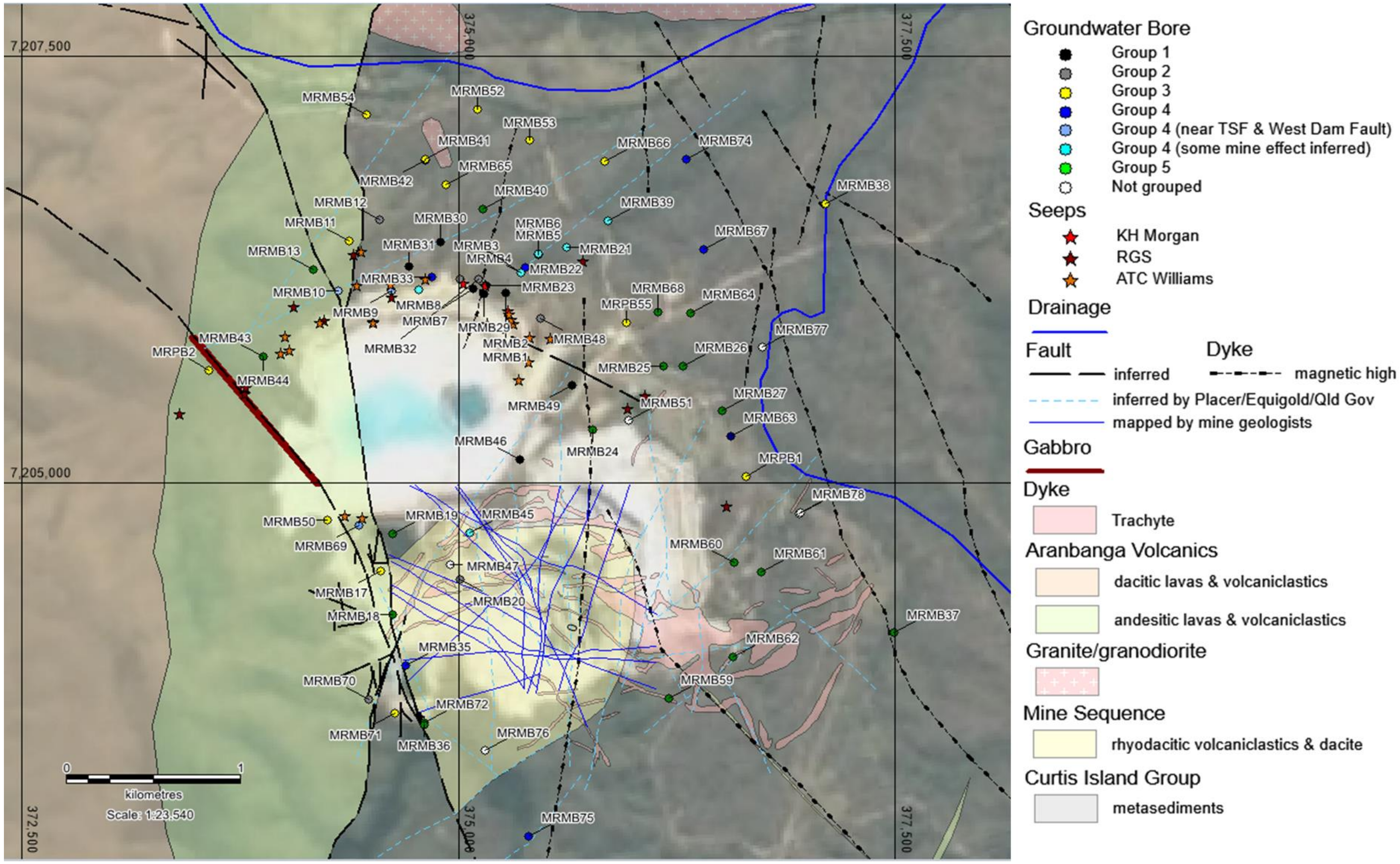

Figure 4: Spatial locations of groundwater classification groups in the Mount Rawdon Mine Lease [4]

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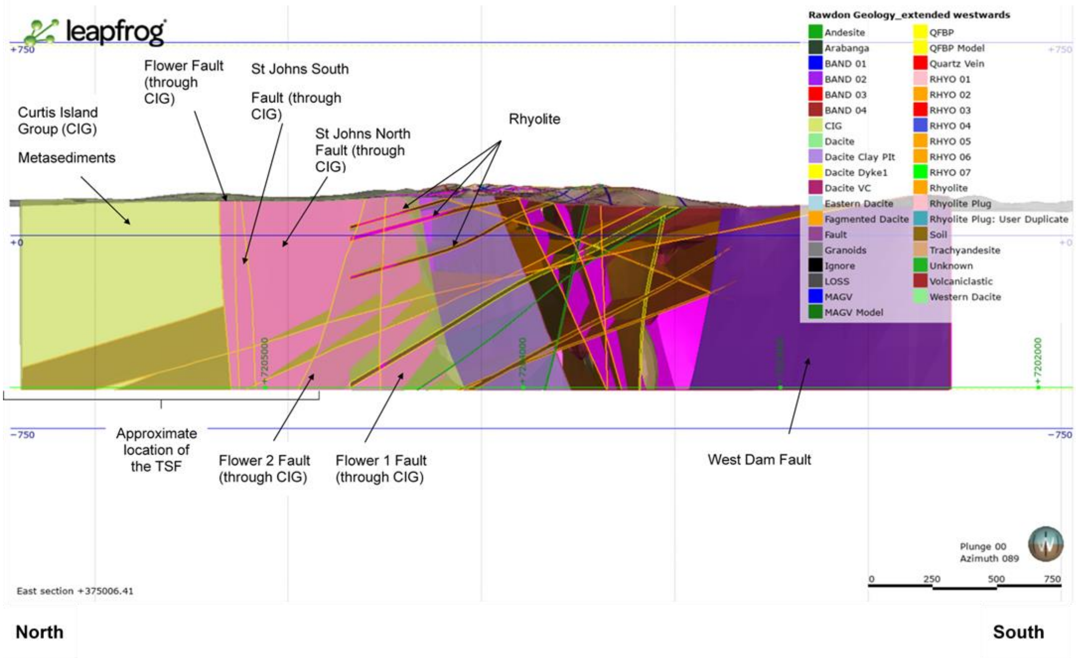
[4] Groundwater in Group 1 has been influenced by tailings seepage, in Group 2 by waste rock seepage (except for MRMB70 for reasons discussed in the Stage 1 report), and in Groups 3, 4 & 5 predominantly by geology and landscape. Mine influenced groundwater has affected Group 4 groundwater near the intersection of the TSF & West Dam fault, and along Rawdon Creek. Refer to section 3.3.1 of the Stage 1 report for information about the multivariate classification of groundwater types at the Mine.



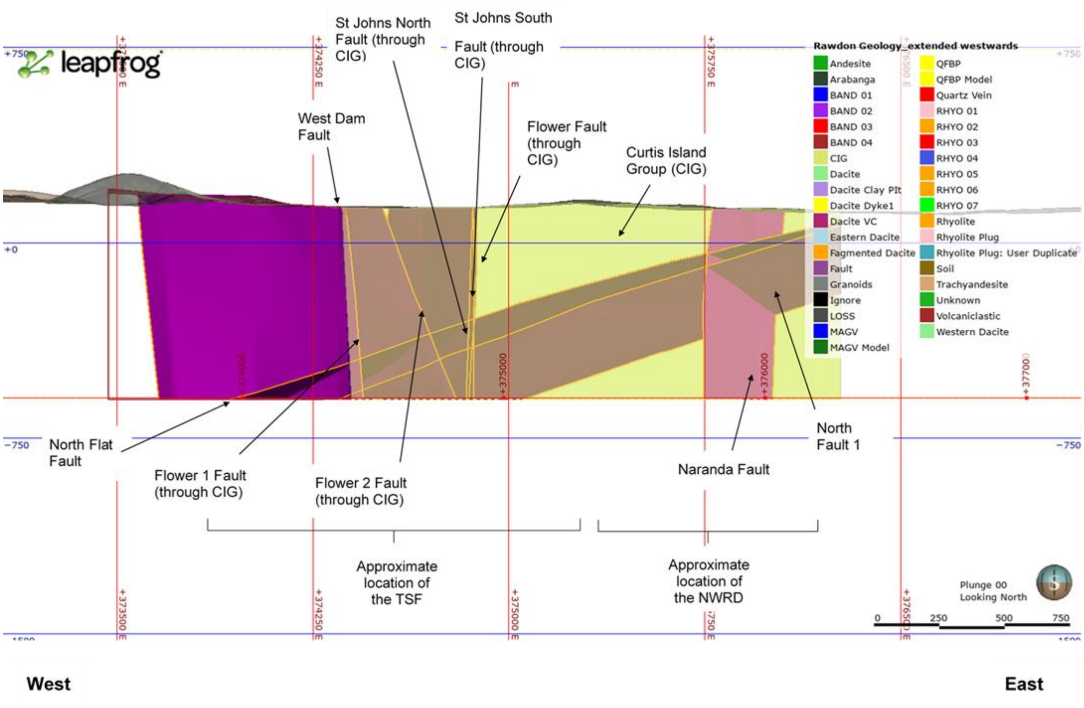


Figure 5: Geological cross sections [5]

[5] Geological cross-sections north to south along easting 375000 (top image), and west to east along northing 7205200 (bottom image). The sections show the major faults, colour enhanced to represent their apparent strike and dip, which are at oblique angles to the cross-sections. The Curtis Island Group lithology, TSF and NWRD are shown. The cross-sections are located approximately along the edges of the raised block shown in Figure 3.

After mining commenced, the TSF and later the NWRD became the highest hydraulic head features in the area leading to groundwater mounding below these post mining features. The pit became the lowest head feature in the area. The presence of flow barriers (faults between the pit, TSF and surrounding strata) has led to little inflow into the pit. Groundwater flow potentials across the Mine have been altered by mining-related water management activities.

The conceptual model described in the Stage 1 EE report (the Stage 1 report) proposed that the principal components of the median ionic compositions of water in the Mine were consistent with:

1. Weathering of feldspar and carbonate in the natural landscape, represented by high concentrations of sodium and bicarbonate coupled with low concentrations of nitrate,
2. Solutes generated by the mining and processing of mineralized ore, represented by high concentrations of sulfate, nitrate, total cyanide and arsenic, and
3. Solutes generated from unmined mineralized sources, represented by high concentrations of arsenic and low concentrations of total cyanide.

The Stage 1 report identified the interactions of mining-affected waters with groundwater along flow-paths towards Swindon Creek, Rawdon Creek, Twelve Mile Creek and Mingham Creek. More detailed interpretations at the sub-catchment scale were made, which allowed identification of influences on the ionic composition of groundwater caused by soil and geology variations in the landscape, as well as periods when groundwater was not mine affected prior to breakthrough of mine related seepage. This interpretation also identified the ambiguity of some tracers of mine influence, where landscape processes that influence the concentration of NO_3 in groundwater (nitrogen fixing plants, nitrification in organic rich soil), as well as As and SO_4 in groundwater (unmined sulfide in a mineralized area), were recognized as potential confounds to interpretations of mining influence.

Further details of the conceptual model, and hydrogeological features of the Mine, are provided in the Stage 1 EE (refer sections 3 and 4 of the Stage 1 report).

3.3 The Hydrogeological Model

The MODFLOW-SURFACT model in the Stage 1 report described the contemporary groundwater flow across the mine lease. The model identified that parts of the mine lease are likely compartmentalized. These compartments include the landscape spur between Rawdon Creek and Twelve Mile Creek, the lower portion of Twelve Mile Creek towards the impounded zone of the Perry River, and the headwaters of Twelve Mile Creek.

The hydrogeological model supported the conceptual model, by describing groundwater flow that follows the topographic gradient and flows below drainage lines within 3 meters of creek beds. Groundwater flows below the TSF towards the Perry River. The TSF and the NWRD have generated topographic highs, where groundwater mounding under these structures provide hydraulic head for groundwater flow towards the Perry River via Rawdon Creek and via Twelve Mile Creek. In the Southern Corridor, groundwater flows westward into Mingham Creek, with the pit providing a localised sink for groundwater flow.

