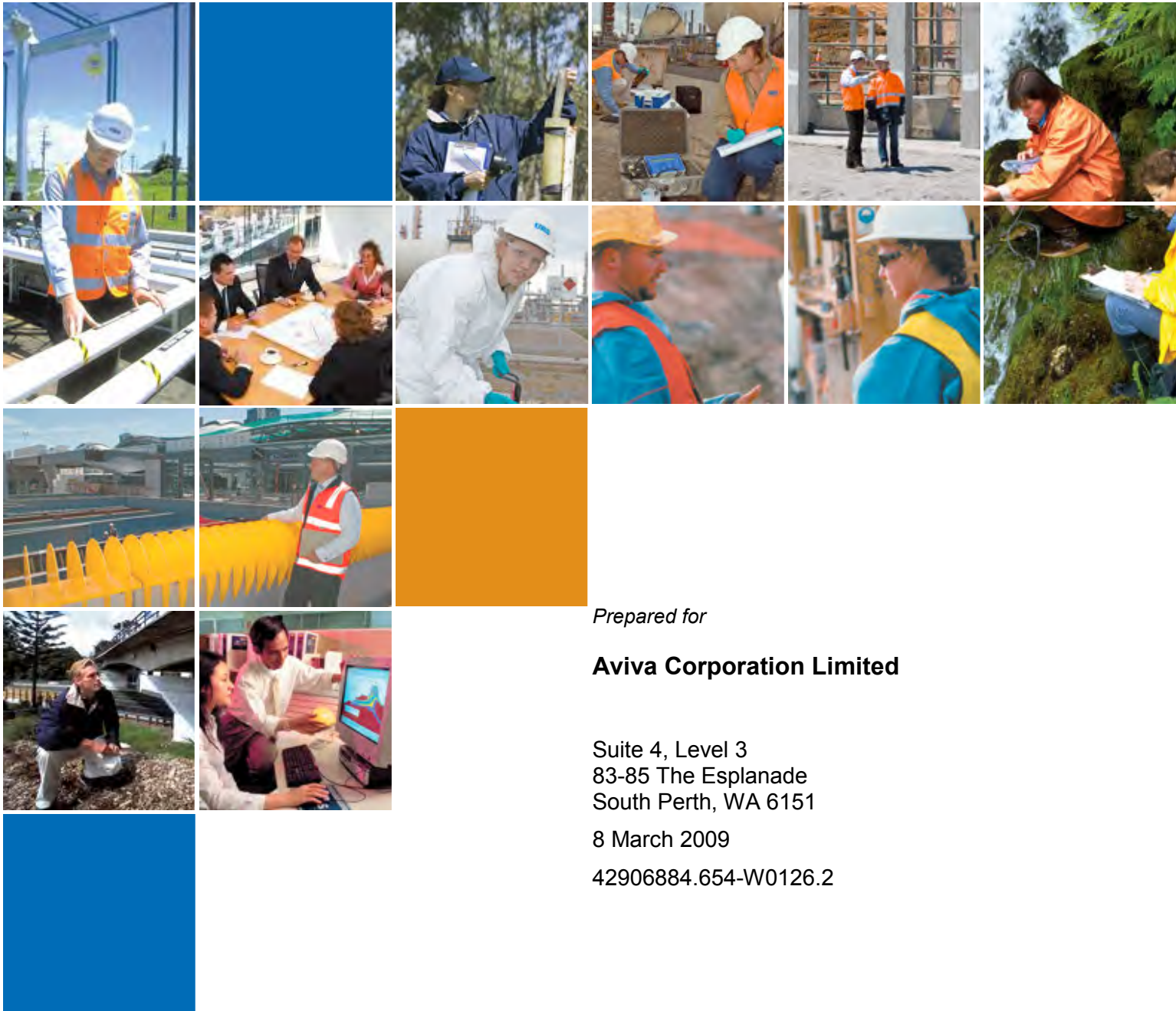# FINAL REPORT

Predicted Impacts of Co-Disposal Options on Groundwater - Central West Coal and Coolimba Power Projects

Prepared for

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Executive Summary

URS Australia has carried out an assessment of potential impacts on groundwater from the co-disposal of coal combustion ash with waste rock at the Central West Coal and Coolimba Projects in Eneabba. The assessment reviewed baseline groundwater quality data and assessed the potential impacts of pit dewatering, ash co-disposal and salt co-disposal on groundwater levels and quality during mining and for closure. Solute transport models were developed to predict water levels and water quality changes during mining and after closure, including the final void.

Two co-disposal options have been evaluated with the solute transport models. The disposal options are defined below.

- **Ash Co-Disposal Option** – This includes annual disposal of waste rock (overburden) and coal combustion ash to the mine pit. Details are provided in Section 4.

- **Salt Co-Disposal Option** – This includes annual disposal of waste rock (overburden) and coal combustion ash to the mine pit, and disposal of evaporative salt every four years. Details are provided in Section 5.

Baseline Groundwater Quality

Groundwater chemistry maps (including salinity, pH and trace metals) have been produced for groundwater in the shallow superficial aquifer and deep confined aquifers (including the Cattamarra Coal Measures, Yarragade aquifer and Eneabba Formation). Groundwater salinity in the superficial aquifer changes generally from fresh/brackish in the east to very saline towards the west. Local hypersaline areas are present where the water table is close to the land surface and subjected to a strong evaporation process.

Groundwater salinity in the confined aquifers is generally fresh/brackish in the area to the east, north and south of the mine area, and increases towards the west and north-west.

There is a high salinity zone in both the superficial and deep confined aquifers extending from the west of the mine area. This high salinity zone may be correlated with the discharge of saline groundwater from the Eneabba Formation and the Cattamarra Coal Measures.

Groundwater pH values in the project area range from 6 to 8 and are neutral on average.

Groundwater chemistry varies considerably with location and depth. In both shallow and deep aquifers, the baseline groundwater concentration values of Aluminium, Copper (Cu), Molybdenum (Mo), and Zinc (Zn) generally exceeded the guideline values applied to various end users including aquatic ecosystems, terrestrial groundwater dependent ecosystems as well as that required for primary production (stock) and recreational purposes. The general applicable guidelines used in this project include ANZECC Guidelines for Fresh and Marine Water Quality - Aquatic Ecosystems; and ANZECC Guidelines for Freshwater and Marine Water Quality - Livestock Drinking Water.

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Hydrochemistry of Leachate from the Co-Disposed Waste

The mining operation will involve a backfilling operation, where the waste rock (overburden) will be progressively spread out in the void together with the coal combustion ash generated from the Coolimba Power Station, and salts generated from the evaporative ponds of the Power Station.

Leaching tests were carried out by Terrenus Earth Sciences (2008) to identify the chemical characteristics of the leachate that may be released from the backfilled waste rock and coal combustion ash. The tests found that the salinity of the leachate from the waste rock ranges from 92 mg/L to 604 mg/L, and pH ranges from 5.5 to 7.0. The soluble metal concentrations are all below the detection limits.

Leaching tests on the coal combustion ash showed that the salinity ranges from 505 µS/cm to 558 µS/cm, and pH ranges from 7.1 to 8.2. Solute metal concentrations in the leachate from the coal combustion ash are higher than that from the waste rock. The metals of concern are Al, As, B, Cr, Cu, Mo and Zn. Concentrations for other metals are below the detection limits.

Multi-element tests by Terrenus Earth Science (2008) indicated that nutrient and organic matter concentrations in ash solids are low, and in most cases, below the detection limits. Leaching tests indicated that nutrient and organic matter concentrations in ash leachate are low and below the relevant guidelines.

It is estimated that the Power Station will generate about 19,829 tonnes of salts per year, of which sodium chloride will be the predominant salt (15,365 tonnes). The salts, if dissolved by rainfall infiltration or groundwater, will entre the local groundwater system and cause the groundwater salinity to increase. However, it is thought that the salt plume would be largely captured through pit dewatering and by the final void.

Impact of solute transport from the ash co-disposal on the superficial aquifer was thought to be insignificant, as the leachate concentrations (including salts and metals) entering the groundwater system are lower than that in the local groundwater. The pit dewatering and final void would capture the solute plume during the mining and post mining period. The final vid would become a saline lake that receives groundwater seepage and would not cause significant impact on the local groundwater system.

In summary, salinity and metal concentrations of the leachate from the waste rock and coal combustion ash are lower than the baseline levels in the local groundwater. Potential impact from the metals with elevated concentrations was assessed with a solute transport model.

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Prediction of Ash Co-Disposal Impacts on Groundwater Quality

A solute transport model was developed based on an existing flow model prepared by Rockwater to assess potential impacts of ash co-disposal on groundwater quality during mining and for closure, including the final void. Based on the mining scheduling, the model simulated the solute movement and delineated the relative concentration plumes of a general solute and the metals associated with the coal combustion ash.

The following results were obtained from the solute transport modelling for the 30-year mining operation period.

- Solute concentration plumes occurred in the CCM. These plumes should remain primarily within the CCM due to operational drawdowns (dewatering) and post closure residual drawdowns (close to the final void)

- The highest solute concentrations always followed the backfilling process. The solute concentration plume was generally retained in the mined area and captured by pit dewatering. Dispersion effects to areas outside the pit were small due to steep hydraulic gradients close to the pit. Solute left behind in the backfilled areas are expected to be partially captured by dewatering due to preferential flow through backfill that is likely to be more transmissive than undisturbed CCM.

- The model also predicted that a large part of residual plume to the south of the mined area will remain within the mined area and may be transported very slowly by the north-westerly groundwater flow. However, the salt and metal concentrations in the residual plume, as well as the groundwater flow rate are expected to be very low. As a result, the change that this residual solute will impart on the local groundwater system is probably also very small.

Predictions of water level and water quality for mine closure and the final void showed that:

- Water level in the final void and groundwater level around the final void recovered after the mining. Due to high evaporation rate from the open surface water, the final mined void will form a groundwater sink and be the local focus of groundwater flow. It was predicted that the final water level in the void is within 4 m of the original watertable. The residual drawdown cone should stabilise after 60 years of mine closure.

- The final void is expected to form a capture zone where metals and salt in the concentration plume generated from the ash co-disposal will be transported by groundwater flow to the final void.

- At the same time, the void will continue to receive the groundwater seepage, which brings salt and dissolved metals from the surrounding areas. It is predicted that the salt concentration will be built-up progressively in the void, due to evaporation of water from the void.

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- The water quality in the final void will be determined by a cumulative process that comprises capturing of the residual solute plume generated from the ash co-disposal and concentrating of natural salts through evaporation. Long term salinity and metal concentrations were determined by water and mass balance calculations on outputs from the flow and solute transport models. It was found that the pre-mining groundwater salinity at the final void site was at 1,575 mg/L. This salinity level was reduced slightly when the leachate plume (dominated by rainfall recharge) reached the site. Water salinity in the final void increased gradually over the long term and reaches super-saline (with TDS over 100,000 mg/L) after 500 years of the mine closure. This trend is likely to continue until the water reaches its maximum concentration as determined by salt saturation indices, probably in excess of 300,000 mg/L TDS.

- The influence of the final void on the local groundwater level and water quality is considered to be small and localised. Salt and metal concentration plumes should be confined within the mined area and continually captured by the final void.

- Movement of solute, especially metals, can be affected by geochemical reactions, including absorption and cation exchange where metals are bound to aquifer materials. It is noted that the above solute transport modelling is based on non-reactive solute transport and the model result therefore represents a worst case scenario.

- Results from the solute transport model reflect predominantly the hydraulic and solute transport processes for the Cattamarra Coal Measures, Yarragadee Formation and Eneabba Formation. The modelling results should not be used to emphasise the effect on the shallow watertable aquifer.

Prediction of Salt Co-Disposal Impacts on Groundwater Quality

The following results were produced by the 30-year mining dewatering simulation of the salt co-disposal option which includes disposal of evaporative salt (every four years), and waste rock and coal combustion ash (every year).