3.3.1 Groundwater Flow

Figure 6 summarises the location of groundwater monitoring bores in the context of the potentiometric surface and inferred fault structures that might provide barriers to groundwater flow. This figure shows that groundwater mounding below the NWRD provides hydraulic head potential for groundwater flow towards Twelve Mile Creek, and shows the influence of the pit as a sink for groundwater flow towards Mingham Creek. Groundwater flow directions across the mining lease are also shown in Figure 6. Figure 7 is a variation of Figure 6, with colour inversion to represent the depth of groundwater below the landscape. Hence while groundwater level contours represent the same data shown in Figure 6, red signifies that the soil and weathered rock aquifer are dry while blue signifies relatively wet soil and weathered rock conditions.

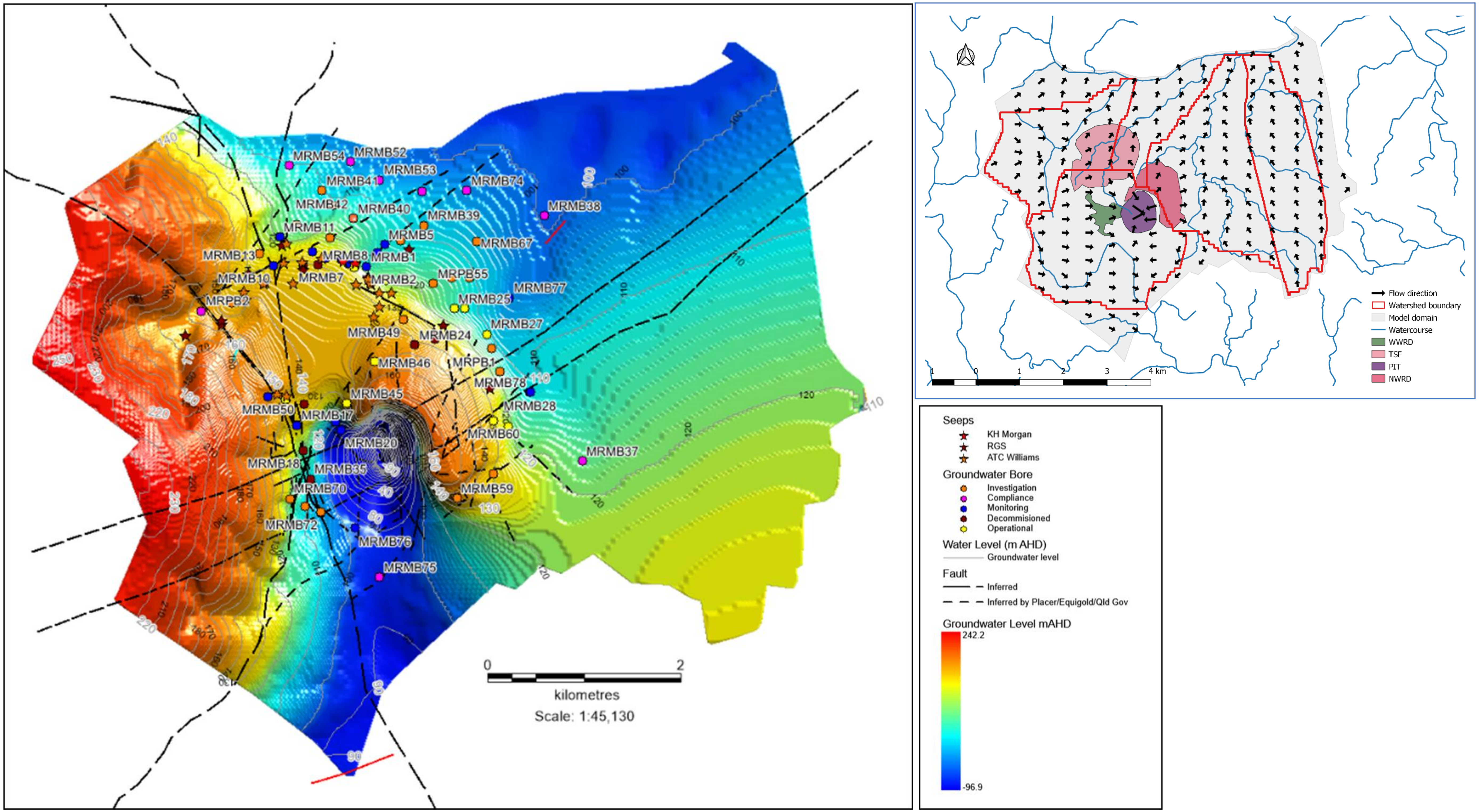


Figure 6: Groundwater model of the potentiometric surface across the Mount Rawdon mine lease, with insert showing the groundwater flow directions (Source: Northern Resources Consultants, 2019)

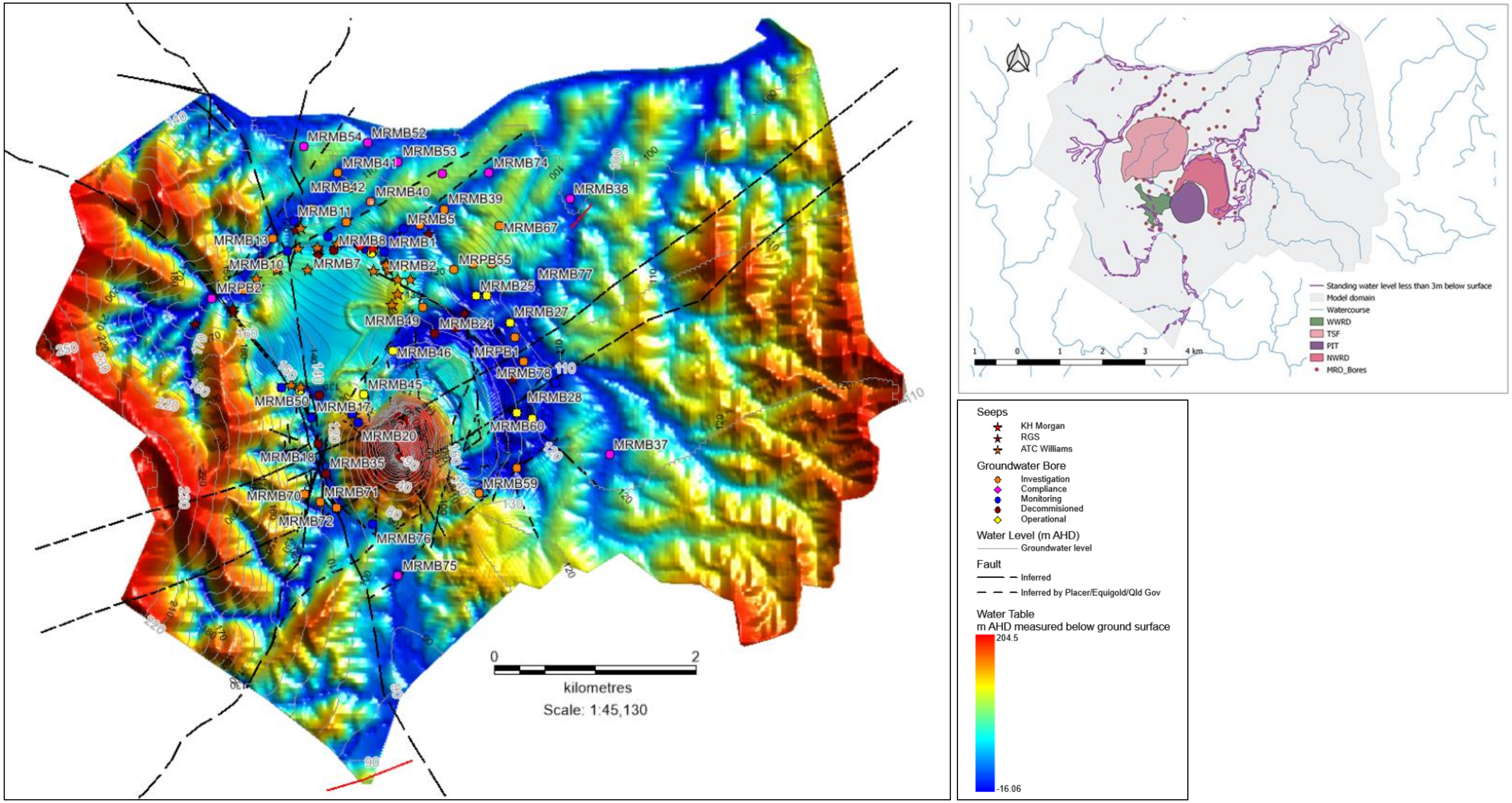


Figure 7: Groundwater model of the depth to water across the Mount Rawdon mine lease, with insert showing groundwater less than 3 meters below the ground surface (Source: Northern Resources Consultants, 2019).

3.3.2 Groundwater Exfiltration

Surface expressions of groundwater into creek lines were identified in the upper catchment of Swindon Creek, and in the headwaters of Mingham Creek, most of which occurs upstream of mining activity (Figure 8).

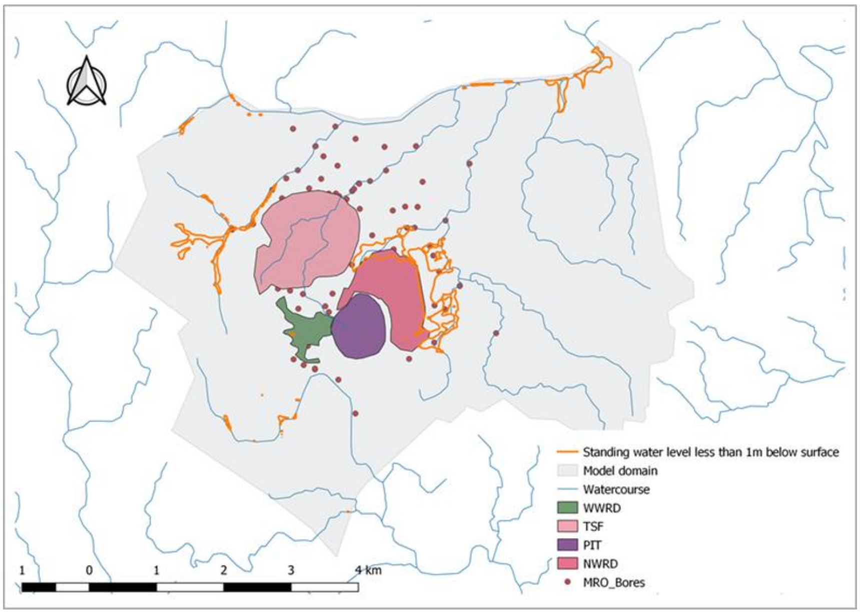


Figure 8: Groundwater egress into surface water drainages (Source: Northern Resources Consultants, 2019)[6]

The groundwater model identified rates of groundwater exfiltration into drainages for nine water budget zones (Figure 9). The most significant groundwater input is to zone 5 in the middle of Twelve Mile Creek, downslope of the NWRD. There is potential for hyporheic groundwater flow less than 3 meters below drainage lines, including in areas downstream of mining influence in Swindon Creek, Rawdon Creek, Twelve Mile Creek, and in the southern drainage line leading into Mingham Creek (Figure 7).

In summary the post mining topography of the mine lease has generated groundwater mounding below the TSF and NWRD, which has enhanced groundwater expression into Twelve Mile Creek. The greatest potential for surface expression of groundwater is to zone 5 of Twelve Mile Creek, which discharges 10 m^3/day under steady state groundwater flow conditions.

[6] The groundwater model generally overestimated the groundwater level in fractured rock aquifer by about 1 meter. Therefore, model prediction within 1 meter of the landscape surface, as shown in Figure 8 is most likely to represent surface expression of groundwater.

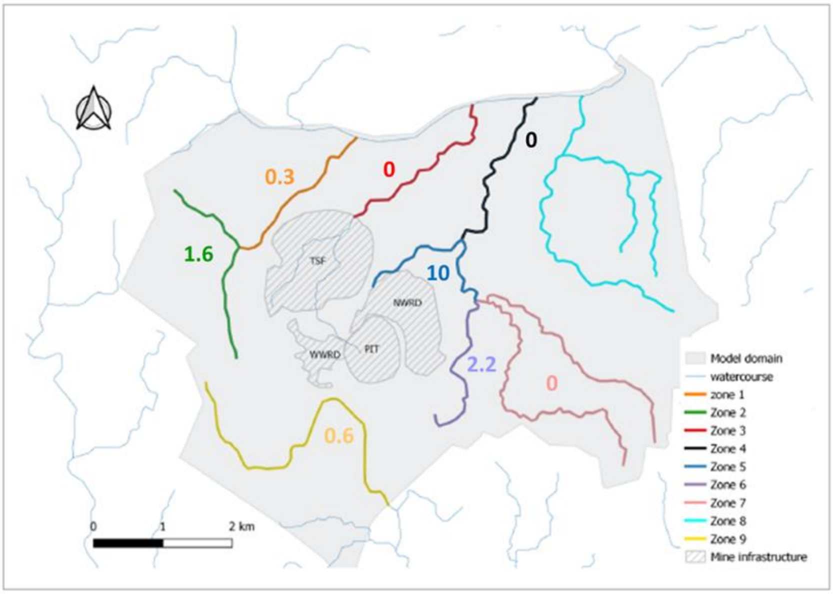


Figure 9: Groundwater exfiltration (m^3/day) into water budget zones (Source: Northern Resources Consultants, 2019)

3.3.3 Groundwater Flow Paths Selected for Geochemical Transport Modelling

Five groundwater flow pathways across the mine lease, between the mining infrastructure and drainage lines, were selected for geochemical transport modelling (Figure 10) as described in section 2.2. The groundwater flow and velocities along these flow paths were calculated from the hydraulic head potentials identified from the MODFLOW-SURFACT model and the aquifer properties identified in the Stage 1 report. Refer section 3.1 and section 4.2 of the Stage 1 report for more detail.

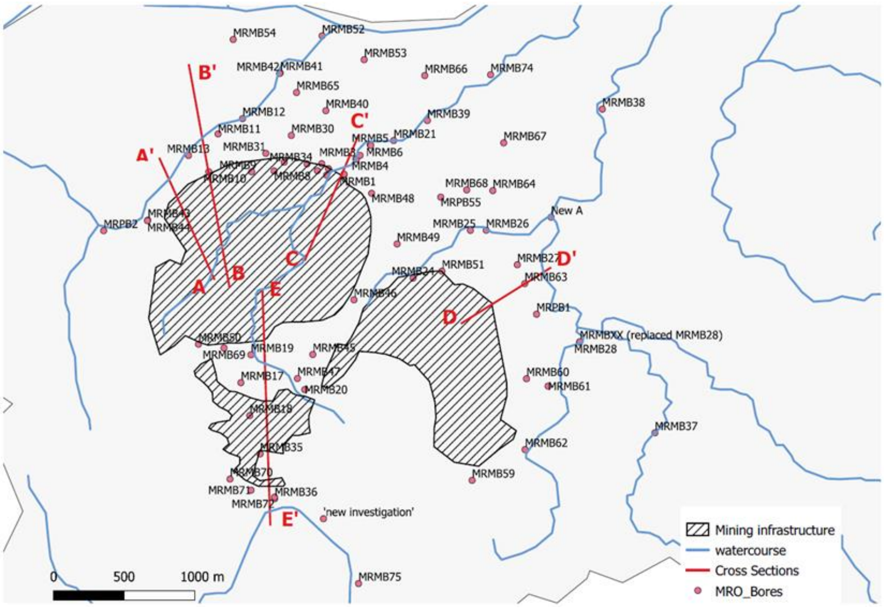


Figure 10: Reference map of groundwater profile sections in the Mount Rawdon mine lease (Source: Northern Resources Consultants, 2019)

4 The Geochemical Transport Model

4.1 Sources of Uncertainty, Calibration and and Sensitivity Analysis

Agreement between modelled PHREEQC outputs and measured groundwater data was done manually by matching modelled sulfate concentrations with observed sulfate concentrations in breakthrough curves in monitoring wells. Sulfate was chosen because it is a relatively conservative ion associated with mining, noting that some non-conservative behaviour in the form of gypsum precipitation and redissolution associated with rainfall patterns does occur in the Mine. A sequence of processes was simulated, with the best match at each step being built on from the sequential step. The stepwise processes were (1) Darcian flow velocity controlled by adjusting the hydraulic conductivity of the aquifer across the measured range in the mining lease, followed by (2) exchange of cations between aquifer minerals and groundwater controlled by adjusting the CEC for sodium, potassium, magnesium and calcium and adjusting the specific surface areas attributed to hydro-ferric oxide (HFO), across the ranges of measured CEC and measured iron oxides from representative aquifer samples, and (3) evaporation. These three processes were evaluated sequentially and in combination, using a progression of 4 or 5 steps representing low, middle, high and extreme values across the measured data ranges. The models that agreed best with data observed in monitoring bores along the respective flow-paths, were used to represent the unmitigated (base-case) models.

The geochemical transport model simulations performed for the selected flow-paths represent substantial simplifications of the natural system. Some of the key assumptions, uncertainties and exclusions were:

- The conceptual hydrogeological model underpinning the geochemical transport model was based on an earlier developed model describing landscape processes associated with the Mount Rawdon Mine. It is assumed that this model can also apply to the post mining period if suitable modifications were made to represent the final landform after mine closure. The model addresses processes controlling dryland salinity on sloping landscapes, and groundwater mounding below the TSF / waste rock dumps. These assumptions are discussed in more detail in Section 4 of the Stage 1 report.

- All considered chemical processes, i.e. cation exchange, surface complexation and mineral dissolution / precipitation were assumed as equilibrium reactions.

    - The PHREEQC model has assumed that primary and secondary minerals identified in the tailings dam and in the landscape represent weathering pathways, and that these weathering pathways have proceeded to equilibrium in the TSF. Note that equilibrium might not occur during weathering of some primary minerals in the TSF (particularly minerals that that are slow to react such as albite), which is an uncertainty in the model.

    - The cation exchange capacity of the aquifer matrix represents the equilibrium state of cations adsorbed to aquifer clays at the time of sampling. As such the PHREEQC model has assumed that cation exchange of solutes with groundwater will be constrained by the charge balance and exchangeable quantity of cations at the time of sampling. This assumption is likely valid given the availability of bicarbonate present in the neutral mine drainage at Mt Rawdon as well as in background groundwater, which maintains the pH of the aquifer system and the cation exchange capacity of the aquifer matrix.

    - Chloride has not been well represented by the model. While chloride is usually a conservative (unreactive) element in groundwater, the likelihood that halite precipitates and redissolves seasonally in the variably saturated aquifer was not represented in the PHREEQC model (the PHREEQC model used here assumes steady-state, one-directional groundwater flow).

    - Organic Matter (dissolved organic carbon) mineralisation was not considered, which could have had some influence on the redox conditions. Neither the NETPATH or PHREEQC models have represented biogeochemical processes that degrade cyanide and nitrogen in the TSF and the landscape, nor have they represented processes that add nitrogen to groundwater in the landscape (nitrogen fixation by wattles, hot bushfires). Some chemical processes relevant to cyanide and nitrogen are uncertainties that have been excluded from the model.

    - There is a potentially a wide range of solute concentrations that can occur in leachates from waste rock and from tailings, as discussed in Section 3.4.1 of the Stage 2 report for manganese and iron in waste rock leachate, and for copper in tailings fluid. By adjusting the model inputs for iron, manganese and copper from the median value to lower values within the known range of concentrations, it is possible to reconcile model outputs with observed concentrations in the seepage expression. However, it needs to be noted that even models that perfectly match observations will contain non-unique outcomes and uncertainties.

- Uniform, steady-state groundwater (advective) flow through homogenous aquifers along the A-A', B-B' … E-E' transects. This assumption might hold for saturated groundwater flow through soil and strongly weathered saprolite. However, it might not hold for aquifer recharge by rainfall in the variably-saturated zone that seasonally precipitates then re-dissolves solutes during downslope movement towards valleys. This assumption might explain the poor reconciliation of model results for chloride, and for sulfate/chloride ratios, with observed data from the monitoring bores.
- Fractured rock features compartments of stagnant groundwater in isolated or semi isolated pockets, which form against fault barriers and planes of weakness along lithological contacts, foliations and bedding planes. Groundwater in these compartments may not mix along the flow-paths as assumed by the employed advection - dispersion model, and bimodal flow through fractured rock has not been included in the model.
- Faults were assumed by the earlier MODFLOW-SURFACT model to be impermeable boundaries to flow. As such the flow-paths and hydraulic-head potentials developed as output of the MODFLOW-SURFACT model groundwater flow model, and subsequently used as input for the 1-dimensional PHREEQC model, tacitly carry the assumption that faults represent flow barriers. Alternatively, the other extreme case some faults may act as preferred pathways for solute flow. However, the process of calibration of the MODFLOW-SURFACT groundwater model identified, and allowed for, some permeability of groundwater flow through faults.
- The simplified one-dimensional solute transport in a three-dimensional landscape has not accommodated for converging flow-paths in the spur-valley landscape of the mine lease (e.g. the amphitheatre-shaped valley that converges slightly downstream of monitoring bores MRMB43 and MRMB44). The assumption of the model may be a limitation because it only represents one-directional groundwater flow from the TSF to the creek and does not represent aquifer recharge from the creek (i.e. backflow from the creek into the aquifer).
- Changes in redox potential will have a strong influence of the precipitation of Fe and Mn in particular. These hydrous ferric oxyhydroxides have the potential to act as major sorbents for a range of metals and metalloids and hence regulate their concentration.

In summary, while solute transport of major cations and some trace elements are well represented by the model, the behaviors of chloride, nitrogen, and some trace elements are not well represented by the geochemical transport model. The discrepancies can mostly be attributed to the above-listed simplifying assumptions, uncertainties and some exclusions relevant to advective flow, weathering pathways, and solute interactions within the aquifer.

4.2 Results

Refer to section 5 of the Stage 2 report, which customised interpretations to each sub-catchment in order to explicitly address these parts of the Mine. Some general statements are provided below.

4.2.1 Speciation Modelling

Speciation modelling showed that the activities of Al^{3+}, Cr^{3+}, Cd^{2+}, Cu^{2+}, Fe^{2+}, Mn^{2+}, Pb^{2+} and Zn^{2+} in groundwater were lowered substantially relative to total concentrations. Several dissolved metals (Cd^{2+}, Fe^{2+}, Mn^{2+} and Zn^{2+}) exist dominantly as free ions, with bicarbonate and carbonate species of these metals being present in lesser amounts. Carbonate complexes are the dominant dissolved forms of Cu^{2+} and Pb^{2+}. Hydroxyl complexes form with Al^{3+} and Cr^{3+}. Boron and arsenic occur as boric acid (H_3BO_3) and arsenous acid (H_3AsO_3). As such, carbonate, hydroxyl and oxide complexes are important components of the groundwater matrix that moderate the release and transport of solutes in groundwater.

Aluminium (Al^{3+}) and Mn^{2+} occur at concentrations that allow the formation of mineral precipitates (e.g. gibbsite, pyrolusite and potentially other minerals), which immobilise Al^{3+} and Mn^{2+} and limit their transport in groundwater. Coprecipitation of clay and carbonate (e.g. kaolinite, calcite and other minerals) generate fracture minerals with charged surfaces that adsorb Pb^{2+} and Cu^{2+}, thereby assisting the aquifers capacity to attenuate these COPCs.

Beyond the prediction of metal speciation based on major ions and trace metals using Visual MINTEQ, we have not measured for dissolved humic or fulvic substances that might complex metals such as cadmium, copper and zinc in receiving surface water or groundwater environments. Nor have we progressed with more detailed interpretation of the possible role that metal speciation, might have on toxicity (i.e. the valence state of the metal and whether the metal is in an inorganic form or complexed with an organic ligand). The reasons for not progressing with a toxicity investigation was because there is no evidence for contamination of waterways based on seven years of AUSRIVAS monitoring of aquatic macroinvertebrates in local streams, as well as a recent investigation of stygofauna that reported low occurrence and limited range of taxa in groundwaters of the Mine.

In summary dissolved complexes lower the activities of Al^{3+}, Cr^{3+}, Cd^{2+}, Cu^{2+}, Fe^{2+}, Mn^{2+}, Pb^{2+} and Zn^{2+} in groundwater solution, which moderates their behaviour in groundwater. A proportion of these COPC's are expected to be transiently to permanently immobilised by coprecipitation and/or adsorption to fracture minerals such as gibbsite, pyrolusite, clay and carbonate depending on the prevailing physico-chemical conditions.

4.2.2 Solute Movement

Inverse models (NETPATH) were used to evaluate the processes of rock weathering, and formation of secondary minerals in the weathering profile, which were identified by the conceptual model (discussed in section 2.2 and tabled in Appendix 1 of the Stage 2 report). Models converged, which indicate that there are stoichiometrically plausible descriptions of the release and transport of solutes in groundwater moving from the TSF or NWRD to ephemeral creeks.