- Salinity level in the salt disposal areas is higher than the surrounding groundwater salinity, and increases.

- There is a strong dilution effect. The mined area is a local sink due to dewatering and preferential flow. As a result, the local groundwater flows towards the mined area, mixes with the hypersaline leachate and dilutes the hypersaline leachate salinity concentrations.

- Although the simulated leachate concentration is 320,000 mg/L, the loading is comparatively small due to the low flux or infiltration rate. So the effect is local during the mining period.

- Groundwater flow within the mined area is northerly, aligned to the mine path. The groundwater flow velocity is low as the CCM is of comparatively low transmissivity. This results in low dispersion of the salt.

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The following results were produced by the 500-year post mining simulation.

- Salinity concentrations beneath the salt co-disposal areas increase continually and become hypersaline after only a few years. The effect of this hypersaline salinity on the local groundwater flow is poorly defined by the current model. Density driven solute transport modelling would be required to provide a reasonable assessment.

- The plume is predominantly located in the mined area and moves towards the final void.

- The final void would host crystalline salt. The final void has an estimated capacity to contain between 27 Mt to 51 Mt of crystalline salt. The annual NaCl salt production from the evaporation ponds is 15,438 tonnes/yr), about 0.03% to 0.06% of the final void capability.

- The long-term salinity concentration increases much more rapidly for the salt co-disposal option than for the ash co-disposal alone.

In summary, the result from the simplistic solute transport simulation of the salt, overburden and ash co-disposal suggests that the salt disposal option may produce significant salt accumulations in the groundwater system in the mined area, in particular in the final void. The system does not reach a steady state condition during the 500-year simulation period.

It is noted that this work is based on non-reactive and non-density driven simulations. The parameters used in the current modelling may only provide an order-of-magnitude estimate, due to the lack of geochemical testing data and absence of a density-driven modelling platform.

It is recommended that a groundwater monitoring program be developed to monitoring the solute plume movement and capture by the final mine void. The model also needs to be refined to provide necessary absolute water level and water quality predictions for the mining operation.

It is also recommended that water quality criteria be developed for the major groundwater users in the area.

1.1   Background

Aviva Corporation Limited (Aviva) proposes to develop the Central West Coal and Coolimba Power Projects located 6 km south of Eneabba (Figure 1.1). The Projects will involve the mining of the Cattamarra Coal Measures (CCM) to extract sub-bituminous coal resource from the Central West Coal Mine to energise the adjacent proposed 440 MW Coolimba Power Station.

It is estimated that the power station will generate about 820,000 tonnes of ash each year which is intended for co-disposal at the Central West Coal Mine, within backfilled waste rock (overburden) above and below the water table. Salts generated from the evaporative ponds of the Coolimba Power Station will also be co-disposed with overburden and ash. About 19,829 tonnes of salt would be disposed each year of which sodium chloride will be the predominant salt (15,365 tonnes).

The proposed co-disposal of overburden rocks, evaporative salts and power station ash has potential infiltrate the water table and saturated superficial formations and CCM successions with salts and metals and impact upon the beneficial use of the local groundwater resources. Previous studies on similar coal mine and power station operations, such as Collie, indicate that solutes from the co-disposed ash may be expected to be transported by rainfall infiltration to both the active mining environment and the water table (URS, 2007). As the water table recovers after mining, the solutes would also be transported by groundwater flow to the areas being mined and associated dewatering infrastructure. Subsequent to mining, the final mined void would be expected to form a long-term groundwater sink and be the local focus of groundwater flow. A solute transport model was developed based on an existing Rockwater groundwater flow model (Rockwater, February 2009) to simulate the above processes and predict the water quality changes during mining (30 years) and for closure, including the final void (500 years).

1.2   Mining and Dewatering Plan

The life of the mining operations is 30 years. The coal will be used as a fuel source for the proposed Coolimba Power Station. The mining operations will involve a backfilling operation, where waste rock will be progressively deposited in the mined void behind the active mine area. The waste rock (overburden) and ash generated from the Coolimba Power Project will be included in the backfill to the pit void. Following backfill, progressive rehabilitation of the mined areas to a final landform will be undertaken. The key characteristics of the Project are presented in Table 1-1.

The dewatering plan will involve in-pit sumps and/or production boreholes to dewater and depressurise the CCM successions being mined. This dewatering will create a drawdown cone in the local and regional CCM aquifer systems. The potential drawdown impact has been assessed by Rockwater based on data from drilling programmes and groundwater flow model (Rockwater, 2009).

Table 1-1   Key Characteristics of the Central West Coal Project

Element	Description
Life of Operations (excluding construction and closure activities)	Approximately 30 years
Mining Method	Open cut mining
Dewatering Method	In-pit sumps and/or bores.
Backfilling	Waste rock and coal combustion ash.
Final Void	Northern end of the mine within the former Iluka Eneabba West Mine.

1.3   URS Project Scope

The objective of URS's project is to predict the water quality changes as a result of the mining operation and ash co-disposal, and estimate the long term water level and water quality in the final void. Baseline hydrogeological and water quality assessments have been carried out to establish the existing (pre-mining) groundwater flow and hydrochemical processes by URS (2005) and Rockwater (2004, 2005 and 2009). Geochemical tests of backfilled waste rock and coal combustion ash were carried out by Terrenus Earth Science (2008) to identify water chemical parameters that may be of concern based on the requirements of groundwater end users and the environment.

The project scope included:

- **Undertake a review of the potential for leaching of metals and organics from the run-of-mine overburden and ash.**

Aviva has recently performed acidity testing on CCM samples from cored holes at the Central West Coal Project. URS will prepare a report outlining findings and addressing the requirements of the Department of Water (DoW), Department of Industry and Resources (Environment) (DoIR), and the Department of Environment (DEC), with specific reference to:

    a. DoW, *Mine Void Water Resource Issues in Western Australia*, 2003.

    b. DoIR, *Guidelines for Mining Proposals in Western Australia*, 2006.

    c. ARMCAMZ, *Australian Drinking Water Guidelines, 1996*.