In general, the inverse models were consistent with the presence of a small amount of mining - affected water entering the riparian zones of Swindon Creek and Rawdon Creek via the Northern Seepage Zone. The mining - affected water had reacted with the aquifer during groundwater movement from the mining areas to the monitoring bores.

The PHREEQC solute transport model results (refer Appendices 7 – 11 of the Stage 2 report), which included mitigation scenarios, were as follows:

- The graphs show how modelled data for the breakthrough curves responded to varying amounts of evaporation/evapotranspiration during downslope flow of groundwater, with modelled results being compared with measured data. Results indicated reasonable fit for some ions and poor fit for other ions (the outcome of the sensitivity tests and calibration are discussed below).
- Graphical PHREEQC output for unmitigated seepage, which indicates the projected long-term reporting of mining - affected water to receptors, predicted that sulfate, cyanide and arsenic (anions) are mobilized faster than copper and lead in seepage.
- The effects at slowing the breakthrough of mining - affected water to receptors were modelled for a combination of mitigations.

Sensitivity testing and calibration of the solute transport model indicated that the best model fits were for groundwater flow through saprolite with hydraulic conductivities that aligned with measured values, low CEC capacities, low availabilities of hydro-ferrous oxide surfaces, and some evaporation. The models provided the best possible fits for SO_4, Ca, Mg, Na and K (refer appendices 7 – 11 of the Stage 2 report). However, the fit for some other elements was poor (N, Cl), possibly because processes responsible for groundwater concentrations of these elements are not represented by the PHREEQC model (refer discussion in section 4.1).

Model results from flow path A-A' are shown for scenarios with between 1% and 5% evaporation and are graphed alongside observed data from a monitoring bore to demonstrate goodness of fit (Figure 11).

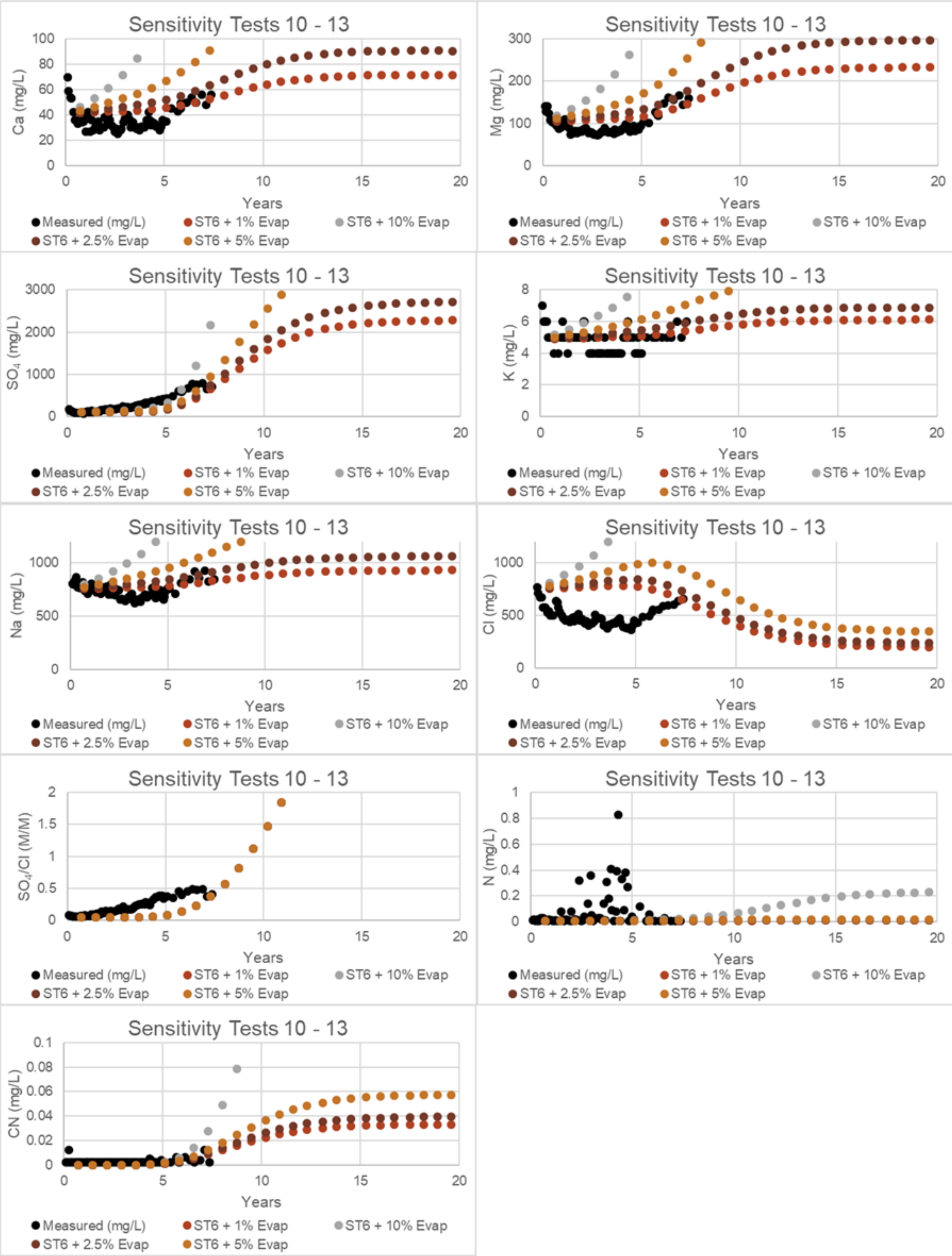


Figure 11: Model Outputs for Flow-path A-A'[7]

[7] Measured data from MRMB44

5 Results and Assessment of Impacts and Potential Impacts to Groundwater and Surface Waters

The EE developed and applied a decision tree to distinguish background groundwater quality from mining-affected water, and proposed groundwater trigger limits for reviewing and analyzing historical monitoring data. Exceedances of COPCs that were above the proposed limits were discussed in the Stage 2 report. There were instances where exceedances were attributed to non-mining phenomena such as lateral displacement of groundwater from the Perry River Weir, and unmined sulfide mineralization in the mining lease. There were other instances where mining related exceedances were identified, most of which were promptly brought under control during the history of Mine operations as discussed in the Stage 1 report (refer appendix 8.5 of the Stage 1 report). Further mitigation involving deep dewatering bores need to be implemented during the remaining operational life of the Mine, to stabilize and reverse some mining-related seepage that has continued to pass below the Seepage Interception Drain, and address the risk of seepage in the Twelve Mile Creek catchment. These management options are discussed in section 6.

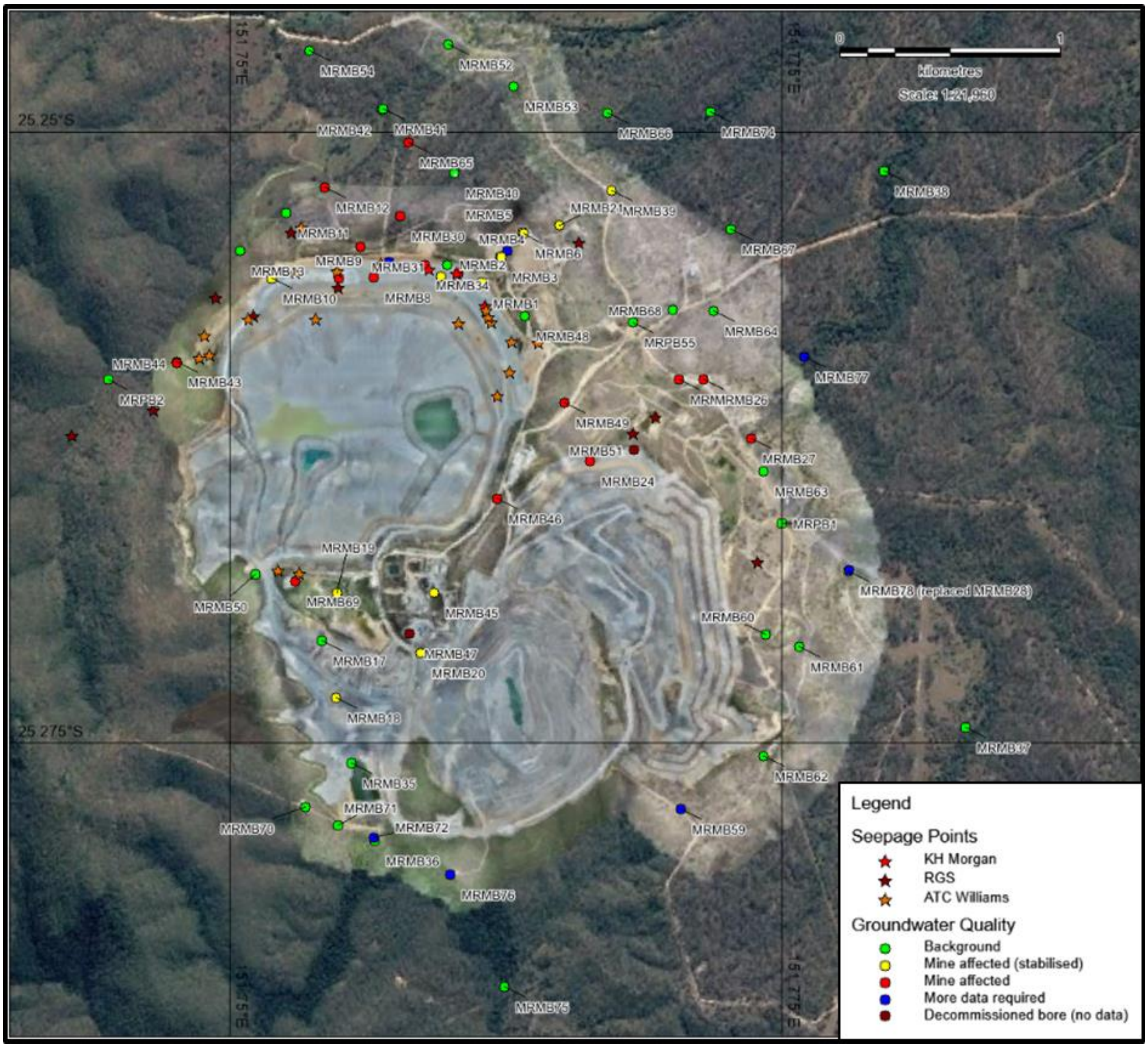


Figure 12: The lateral extent of mining - affected water across the mine lease

Graphical results that address Item 12 of the EE are provided in appendices 12 – 18 of the Stage 2 report, and are descibed in section 5 and section 6.6 of the Stage 2 report. A map summarising the current status of mining affected groundwater is provided in Figure 12, which can be compared with the water quality groupings developed for the conceptual model (Figure 4) to indicate where the mining infuences are greatest and therefore where management options need to be directed.

5.1 Swindon Creek

The PHREEQC solute transport model was used to estimate the effects at slowing breakthrough of mining - affected water to MRMB44 using four mitigation options:

- **Mitigation 1.** Lowering the effective head from the base-case (A-A' Saprolite + 2.5% Evaporation), by 0.75 m by extending the northern seepage drain to include seepage from the Seep 26. This would lower the rate of groundwater flow from the TSF from 7 m^3/d to 6 m^3/day.
- **Mitigation 2.** Lowering the effective head from the base-case (A-A' Saprolite + 2.5% Evaporation) by installing MRDB2 and MRDB3 to intercept seepage from the Seep 26 (this has already been done). This theoretically lowered the rate of groundwater flow from the TSF from 7 m^3/d to 6 m^3/day.
- **Mitigation 3.** Combined effects of mitigations 1 and 2. This would lower the rate of groundwater flow from the TSF from 7 m^3/d to 5.3 m^3/day.
- **Mitigation 4.** Mitigation 3 plus installing an extra dewatering well. This would lower the rate of groundwater flow from the TSF from 7 m^3/d to 4.9 m^3/day.