- **Review geochemical parameters derived from the pilot scale test work undertaken by Aviva.**

Geochemical results on the waste materials and ash derived from the CCM were reviewed against other coal mines (including data from mines in the Collie Basin in Western Australia) to assist with parameter development. Groundwater and surface water quality data will be drawn from samples taken by Aviva. The data was incorporated into the model.

- **Review the current MODFLOW groundwater model developed by Rockwater for the dewatering assessment for the Central West Coal Project.**

The Rockwater model was modified (including boundary and initial conditions, aquifer layering, hydraulic parameters) to enable long-term groundwater flow and solute transport simulations. The modified model uses Visual MODFLOW for the flow and solute transport prediction. This model was parameterised using appropriate hydraulic, water balance and solute transport variables. The predominant groundwater model flow and recharge characteristics were derived from the Rockwater model where they have been established and calibrated. Other parameters were sourced from similar settings, such as the Collie Basin. Three models were used to simulate groundwater flow and solute transport for pre-mining, 30-year mining operation, and post mining with a final void scenarios.

- **Develop a transient water balance for the Central West Coal Mine during mining and for closure**

Groundwater flow during the mining operation (dewatering) and for closure (groundwater level recovery) was simulated with a transient model.

- **Develop salt and soluble metals balances for the Central West Coal Mine during mining and for closure (final voids).**

These salt and soluble metals balances were calculated using the result of the solute transport modelling and the laboratory leachate tests.

- **Simulate groundwater flow and solute transport at the Central West Coal Mine to investigate ash co-disposal impacts on the water table.**

The spatial distribution of the potential leachate plume was simulated using the model. The simulations took into account ash co-disposal as integrated with current mine plans, design pit dewatering borefields and the final mined void.

- **Simulate groundwater flow conditions and hydrochemical development of the final void to predict the final level and water quality after mine closure.**

A range of model scenarios have been run to simulate the groundwater level recovery and geochemical evolution in the final void. The long-term simulation predicted the final water level and water quality 500 years after mine closure, investigated impacts of the mine as a hydraulic sink and investigated the final void in terms of solute capture.

- **Simulate groundwater flow and solute transport at the Central West Coal Mine to investigate evaporative salt co-disposal impacts on the groundwater system.**

This is an additional scope. Spatial distribution of the potential leachate plume generated from co-disposal of the evaporative salt, waste rock and coal combustion ash was simulated for the mining operation period and closure including final void.

Section 2   Physical Setting

2.1   Climate and Geomorphology

The Eneabba region has a dry Mediterranean climate. The historical annual rainfall ranges from 286 mm to 968 mm. About 86% of mean annual rainfall occurs in winter months from May to October. The average annual evaporation for this area is 2,439 mm. Mean monthly rainfall in the mid-winter months from June to July generally exceeds evaporation (Figure 2.1). The recent annual rainfall data is also plotted in Figure 2.1. The annual rainfall has a declining trend, from about 570 mm/yr in 1980 to 420 mm/yr in 2007 with an average of 539 mm/yr.

The project is situated within the Swan Coastal Plain which comprises a series of late Tertiary and early Quaternary marine and eolian dune deposits. The drainage system, including the Erindoon Creek, Bindoon Creek and Eneabba Creek runs north westerly.

Lake Indoon is a permanent lake and a Class "A" reserve for public recreation registered by the Shire of Carnamah. Lake Logue is a seasonal lake and is classified as a "Natural Important Wetland" by the Department of Environment and Conservation (DEC).

2.2   Hydrogeological Setting

Early regional hydrogeological investigations in the study area have been summarised by WRC (1997), which included geological mapping of the superficial formations (Lowry, 1994; Mory, 1994), and hydrogeological interpretations by Commander (1978, 1980 and 1981). The regional stratigraphical units include superficial formations overlying a Mesozoic succession including the Eneabba and Lesueur Sandstone Formations in the west, and CCM in the Project area and the Yarragadee Formation in the east (Figure 2.2).

In the Project area, the superficial formations are underlain by the CCM. The Project site and CCM are situated between the Peron Fault to the west and the Warradarge Fault to the east (WRC, 1997; Rockwater, 2009).

Local hydrogeological conditions have been characterised by earlier investigations and groundwater monitoring by Iluka Resources Limited at the Eneabba West Mine (e.g. Rockwater 1990, 2002), and drilling and groundwater monitoring for the Eneabba East Mine (Rockwater, 2004, 2008; URS, 2007).

2.2.1   Superficial Formations

The superficial formations in the study area are generally less than 10 m thick. West of the Warradarge Fault (and Brand Highway), the superficial formations comprise laterite and sandy bleached sediments overlying CCM. Beneath low-lying areas to the west, the superficial formations thicken and extend below the water table. These successions typically include bedded clayey silt to silty sand of shallow marine and coastal origin. Overlying the deeper sediments are reworked deposits of the Bassendean Sand and (further west) Tamala Limestone.   The superficial formations form an unconfined superficial aquifer. The hydraulic conductivity of the superficial formations is about 1 m/day (Rockwater, 2009).

The water table is recharged by rainfall infiltration with an estimated recharge rate of 1 to 5 percent of the annual rainfall. The superficial formations also form a discharge zone for saline groundwater leakage from the underlying confined aquifers in the west where upward hydraulic gradients occur (Kern, 1997). Groundwater flow is west to north-westerly. Hydraulic gradients flatten across low-lying areas west of the Warradarge Fault escarpment.

2.2.2   Cattamarra Coal Measures

The CCM dips at a low angle towards the east and hosts the coal resource – the Central West Coal Deposit. The CCM comprises interbedded sandstone, siltstone, shale and coal seams, and form multilayered succession of semi-confined to confined aquifers and aquitards. The average hydraulic conductivity is 0.7 m/day according to interpretations of pumping tests in the Project area (Rockwater, 2009). Aquifer parameters from early drilling programmes at the Eneabba West Mine (Rockwater 1990) indicate transmissivity of 15 to 261 m^2/day, with an average of about 50 m^2/day. Recent pumping test results obtained by Rockwater in 2008 suggest a higher hydraulic conductivity in the Project area.