Mitigation 2 has been actioned, while mitigations 1 and 4 will be actioned in FY20-21

5.2 Northern Seepage Zone (including the Unnamed Creek)

The PHREEQC solute transport model was used to estimate the effects at slowing breakthrough of mining - affected water through the Northern Seepage Zone using 4 possible mitigations:

- **Mitigation 1:** The Seepage Interception Drain (current case) lowers the effective head potential of the seepage pathway by 0.75 m (from 11.3 m to 10.5 m). This changed groundwater discharge along the flow-path from 2.5 m^3/d to 2.3 m^3/d and the seepage velocity from 20 m/year to 18 m/year.
- **Mitigation 2:** Installing a dewatering bore capable of extracting 1.2 m^3/day. This is predicted to limit groundwater discharge from 2.5 m^3/d to 1.3 m^3/d and the seepage velocity to 13 m/year along the identified flow-path.
- **Mitigation 3:** Combining mitigation 1 and mitigation 2 is predicted to limit groundwater discharge to 1.1 m^3/d and the seepage velocity to 9 m/year along the flow-path.
- **Mitigation 4:** Mitigation 3 plus additional dewatering capacity of 0.8 m^3/day (totaling 2 m^3/day) is predicted to limit groundwater discharge to 0.3 m^3/d and the seepage velocity to 3 m/year along the identified flow-path.

Mitigation 1 has been actioned, while mitigation 4 will be actioned in FY20-21

5.3 Rawdon Creek

The PHREEQC solute transport model was used to estimate the effects at slowing breakthrough of mining - affected water into Rawdon Creek using 4 possible mitigations:

1. **Mitigation 1:** Existing mitigation includes the Seepage Interception Drain (the current case), which lowers the effective hydraulic head by 0.75m and slows groundwater discharge along the C-C' transect from 3.45 m^3/d to 3.23 m^3/d and the seepage velocity from 22 m/year to 20 m/year,
2. **Mitigation 2:** The addition of a dewatering bore dewatering at 0.8 m^3/d, which would limit groundwater discharge from 3.45 m^3/d to 2.65 m^3/d and the seepage velocity from 22 m/year to 17 m/year,
3. **Mitigation 3:** Combining mitigation 1 and mitigation 2, which would limit groundwater discharge from 3.45 m^3/d to 2.43 m^3/d and the seepage velocity from 22 m/year to 15 m/year, and
4. **Mitigation 4:** Combining mitigation 1 and a second dewatering bore dewatering at 1.2 m^3/d, which would limit groundwater discharge from 3.45 m^3/d to 2.03 m^3/d and the seepage velocity from 22 m/year to 13 m/year.

Mitigation 1 has been actioned, and mitigation 3 will be actioned in FY20-21. Operational monitoring suggests that the current strategy (Mitigation 1) is stabilising the trends of mining related solutes (total cyanide and the SO_4/Cl ratio) reporting to MRMB21 and MRMB39 in Rawdon Creek.

5.4 Landscape Spur and Twelve Mile Creek

Existing mitigations include MRMB27 and a sump downgradient of WD2, which intercept mining-affected water seeping from WD2. The combined effect of the sump and dewatering bore is shown in Figure 13 where the modelled basecase trend (with no mitigation) is graphed with observed data collected before and after mitigation. It is possible that the cones of depression developed by the dewatering infrastructure do not intercept all the seepage from WD2, therefore a second dewatering bore will be installed on the south eastern flank of WD2. The location of this additional bore has been guided by geophysical surveys commissioned by the EE, to ensure that the groundwater pathway is connected.

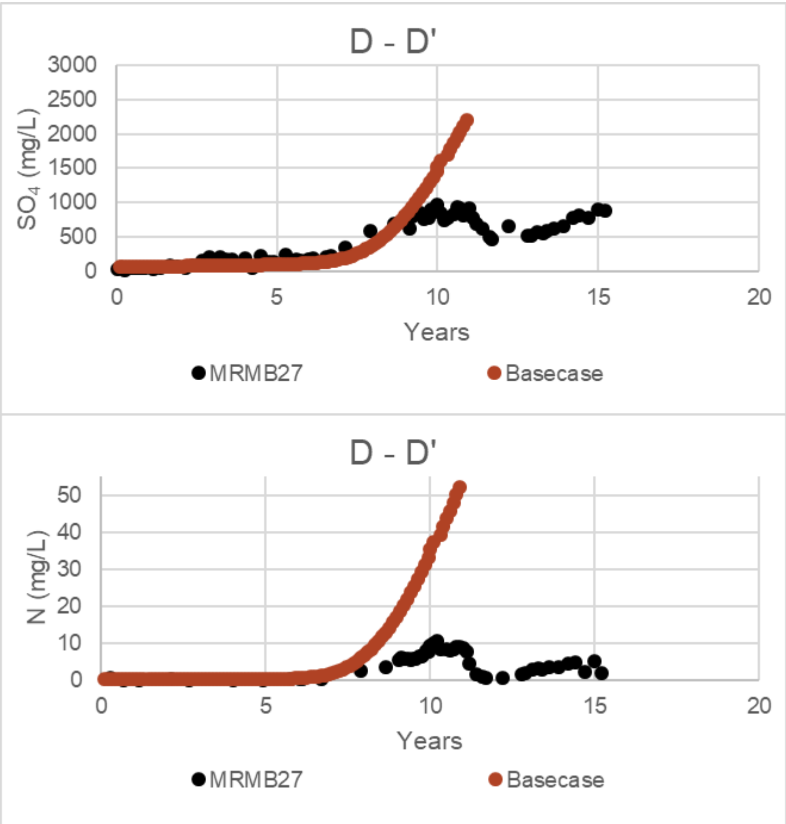


Figure 13: Model Outputs for Flow-path D-D'[8]

5.5 Mingham Creek

There is no evidence that TSF seepage has penetrated significantly downgradient of South Dam.

Further down the southern drainage the aquifer, standing water levels in groundwater monitoring bores have responded to the commissioning of West Dam. Some of the standing water level responses to increased hydraulic loading from West Dam have been slow, and slow hydraulic responses that have been observed in some dewatering bores also provide evidence of weak hydraulic connection in this aquifer. In summary, the groundwater responses observed in several

[8] Seepage mitigation has included equipping MRMB27 with a pump and commissioning a sump with enhanced drainage immediately downgradient of WD2. The observed data represent the effect of mitigation, which is shown alongside the modelled unmitigated trend (the basecase).

bores near West Dam are interpreted as screening weakly connected groundwater compartments in semi-pervious to impervious rock. As such seepage of mining affected groundwater into Mingham Creek was interpreted as being of low risk, and additional seepage mitigations are not thought to be necessary along the E-E' groundwater pathway.

5.5 The Surface Water Model

The surface water model highlighted the seasonal variation in COPC concentrations in surface water foreshadowed for wet and for dry climate regimes over the next decade, and the predicted role that seepage mitigation will have at stabilising and reversing mining influences on downstream receptors.

Results between 2020 and 2033 are shown graphically in the appendix for rainfall intensity (mm/hour) in sub-catchment (S10), as well as for water depth (m) and for sulfate, arsenic and cyanide (mg/L) in Swindon Creek immediately downstream of MRMB44 (C231_2) and MRMB12 (C166), as well as in Perry River immediately downstream of the Swindon Creek confluence (C14). The results show flow pulses along the creek lines that rapidly dilute and flush solutes accumulated in the effectively dry stream bed, with the flow pulses then ebbing and drying off unless replenished by a subsequent flow pulse. While modelled concentrations of COPC exceeded water quality guideline limits, the high concentrations represent periods when the creek lines are effectively empty of water.

To better visualise how COPC concentrations vary during flows along Swindon Creek and Perry River, a short period between 02/03/2028 and 02/08/2028 involving 8 flow pulses are represented as time series graphs in the appendix, and discussed below:

- **Wet Unmitigated** – Average concentrations in Perry River (C14) were 0.16 mg/L for sulfate and below detection for cyanide and arsenic when the average water depth was 0.81m. In Swindon Creek near MRMB44 (C231_2) average concentrations were 85 mg/L for sulfate, and below detection for cyanide and arsenic when the average water depth was 14 cm. In Swindon Creek near MRMB12 (C166) average concentrations were 533 mg/L for sulfate, 0.033 mg/L for cyanide and 0.025 mg/L for arsenic when the average water depth was 3 cm.

- **Wet Mitigated** – Average concentrations in Perry River (C14) were 0.14 mg/L for sulfate and below detection for cyanide and arsenic when the average water depth was 0.81m. In Swindon Creek near MRMB44 (C231_2) average concentrations were 85 mg/L for sulfate, and below detection for cyanide and arsenic when the average water depth was 14 cm. In Swindon Creek near MRMB12 (C166) average concentrations were 337 mg/L for sulfate, and below detection for cyanide and arsenic when the average water depth was 3 cm.

- **Dry Unmitigated** - Average concentrations in Perry River (C14) were 5 mg/L for sulfate and below detection for cyanide and arsenic when the average water depth was 0.69m. In Swindon Creek near MRMB44 (C231_2) average concentrations were 111 mg/L for sulfate, below detection for cyanide and 0.001 mg/L for arsenic when the average water depth was 12 cm. In Swindon Creek near MRMB12 (C166) average concentrations were 650 mg/L for sulfate, 0.032 mg/L for cyanide and 0.026 mg/L for arsenic when the average water depth was 2 cm.

- **Dry Mitigated** - Average concentrations in Perry River (C14) were 5 mg/L for sulfate and below detection for cyanide and arsenic when the average water depth was 0.69m. In Swindon Creek near MRMB44 (C231_2) average concentrations were 111 mg/L for sulfate, below detection for cyanide and 0.001 mg/L arsenic when the average water depth was 12 cm. In Swindon Creek near MRMB12 (C166) average concentrations were 460 mg/L for sulfate, below detection for cyanide and 0.004 mg/L arsenic when the average water depth was 2 cm.

Mitigation effects were evident for sulfate, cyanide and arsenic in groundwater that exfiltrates into Swindon Creek near MRMB12 during wet and dry climate regimes. When the river bed is effectively dry the mitigated scenarios predict lower COPC concentrations when compared with unmitigated scenarios (appendix), with solutes moving along the ephemeral stream network as predicted by the conceptual model (Figure 2, Figure 3, Figure 7).

To understand how the mitigations will stabilise and reverse the potential for mining influence on wetted habitat, model outputs for wetted habitat were partitioned as a subset of data requiring closer review by removing data representing water depths shallower than 1mm. The minimum values, 20th, 50th and 80th percentiles and maximum values were calculated for wetted habitat, and compared with the default toxicant guideline (ANZG,2018). Without mitigation there is risk that surface water cyanide and arsenic concentrations in Swindon Creek near MRMB12 (C166) may periodically exceed the default guideline. With the management actions proposed for the B-B' flowpath, the risk is substantially mitigated, but the modelled 80th percentile for arsenic indicates that the default guideline concentration for arsenic III (0.024 mg/L) could still be marginally exceeded on occasion. As such the efficacy of mitigations will need to be monitored with a view to supplementary mitigation if required.

The proposed mitigation of mining affected groundwater along the A-A' flowpath assumes effective capture of mining-affected stormwater, which underpins the need to extend the northern seepage drain because of its role at capturing stormwater as well as capturing groundwater seepage. The proposed mitigations are expected to slow solute breakthrough along the A-A' flowpath by about 3 years, after which a mine closure strategy for this flowpath needs to be developed and implemented. The 3 year delay in solute breakthough was not well represented during the 13 year response between 2020 and 2033, because of its brevity. Closure may involve wetland remediation if field trials prove effective, and future model scenarios of wetland remediation along the A-A' flowpath can be used to evaluate the efficacy of the proposed closure strategy.

6 Conclusion and Recommendations

A key outcome of the EE is the identification of all potential options to prevent further releases of mining-affected water to groundwater, and remedial measures to reduce contaminant concentrations in groundwaters. Also required is an estimation of corresponding implementation timeframes and costs for these seepage mitigation actions.

The current seepage management plan has demonstrated prevention of AMD at the source through effective measurement of carbon and sulfur in mined rock that has allowed NAF to be seperated from PAF for separate handling. PAF is disposed of either below the water blanket in the TSF (where water limits diffusion of oxygen into the disposal environment, and carbonate, albite and chlorite neutralize acid generation), or in a compartment of non-acid-forming rock in a capped waste rock dump. This grade control and selective waste rock management strategy has been proven to prevent AMD from occuring, and significantly reduce the contaminant load within seepage and rainfall runoff from operational areas.