Regional groundwater flow in the CCM is mainly in a north-westerly direction, with heads ranging from over 70 m AHD in the southeast recharge area, to about 10 m AHD in the northwest discharge area. Groundwater recharge enters the CCM from the overlying superficial formations in marginally low-lying areas or via the lateritic and sandy cover in elevated areas. Groundwater discharge occurs through upward leakage into the overlying superficial aquifer in low lying areas to the west and southwest of the Project area.

2.2.3   Yarragadee Formation

The Yarragadee Formation is located to the east of Warradarge Fault. It comprises sandstone with interbedded shale and forms a regional Yarragadee aquifer. The hydraulic conductivity is about 3 m/day. The known aquifer transmissivity in the vicinity of Eneabba ranges from 366 to 446 m^2/day (Rockwater, 2002). The Warragadge Fault is interpreted to be a low-transmissivity zone trending north to south, separating the CCM and the Yarragadee aquifer approximately along the Brand Highway to the east of the Project.

Groundwater flow is mainly in a northerly direction. The recharge intake area is located in the south and east through infiltration from the superficial formations.

2.2.4   Eneabba Formation

The Eneabba Formation comprises sandstone, siltstone and claystone. According to the WRC (1997), the Eneabba Formation is a multilayered confined aquifer hosting regional groundwater-flow systems. Regional groundwater flows are to the northwest. Recharge to this formation is from infiltration in elevated areas where it outcrops or subcrops beneath lateritic and sandy soils. Discharge from the Eneabba Formation is to the superficial formations to the west along the Beagle Fault.

3.1   Methodology

Groundwater quality data for the Project area were sourced from previous drilling and monitoring reports, (Rockwater, 2002; URS, 2005; Rockwater, 2009). The samples were separated, based on the bore depths to represent the superficial (superficial formations and CCM) and deep (CCM and Yarragadee) aquifers. The shallow aquifer samples include those from the watertable zone and shallow test production bores and monitoring bores with a depth less than 30 m.

The mine waste disposal process will be a progressive backfilling of the mined void using waste rock and coal combustion ash generated from the Coolimba Power Station. This will enable rehabilitation activities progressively to restore the land surface to its original contours as the mine progresses. The overburden rock backfilling will take place over 29 years during the mine operation phase. The final void in Year 30 will not be backfilled.

Interaction between the backfill (overburden, salt and ash), groundwater and rainfall recharge has potential to leach salt and metals that may influence the local groundwater quality. A series of analyses were carried out by Terrenus Earth Sciences (2008) to evaluate the geochemical characteristics of the mine overburden. The geochemical analyses include the following tests:

- Acid-based tests.

- Multi-elements in solids.

- Cation exchange capacity and sodicity.

- Multi-elements in water extracts.

- pH and alkalinity.

- Salinity measurements.

3.2   Baseline Groundwater Hydrochemistry

Salinity, as Total Dissolved Solids (TDS), and chloride (Cl) concentrations were mapped for the superficial and deep aquifers (Figures 3.1 to 3.4). Regional groundwater salinity maps (WRC, 1997; URS, 2006) were taken into account when the local groundwater quality maps were produced. The available groundwater quality data are summarised in Table 3.1 and Table 3.2. Table 3.3 summarises the interpreted baseline concentrations.

Trace metals, including Aluminium (Al), Arsenic (As), Chromium (Cr), Copper (Cu), Boron (B), Molybdenum (Mo) and Zinc (Zn), were identified as the key seepage indicators.

Table 3-1   Groundwater Quality Data for the Shallow Aquifer (Projection MGA50)

BoreID	Easting	Northing	Aquifer	TDS (mg/L)	Cl (mg/L)	Mo (mg/L)	B (mg/L)	Cu (mg/L)	Zn (mg/L)	Al (mg/L)	As (mg/L)	Cr (mg/L)
EM14	334679	6688699	Superficial	400	100	-	0.15			0.5	-	-
EM15	334420	6689024	Superficial	2325	880	-	0.08			0.6	-	-
EM24	333724	6687760	Superficial	1173	400	-				0.8	-	-
EM27	334424	6693554	Superficial	188		-					-	-
EM28A	332299	6690160	Superficial	488		-					-	-
EM29A	333144	6690876	Superficial	968	460	-	0.02			0.5	-	-
EM30A	331632	6689211	Superficial	739		-					-	-
EM31	334000	6690665	Superficial	3773	2600	-	0.03			0.2	-	-
EM32	333800	6690476	Superficial	4750	570	-				0.2	-	-
EM33	333573	6690285	Superficial	973	430	-				<0.2	-	-
EM34	333565	6690543	Superficial	813	360	-	0.01			0.2	-	-
EM36	333504	6691008	Superficial	1575	1000	-	0.02			0.3	-	-
EM37	333735	6691159	Superficial	638	250	-				0.3	-	-
EM38	334003	6691042	Superficial	1433	820	-				0.2	-	-
EM39	334028	6690861	Superficial	1192		-					-	-
EM45	334420	6689335	Superficial	3375	2100	-	0.08			0.2	-	-
EM46	334398	6687361	Superficial	1583	260	-	0.01			1	-	-
EM47	333928	6687567	Superficial	900	320	-	0.01			0.4	-	-
EM48	334064	6687887	Superficial	3750	440	-	0.35			0.2	-	-
CW010P	328938	6687916	Superficial	474	237	-	0.068	0.004	0.009	0.04	-	-
CW014P	328981	6687507	Superficial	1840	962	-	0.175	0.01	0.065	0.11	-	-
EWM10	326109	6689844	Superficial	8920		-					-	-
EWM11	325607	6689832	Superficial	9183		-					-	-
EWM12	325888	6692828	Superficial	1792		-					-	-
EWM13	326389	6692837	Superficial	1567		-					-	-
EWM14	326880	6692845	Superficial	1527		-					-	-
EWM15	327391	6692763	Superficial	1290		-					-	-
EWM16	327689	6692778	Superficial	2043		-					-	-
EWM17	328263	6692795	Superficial	1810		-					-	-
EWM18A	329096	6692879	Superficial	176		-					-	-
EWM19	329397	6692911	Superficial	689		-					-	-
EWM2	329905	6689880	Superficial	427		-					-	-
EWM20	329550	6692943	Superficial	503		-					-	-
EWM21	330392	6693023	Superficial	213		-					-	-
EWM23	330814	6690270	Superficial	712		-					-	-
EWM25	331268	6690471	Superficial	549		-					-	-
EWM3	329404	6689875	Superficial	3960		-					-	-
EWM4	328902	6689875	Superficial	2159		-					-	-
EWM5	328367	6689852	Superficial	1942		-					-	-
EWM7	326898	6689857	Superficial	6270		-					-	-
EWM8	326713	6689854	Superficial	5575		-					-	-
EWM9	326503	6689850	Superficial	3213		-					-	-
LS10A	311400	6661700	Tamala Limestone Superficial	3970	2070	-	0.05				-	-
LS12A	304900	6681500	Tamala Limestone Superficial	33900	18900	-	4.2				-	-
LS15B	304700	6702200	Tamala Limestone Superficial	33400	18500	-	4.1				-	-
LS5B	316400	6635000	Tamala Limestone Superficial	800	326	-	0.06				-	-
LS1B	320900	6625250	Tamala Limestone/Lesueur Sandstone Superficial	1240	580	-	0.1				-	-

Table 3-2   Groundwater Quality Data for the Deep Aquifer (Projection MGA50)

BoreID	Easting	Northing	Aquifer	TDS (mg/L)	Cl (mg/L)	Mo (mg/L)	B (mg/L)	Cu (mg/L)	Zn (mg/L)	Al (mg/L)	As (mg/L)	Cr (mg/L)
LS16A	316700	6702000	Cadda Formation	1310	554		0.58					
LS4B	337000	6625800	Cadda Formation Superficial	3690	139		<0.02					
CW011P	328938	6687916	Cattamarra CM	4740	2680	0.877		0.001	0.14	<0.01		
CW013P	328621	6687804	Cattamarra CM	9260	5000	0.339		0.004	0.015	<0.01		
CW015P	328981	6687507	Cattamarra CM	5040	2870	0.594		0.005	0.022	0.03		
CW035P	328221	6688637	Cattamarra CM	3300	1900	0.6		0.01	0.07	0.36		
CW036P	328224	6688637	Cattamarra CM	6100	3600	2.2		0.02	0.18	0.59		
CW038P	330290	6689217	Cattamarra CM	870	410	0.94		0.07	0.33	4.4		
CW039P	330284	6689215	Cattamarra CM	1100	420	0.14		0.02	5.6	1.4		
CW040P	328819	6687129	Cattamarra CM	3200	1800	0.49		0.02	0.1	0.44		
CW042P	330023	6687363	Cattamarra CM	3600	2000	0.45		0.02	0.11	0.49		
CW043P	330024	6687361	Cattamarra CM	2900	1700	0.36		0.06	0.28	3		
CW044P	328541	6684139	Cattamarra CM	3500	1400	0.42		0.03	0.08	1.7		
CW045P	328541	6684141	Cattamarra CM	1200	740	0.31		0.02	0.08	1.7		
CW047P	327375	6686972	Cattamarra CM	2600	1500	0.16		0.02	0.05	1.3		
CW068PB	330354	6685902	Cattamarra CM	1700	800							
CW069PB	329459	6685655	Cattamarra CM	2500	1400	0.43	0.3			<0.02	<0.001	<0.005
CW070PB	330461	6684591	Cattamarra CM	2800	1700	1.1						
CW071P	331259	6683940	Cattamarra CM	1157								
CW072PB	330828	6684326	Cattamarra CM	1900	1000	0.34	<0.2			<0.02	<0.001	<0.005
EWP21	329103	6691689	Cattamarra CM	1575	825			0.002	0.01	0.05	<0.002	
EWP24	329163	6690191	Cattamarra CM	1551	810							
EWP25	329203	6689192	Cattamarra CM	1757	710							
EWP27	330396	6691738	Cattamarra CM	1560	900							
LS14A	327200	6682300	Cattamarra CM	1040	534		0.07					
LS3A	331100	6625700	Cattamarra CM	1500	740		0.1					
LS4A	337000	6625800	Cattamarra CM	1600	800		0.1					
LS8A	326200	6647200	Cattamarra CM	5630	2900		0.3					
CW041P	328822	6687129	Cattamarra CM ??	2800	1700	0.81		0.02	0.14	1.6		
LS14B	327200	6682300	Cattamarra CM	540	217		0.06					
LS3B	331100	6625700	Cattamarra CM	8270	4600		0.12					
LS13A	318000	6681500	Eneabba Formation	1360	695		0.16					
LS15A	304700	6702200	Eneabba Formation	26600	14800		3.3					
LS7A	330100	6635900	Eneabba Formation	1010	480		0.03					
LS13B	318000	6681500	Eneabba Formation Superficial	2900	1560		0.25					

Table 3-3   Summary of Interpreted Baseline Concentrations for Selected Analytes

Parameters	Concentration (mg/L)
Salinity (TDS)	1575
Chloride	825
Aluminium	0.59
Arsenic	<0.001
Boron	0.3
Chromium	<0.005
Copper	0.02
Molybdenum	2.2
Zinc	0.18

3.2.1   Groundwater Salinity, Chloride Concentration and pH

The interpreted groundwater salinity (TDS) map of the superficial aquifer (Figure 3.1) shows that the local groundwater salinity changes generally from fresh/marginal (TDS from 1,000 to 2,000 mg/L) in the east to brackish (over 5,000 mg/L) towards the west.   Comparatively high salinity values were recorded in shallow bores in areas where the shallow watertable responds to evaporation processes. The map also shows a comparatively high salinity zone extending from the west of the mine area to the coast. This zone (TDS > 2,000 mg/L) may be correlated with brackish groundwater leakage from the Eneabba Formation and the CCM, or with low-lying, shallow water table catchment areas where evaporation processes concentrate salts.