The current seepage management plan captures and reuses or evaporates NMD from the TSF and WRD through a combination of dams, drains, sumps and dewatering bores. These mitigation strategies have been progressively developed during the history of the Mine, and management responses have demonstrated the stabilization or reversal of mining - affected water entering the aquifers connecting with Rawdon Creek, Upper Swindon Creek and Twelve Mile Creek. Despite capture of 564 m^3/day by the Seepage Interception Drain, some mining - affected water has continued to report to downgradient monitoring bores MRMB12, MRMB31, MRMB30 and MRMB65 situated close to the TSF and WRD.

A strategy to improve dewatering has been developed and will be implemented by MRO. These additional mitigation measures will support the interception of seepage pathways in Rawdon Creek and the western side of the TSF, which appear to have started to stabilize as a result of existing mitigations. The additional dewatering capacity is expected to further delay breakthrough of mining-affected groundwater seepage into upper Swindon Creek and Rawdon Creek.

The proposed mitigation will also equip bore MRMB31 with a pump and install two additional dewatering bores in the Northern Seepage Zone, to target deep seepage that passes below the Seepage Interception Drain and reports to MRMB31, MRMB30, MRMB12 and MRMB65. Electrical resistivity imaging transects parallel to the northern wall of the TSF and geological mapping in Swindon Creek, the unnamed creek and along the Seepage Interception Drain, were used to optimize the location of the two new dewatering bores. MRMB31 had already been deliberately positioned to capture underflow from the TSF once breakthrough had been established.

The design intent is to install dewatering bores that will develop cones of depression to draw down the groundwater levels intercepting groundwater flow paths to the level of the downstream flow control (the crest of the Perry River Weir), while utilising existing flow barriers in the aquifer where practical. This strategy will remove the hydraulic head potential that drives seepage from the TSF, with the zone of groundwater capture being the cones of depression developed by the dewatering bores. If monitoring during the 2021 – 2024 financial years do not demonstrate the required stabilization and trend-reversal of the total cyanide concentrations and SO_4/Cl ratios then it will be necessary to widen the cones of depression by installing more dewatering bores in the seepage pathways.

The anticipated diameter of each cone of depression is between 26 and 188 metres, which was calculated using the Sichardt formula for unconfined aquifers, assuming a 30 – 40m drawdown and the hydraulic conductivities assumed for saprolite (K = 0.0537 m/d) and fractured rock (K = 0.00179 m/d). The actual diameters of the cones of depression will be calculated based on pumping tests to be performed in the field after the bores are commissioned.

The proposed implementation schedule and associated costs are included in Table 1.

Catchment Zone	Issue / area of concern	Strategy	Intended outcome	Budgeted Cost ($) approx.	Target completion date
A – A' TSF Upper Swindon Creek	Seepage mitigation between the TSF and Upper Swindon Creek	I. Install solar dewatering bore near MRMB43/44 II. TSF seepage interception and stormwater drainage improvements	Generate cone of depression and limit the effective hydraulic head to slow groundwater flow towards upper Swindon Creek riparian area.	$32,000 $400,000	1st January 2021 1st November 2021
B – B' TSF Lower Swindon Creek	Seepage mitigation between the TSF and Lower Swindon Creek	I. Install solar dewatering bore in MRMB31 II. Install solar dewatering bore downgradient of seepage pathway between MRMB8 and MRMB9 III. Install solar dewatering bore upgradient of MRMB12	Generate cones of depression that slow groundwater flow towards lower Swindon Creek riparian area.	$64,000	1st January 2021
C – C' TSF Rawdon Creek	Seepage mitigation between the TSF and Rawdon Creek	I. Install solar dewatering bore near toe of SD3, upstream of MRMB21 (Rawdon Creek C – C flow-path)	Generate cone of depression that slows groundwater flow towards lower Rawdon Creek riparian area.	$32,000	1st January 2021
D – D' Twelve Mile Creek	Seepage mitigation between the NWRD and Twelve Mile Creek	I. Geophysical survey to investigate the extent of aquifer connections II. Install solar dewatering bore north west of WD2 (near cattle yards)	Generate cone of depression that slows groundwater flow towards Twelve Mile Creek (MRMB27).	$40,000 $32,000	Completed 1st January 2021

Table 1: Implementation Schedule – Preferred options

Figure 14 shows the location of the existing and proposed water management units at the Mine.

Figure 14: Proposed Locations of Water Management Units

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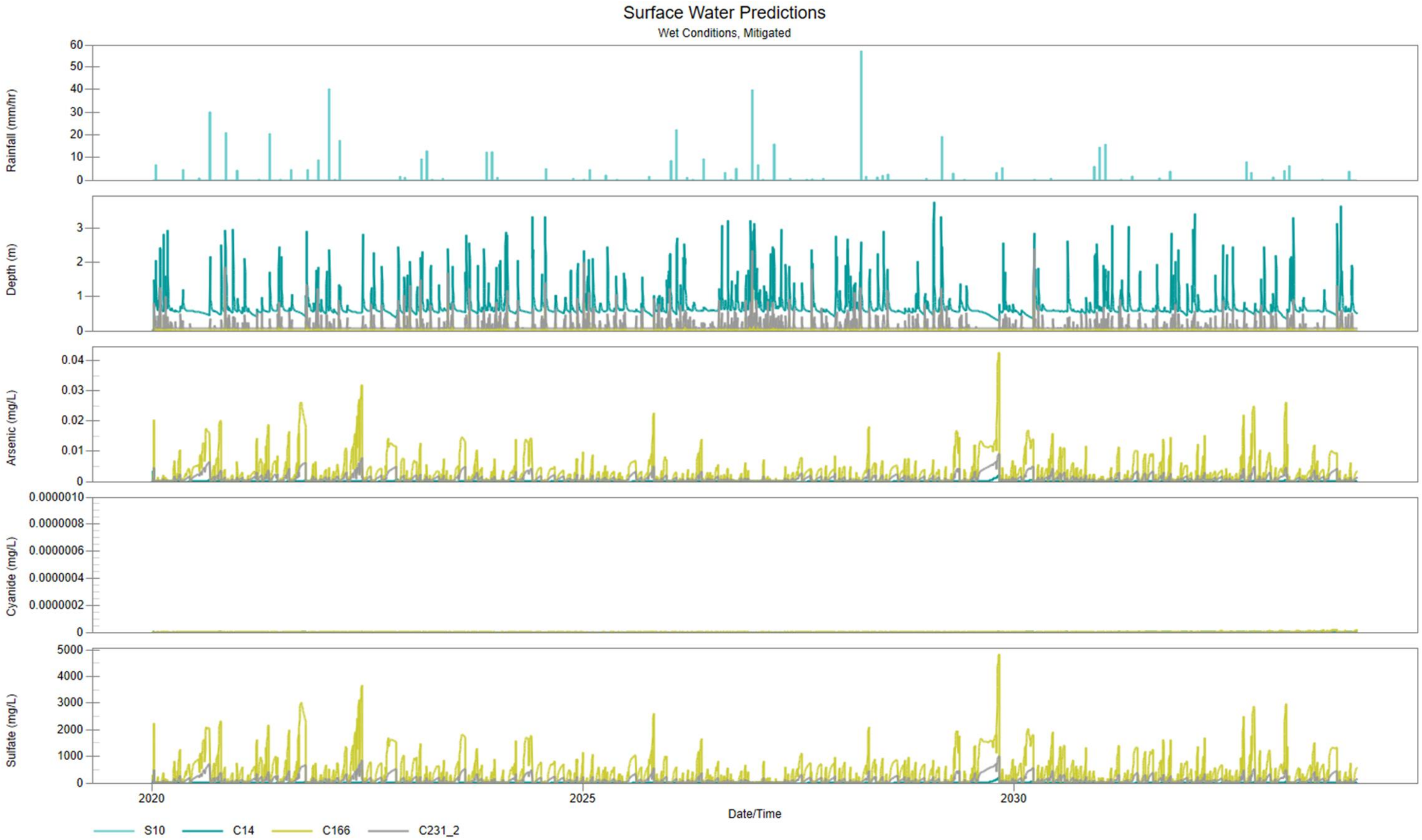
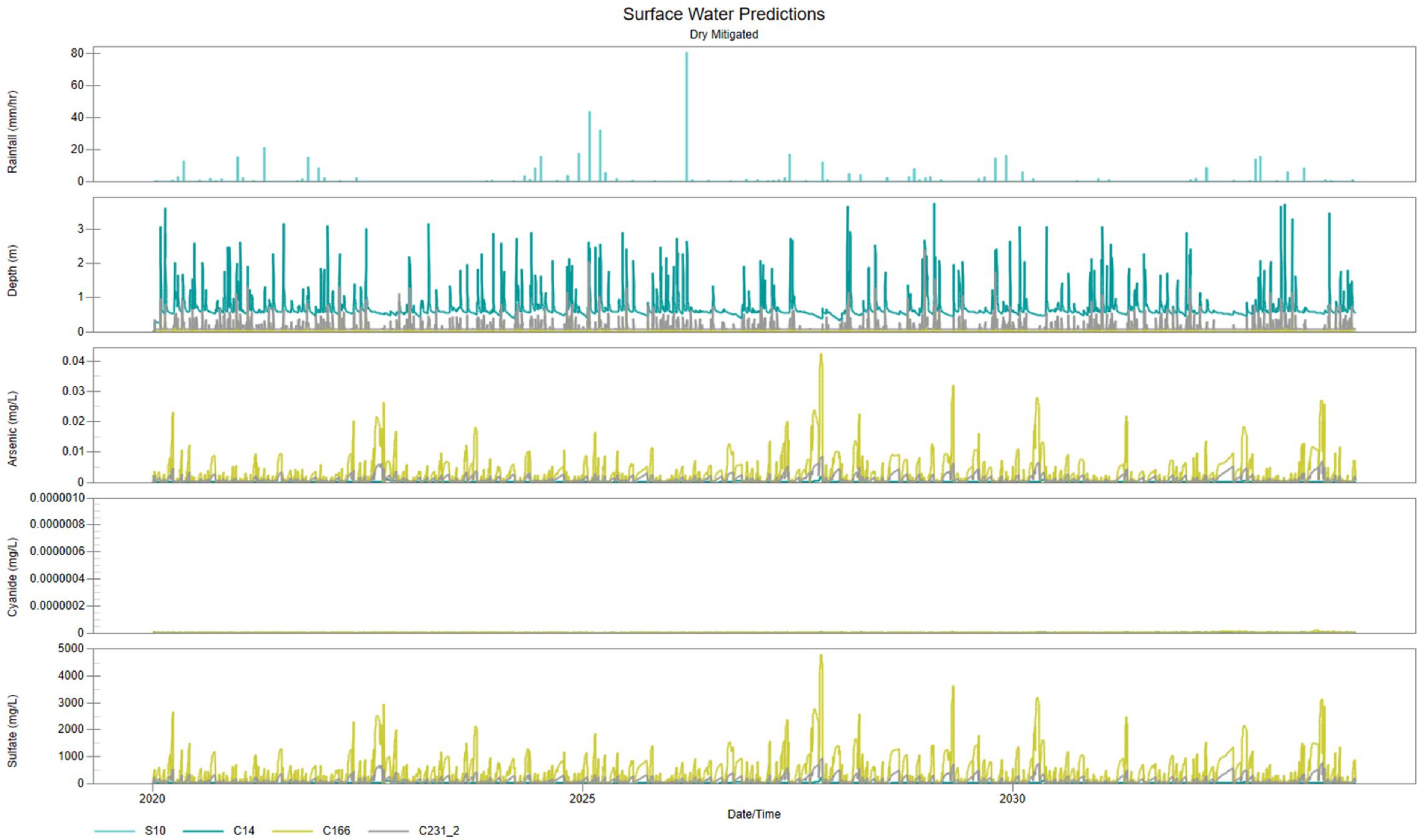
8 Appendix



Surface Water Predictions
Wet Conditions, Mitigated
Rainfall (mm/hr)
60
50
40
30
20
10
0
Depth (m)
3
2
1
0
Arsenic (mg/L)
0.04
0.03
0.02
0.01
0
Cyanide (mg/L)
0.0000010
0.0000008
0.0000006
0.0000004
0.0000002
0
Sulfate (mg/L)
5000
4000
3000
2000
1000
0
2020
2025
2030
Date/Time
S10
C14
C166
C231_2


Surface Water Predictions
Wet Conditions, Unmitigated
Rainfall (mm/hr)
Depth (m)
Arsenic (mg/L)
Cyanide (mg/L)
Sulfate (mg/L)
2020
2025
2030
Date/Time
S10
C14
C166
C231_2


Surface Water Predictions
Dry Mitigated
Rainfall (mm/hr)
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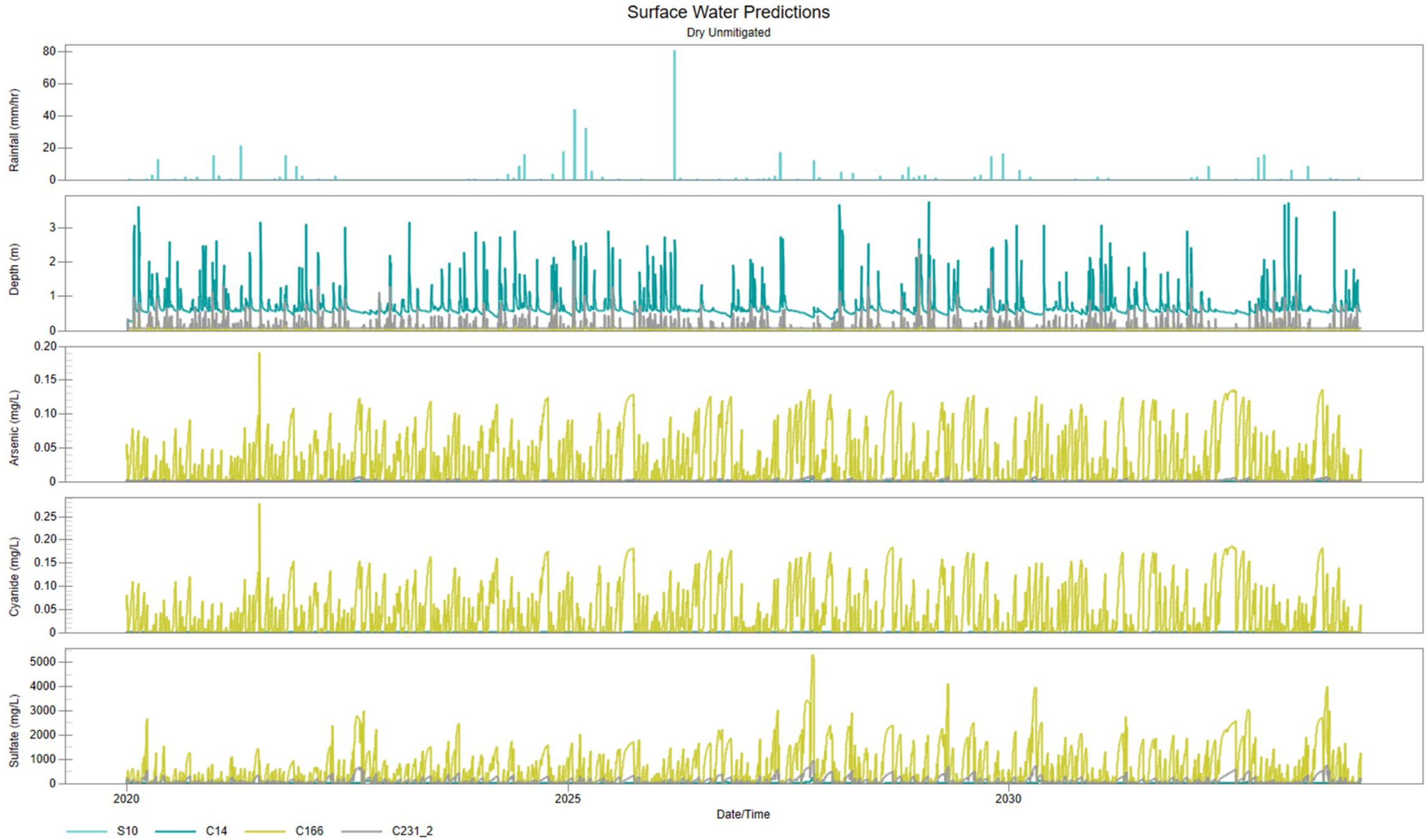
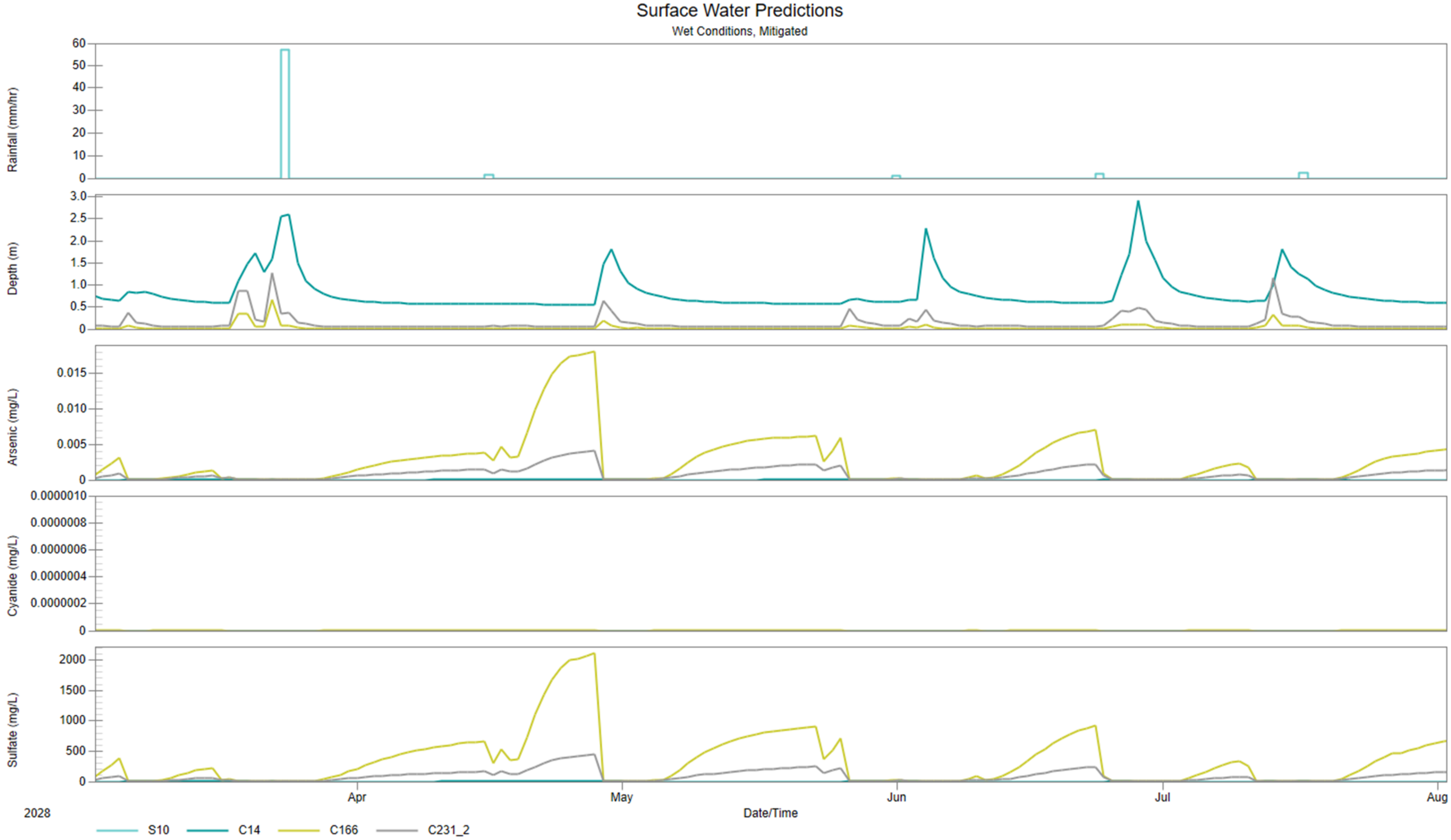
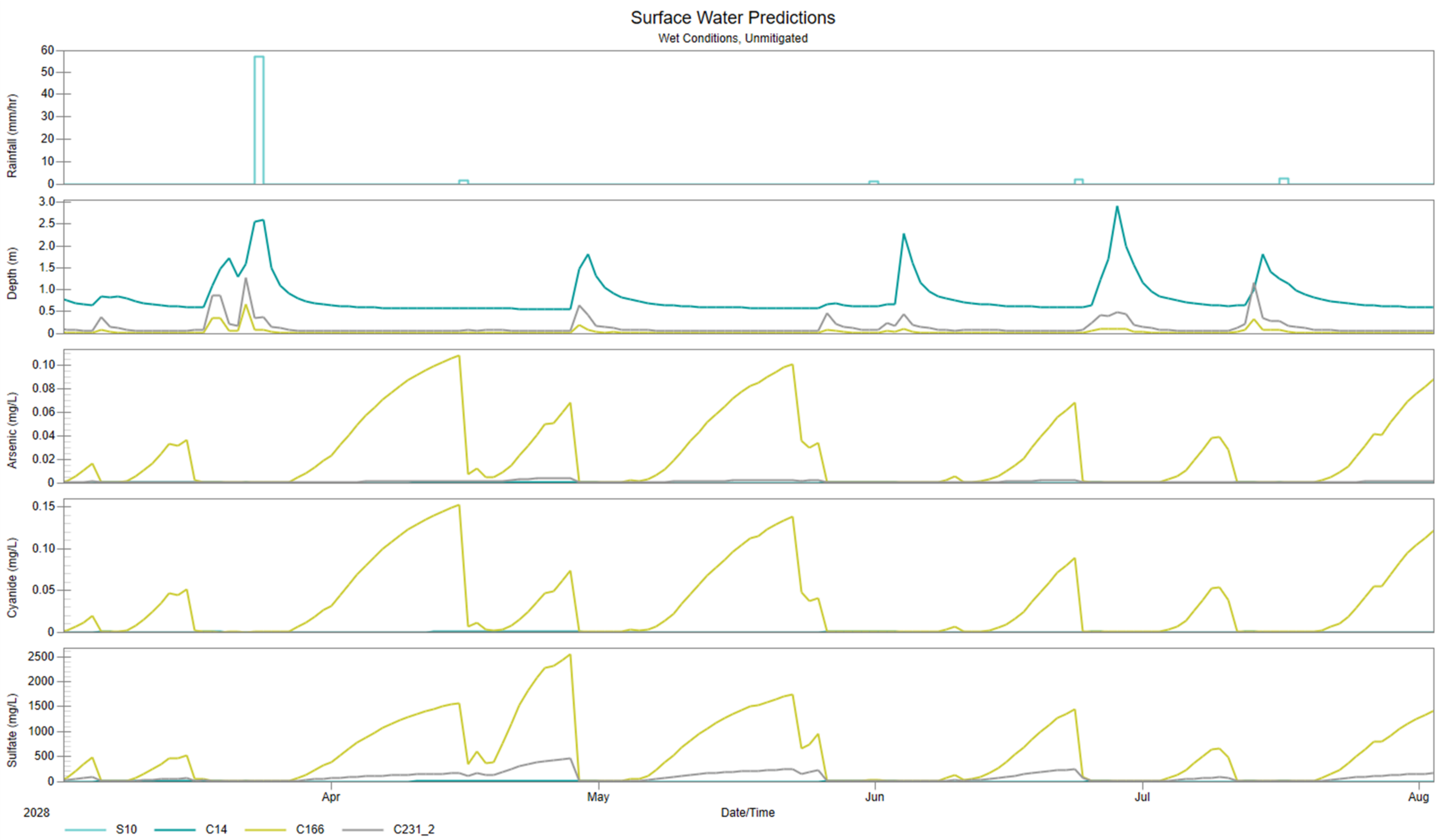
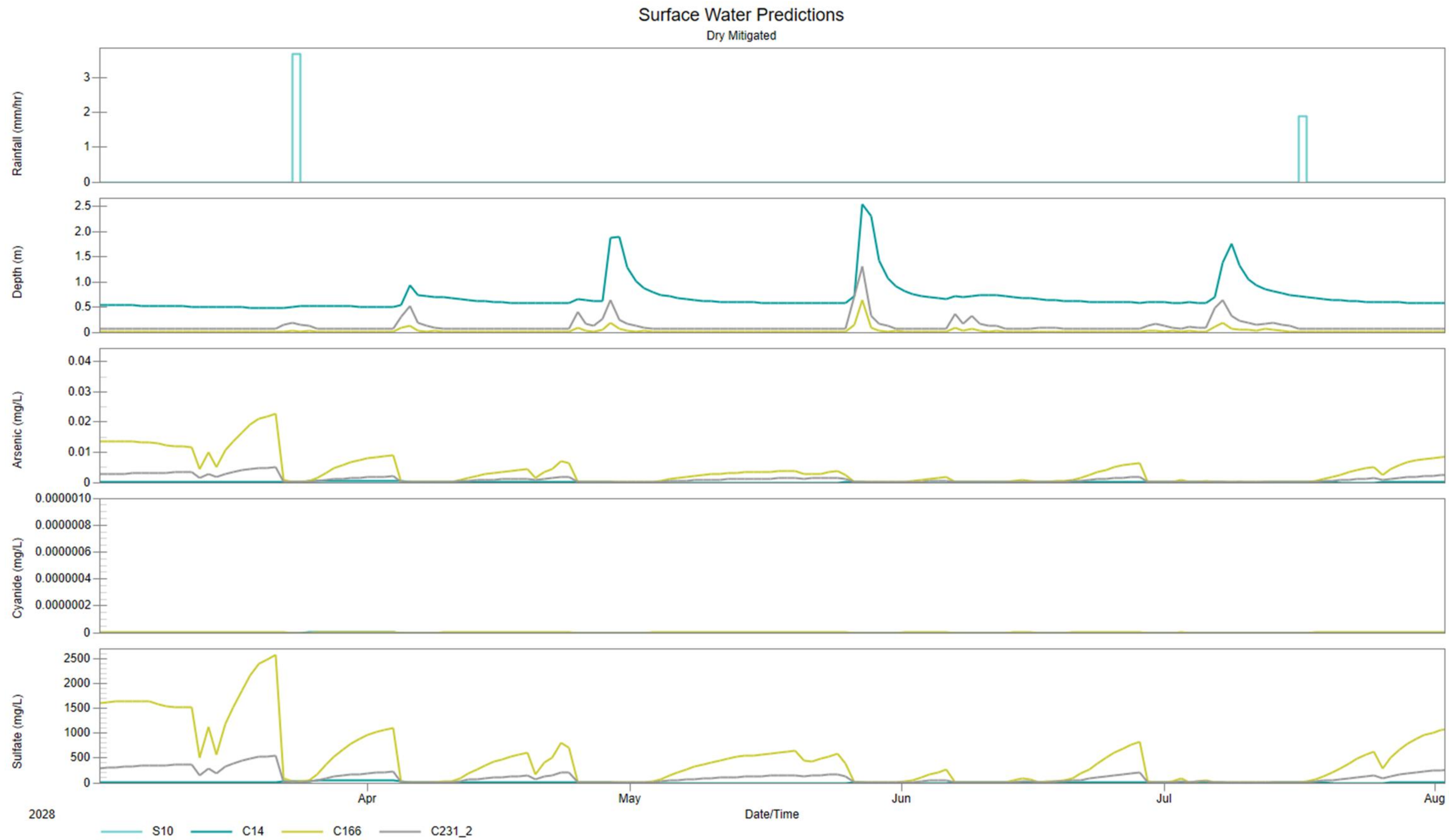
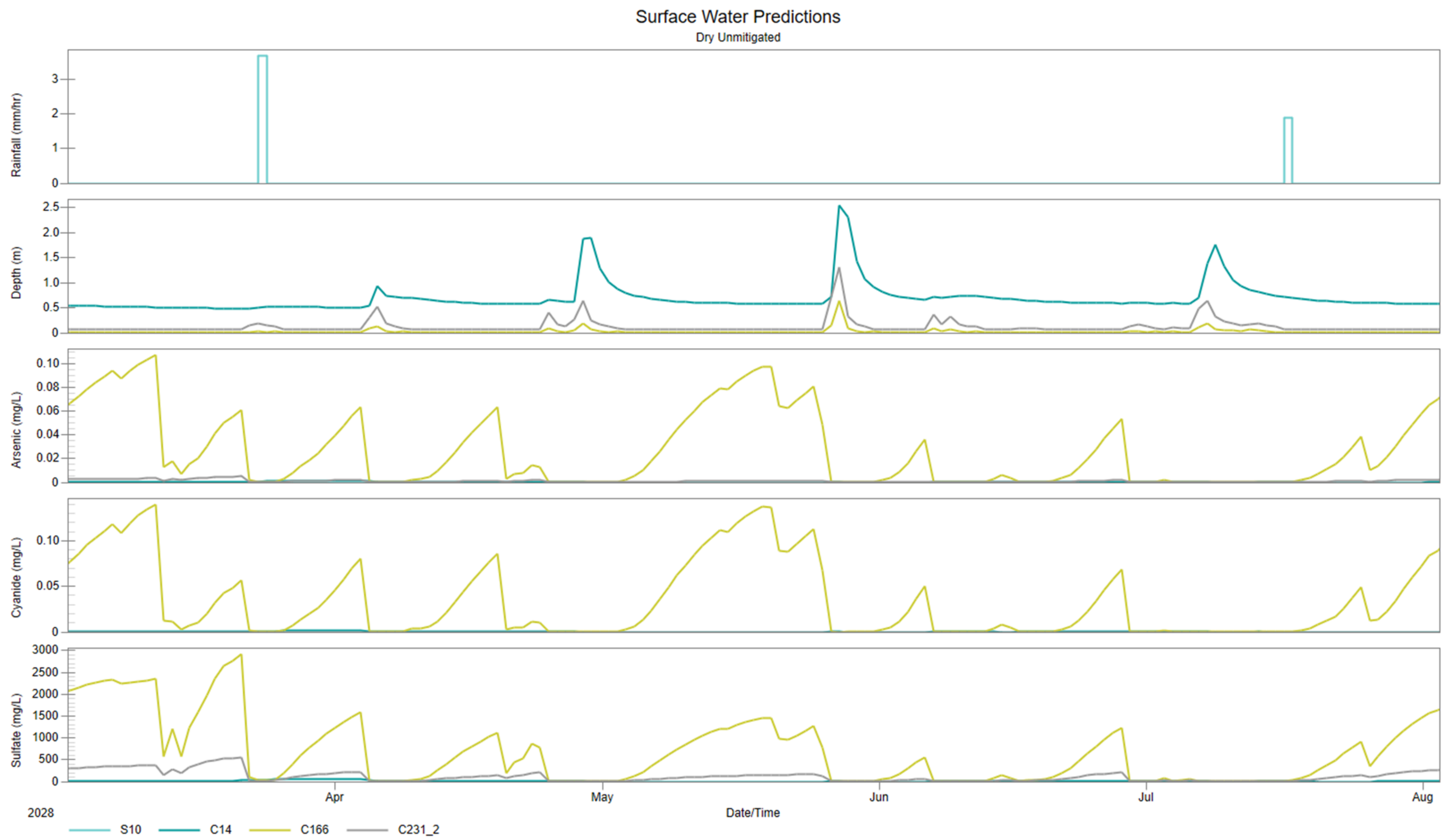
60
40
20
0
Depth (m)
3
2
1
0
Arsenic (mg/L)
0.04
0.03
0.02
0.01
0
Cyanide (mg/L)
0.0000010
0.0000008
0.0000006
0.0000004
0.0000002
0
Sulfate (mg/L)
5000
4000
3000
2000
1000
0
2020
2025
2030
Date/Time
S10
C14
C166
C231_2