As the groundwater is primarily of the sodium chloride type, the chloride concentration distribution (Figure 3.2) follows a similar pattern to that for the salinity. The chloride concentration is generally below 500 mg/L to the east of the mine area and increases to above 4,000 mg/L to the west.

The interpreted distribution of groundwater salinity in the deep aquifer (CCM and Yarragadee aquifers, Figure 3.3) shows TDS concentrations generally below 1,000 mg/L in areas to the east, north and south of the Project. A high salinity zone (TDS > 2,000 mg/L) is located to the west of the Project and extends north-westerly toward the coast.

Chloride concentration distributions in the deep aquifer (Figure 3.4) follow a similar pattern as that for the salinity. The concentration is generally below 500 mg/L to the east, north and south of the Project area. Higher chloride concentrations (above 5,000 mg/L) occur to the west of the Project area.

Groundwater pH values range from 6 to 8.

3.2.2   Trace Metal Concentrations in Groundwater

Available data for the trace metal (Mo, Cu, B, Al, Zn, As and Cr) concentrations in groundwater are provided in Tables 3.1 and 3.2 and presented in Figures 3.5 to Figure 3.9.

Molybdenum concentrations from the deep aquifer ranged from 0.1 to 7.6 mg/L, with an average of 0.5 mg/L (Figure 3.5). No data for the molybdenum were available from superficial aquifer.

Data for copper were available from two shallow aquifer bores (Figure 3.6). The concentration values were 0.01 and 0.004 mg/L. Copper concentrations from the deep aquifer bores ranged from 0.002 to 0.07 mg/L.

Data for the boron are provided in Figure 3.7. Concentration values for the superficial aquifer bores ranged from 0.01 to 0.35 mg/L. The boron values in the superficial aquifer in the Project area ranged from 0.07 to 0.18 mg/L with an average of 0.13 mg/L. Boron concentrations were available from deep aquifer bores with values from 0.01 and 0.3 mg/L.

Data for the aluminium are provided in Figure 3.8. Concentration values for the superficial aquifer bores ranged from 0.2 to 1.6 mg/L, with an average of 0.4 mg/L. Comparatively high aluminium concentration values were found in the Project area. Aluminium concentrations for the deep aquifer bores ranged from 0.05 to 4.4 mg/L, with an average of 1.6 mg/L. Aluminium concentration values were generally higher in the deep aquifer than the superficial aquifer.

Data for the zinc are provided in Figure 3.9. Concentration values for the superficial aquifer bores ranged from 0.01 to 0.07 mg/L. Zinc concentrations for the deep aquifer bores ranged from 0.01 to 0.28 mg/L, with the exception of a high value of 5.6 mg/L west of the Project site.

Data for arsenic and chromium concentrations were sporadic. Arsenic and Chromium concentrations for the deep aquifer were inferior to 0.001 and 0.005 mg/L, respectively. No data for these metals were available for the superficial aquifer in the Project area.

The available data show that the soluble salt and metal concentrations are within the ANZECC guideline values for marine water quality, but may exceed the 90 and 95% trigger values for freshwater aquatic ecosystem protection (2000a) and also livestock drinking water guideline values (2000b). The metals of concern are Al, As, B, Cr, Cu, Mo and Zn.

3.3 Mine Overburden Geochemistry and Leachate Hydrochemistry

The test results from and interpretations by Terrenus Earth Sciences (2008) are summarised in Table 3.4 and discussed in the sections below.

3.3.1 Geochemical Characteristics of the Overburden Rocks

- Overburden rocks generated by the proposed Central West Coal Project is likely to be relatively benign and is expected to generate pH-neutral (~pH 6.5 to 7) runoff and seepage of low-to-moderately salinity following surface exposure.

- Over half of the overburden rocks are expected to have very low (<0.1%) total sulphur contents and can be classified as barren. Ninety seven percent of the overburden rocks sampled had total sulphur contents below 0.5%.

- The risk of acid generation from the overburden rock backfill is expected to be low given the general lack of oxidisable sulphur content. About 82% of the overburden rock samples are classified as NAF (Non-Acid Forming), with a further 8% classified as UC-NAF (Uncertain-Potentially Non-Acid Forming). The remaining 10% (approximately) are classified as PAF (Potential Acid Forming), PAF-LC (Potential Acid Forming-Local Capacity) or UC-PAF (Uncertain Category-Potential Acid Forming).

- Given the generally low sulphate–sulphur content of the samples, total sulphur can be used as a simple, quick and cost-effective method for screening the acid forming nature of backfilled overburden rocks.

- Leaching tests were performed on 13 overburden rock samples. Eleven samples showed a consistent result for the pH, salinity and metal concentrations. Abnormal results were obtained from samples AvC-20 (with a TDS of 5,080 mg/L) and AvC14 (with metal concentrations of Al, Mn, Cr, Fe, Co, Ni, and Zn above the detection limits). These two samples were considered not representative and not included in the following data analysis. Salinity of leachates from the 11 overburden rock samples ranges from 92 to 604 mg/L. pH value ranges from 5.5 to 7.0. All metal concentrations are below the laboratory detection limit, with the exception of Al and Mn.

- All overburden rock materials tested are strongly sodic, with significant exchangeable cation imbalances, and would likely require soil conditioning to be suitable to use as a cover material or as topsoil/growth layer.

- Nutrient and organic matter concentrations of the overburden rocks were thought to be very low, and they were not tested.

3.3.2 Geochemical Characteristics of Coal Combustion Ash

- Coal combustion ash is expected to generate alkaline and relatively low-salinity runoff/seepage following surface exposure.

- The ash materials have a median total sulphur content of 0.25% (although are highly concentrated in sulphate) which results in a negligible TOS (Total Oxidisable Sulphur) of less than 0.1% for all tested materials.