Surface Water Predictions
Dry Unmitigated
Rainfall (mm/hr)
80
60
40
20
0
Depth (m)
3
2
1
0
Arsenic (mg/L)
0.20
0.15
0.10
0.05
0
Cyanide (mg/L)
0.25
0.20
0.15
0.10
0.05
0
Sulfate (mg/L)
5000
4000
3000
2000
1000
0
2020
2025
2030
Date/Time
S10
C14
C166
C231_2


Surface Water Predictions
Wet Conditions, Mitigated
Rainfall (mm/hr)
Depth (m)
Arsenic (mg/L)
Cyanide (mg/L)
Sulfate (mg/L)
Apr
May
Jun
Jul
Aug
2028
Date/Time
S10
C14
C166
C231_2


Surface Water Predictions
Wet Conditions, Unmitigated
Rainfall (mm/hr)
Depth (m)
Arsenic (mg/L)
Cyanide (mg/L)
Sulfate (mg/L)
2028
Apr
May
Jun
Jul
Aug
Date/Time
S10
C14
C166
C231_2


Surface Water Predictions
Dry Mitigated
Rainfall (mm/hr)
Depth (m)
Arsenic (mg/L)
Cyanide (mg/L)
Sulfate (mg/L)
Apr
May
Jun
Jul
Aug
2028
Date/Time
S10
C14
C166
C231_2



Surface Water Predictions
Dry Unmitigated
Rainfall (mm/hr)
Depth (m)
Arsenic (mg/L)
Cyanide (mg/L)
Sulfate (mg/L)
Apr
May
Jun
Jul
Aug
2028
Date/Time
S10
C14
C166
C231_2

PCSWMM MODELLED SCENARIOS			Arsenic (mg/L)			Cyanide (mg/L)			Sulfate (mg/L)		
			C14	C166	C231_2	C14	C166	C231_2	C14	C166	C231_2
Wet	Mitigated	Min	3.4342E-12	5.7442E-09	6.6403E-09	1.8329E-21	6.3421E-20	0	3.8191E-07	0.00064279	0.00073043
		20%ile	7.8066E-10	9.5066E-06	7.6406E-06	3.1705E-19	5.9561E-15	0	8.6954E-05	1.1561206	0.84046188
		50%ile	5.0045E-08	0.00052302	0.00024476	4.6851E-17	1.0183E-12	0	0.00543779	70.69759	26.923965
		80%ile	1.7485E-05	**0.00303017**	0.00090824	2.7083E-14	1.3954E-11	0	1.909161	**383.23902**	99.906336
		Max	**0.0043306**	**0.04205364**	**0.00890964**	1.3793E-10	8.2435E-09	0	**480.7423**	**4753.219**	**980.0599**
	Unmitigated	Min	7.865E-12	1.022E-08	6.6403E-09	6.4926E-12	1.9454E-09	0	4.219E-07	0.00066865	0.00073043
		20%ile	1.656E-09	3.5453E-05	7.6406E-06	1.182E-09	1.9699E-05	0	9.3909E-05	1.497505	0.84046188
		50%ile	1.253E-07	**0.00381438**	0.00024476	1.0079E-07	0.00348625	0	0.00604352	**102.9205**	26.923965
		80%ile	4.9061E-05	**0.02509105**	0.00090824	4.5817E-05	**0.03033187**	0	2.131102	**581.20956**	99.906336
		Max	**0.00686233**	**0.09584825**	**0.00890964**	0.00370871	**0.08694897**	0	**482.108**	**5251.689**	**980.0599**
Dry	Mitigated	Min	1.5159E-11	9.534E-09	1.2916E-08	3.2932E-22	2.1842E-20	0	1.6781E-06	0.00108129	0.00142076
		20%ile	8.3762E-10	9.9805E-06	8.3697E-06	3.0716E-19	5.5629E-15	0	9.3171E-05	1.1802596	0.92066736
		50%ile	4.1026E-08	0.00050433	0.00024308	5.9596E-17	1.0907E-12	0	0.00450292	70.33283	26.73911
		80%ile	2.3317E-05	**0.00302324**	0.0008949	2.1157E-14	1.154E-11	0	2.5002774	**384.6155**	98.43885
		Max	**0.00164678**	**0.04206381**	**0.00802248**	1.0488E-10	7.4445E-09	0	**175.1336**	**4744.804**	**882.4733**
	Unmitigated	Min	2.5737E-11	1.973E-08	1.2916E-08	1.5501E-11	1.8975E-09	0	1.769E-06	0.00121742	0.00142076
		20%ile	1.6313E-09	3.3646E-05	8.3697E-06	1.1425E-09	1.8848E-05	0	9.8225E-05	1.5178282	0.92066736
		50%ile	1.0411E-07	**0.00393233**	0.00024308	8.4558E-08	0.00380364	0	0.00490252	**102.9405**	26.73911
		80%ile	6.7936E-05	**0.02585224**	0.0008949	6.4212E-05	**0.0313296**	0	2.8488066	**577.41352**	98.43885
		Max	**0.00397997**	**0.1906317**	**0.00802248**	0.00341892	**0.2782624**	0	**194.6239**	**5163.146**	**882.4733**

Emboldened font exceeds the default toxicant guideline limits (mg/L) in ANZG (2018): As III 0.0024, As V 0.013, CN 0.007