- On the basis of the negligible TOS, negative NAPP (Net Acid Producing Potential) and high NAGpH, indicate that all of the ash samples are NAF (Non-Acid Forming).

- The multi-element results indicate that the coal combustion ash is expected to have total metal and nutrient concentrations (in solids) well below the applied guideline values.

- Salinity of the leachate from the coal combustion ash ranges from 505 µS/cm to 558 µS/cm, pH value ranges from 7.1 to 8.2. Solute metal concentrations in leachate from the coal combustion ash are higher than from the overburden rocks (Terrenus Earth Science, 2008). Table 3-4 provides metal concentrations from the leaching tests. The metals of highest leachate concentrations are Al, As, B, Cr, Cu, Mo and Zn. Concentrations of other metals are below the detection limits.

- Very low concentrations of soluble uranium and thorium in leachate from the coal combustion ash suggest that radioactivity often associated with coal combustion ash (and coal) is expected to be within the background levels of soil, according to Terrenus Earth Sciences (2008).

- Multi-element tests by Terrenus Earth Science (2008) indicated that nutrient and organic matter concentrations in ash solids are low, and in most cases, below the detection limits. Leaching tests indicated that nutrient and organic matter concentrations in ash leachate are low and below the relevant guidelines.

3.3.3   Evaporative Salts

It is estimated that the evaporation ponds of the Coolimba Power Station will generate 19,829 tonnes of solid salts per year, of which sodium chloride will be the predominant salt (15,365 tonnes per year). The quantity and composition of the evaporation pond salts will depend on the chemical constituents of the power station water supply. Table 3-5 outlines the expected constituents of the evaporative salt. The raw water composition has been calculated based on 8 GL/yr supply from pit dewatering abstractions, with the balance being drawn from the Yarragadee Aquifer. Average data from the CCM successions to be dewatered estimated data for the Yarragadee Aquifer (based on limited available information) has been used.

On the basis of the above data, the calculated quantity of evaporative salts generated from the evaporation ponds is provided in Table 3-6. These data do not account for inputs from any water treatment chemical dosing that may occur in the power station water supply circuits.

Table 3-4 Metal Concentrations in Coal Combustion Ash

Parameters	Detection Limit	Water Quality Guidelines (mg/L)				Fly Ash (ACIRL, 2007)	Fly Ash	Bottom Ash	Mix Ash	Guidelines Comments
	mg/L	Freshwater (95%)	Freshwater (90%)	Livestock Drinking Water	Marine Water (90%)	mg/L	mg/L	mg/L	mg/L	
Aluminium	0.01	0.055	0.08	5	-	-	0.62	0.15	0.35	90%&95% all samples
Arsenic	0.001	0.013	0.042	0.5	-	0.09	0.066	0.006	0.055	90%&95% fly and mix ash
Boron	0.1	0.37	0.68	5	-	<0.01	1.9	0.2	1.6	90%&95% fly and mix ash
Chromium	0.001	0.001	0.006	1	0.0486	0.025	0.029	<0.001	0.02	90%&95% fly and mix ash
Copper	0.001	0.0014	0.0018	0.5	0.003	<0.01	0.003	<0.001	<0.001	90%&95% fly ash
Molybdenum	0.001	-	-	0.15/0.01	0.023	-	0.11	0.005	0.082	NEPC livestock drinking water in fly and mix ash
Zinc	0.005	0.006	0.015	20	-	<0.01	0.082	0.027	<0.005	90%&95% fly and bottom ash
Notes: Data Source: Terrenus Earth Sciences, 2008										
Cells in pink contain the highest detected dissolved metal concentrations										

Table 3-5 Coolimba Power Station Evaporation Pond Salt Composition

Parameter	Value
Net plant output (2 units) 404.4 MW	404.4 MW
Net plant heat rate (HHV) 9917 kJ/kWh	9917 kJ/kWh
Power plant capacity factor 98 %	98%
Raw Water Composition*	Calcium as Ca – 11.76 mg/L Magnesium as Mg – 63.30 mg/L Sodium as Na – 692.81 mg/L Potassium as K – 21.97 mg/L Iron as Fe – 7.58 mg/L Manganese as Mn – 0.51 mg/L Aluminium as Al – 1.30 mg/L Copper as Cu – 0.02 mg/L Boron as B – 0.22 mg/L Chloride as Cl – 1,246.55 mg/L Sulfate as SO_4 – 154.15 mg/L M alkalinity as $CaCO_3$ – 42.44 mg/L Phosphate as PO_4 – 0.07 mg/L Silica as SiO_2 – 53.24 mg/L Nitrate as NO_3 – 1.02 mg/L Fluoride as F – 0.22 mg/L TDS – 2,275.75 mg/L

Table 3-6 Coolimba Power Station Evaporation Pond Salt Quantity

and Composition

Constituent Salt	Annual Weight (tpa)
Calcium as Ca	103
Magnesium as Mg	555
Sodium Chloride as NaCl	15,438
Chloride as Cl	1,561
Potassium as K	193
Iron as $4Fe(OH)_3$	81
Manganese as $2MnO_2$	5.2
Aluminium as Al	11
Copper as Cu	0.2
Boron as B	2
Sulphate as SO_4	1,401
Phosphate as PO_4	1
Silica as SiO_2	467
Nitrate as NO_3	9
Fluoride as F	2
Total	**19,829**

4.1   Methodology

Solute transport modelling in this section only takes into account the waste rock (overburden) and coal combustion ash. The evaporative salt disposal modelling will be discussed in Section 5.

The groundwater flow model for this Project was adapted from one developed by Rockwater (2009). The movement of solute was predicted using a modified groundwater flow model and a non-reactive solute within a transport model. The groundwater flow model developed by Rockwater (2009) was converted and modified using VISUAL MODFLOW version 4.3 and MODFLOW 2000 and MT3DMS.

The Rockwater model was also refined to provide the required groundwater flow simulations for the mining operation, and final void before and after closure. The model covers an area of 26 by 38 km and comprises irregular grid sizes ranging from 250 to 500 m. The model had a total of 8 layers representing the confined aquifers. The superficial formations were incorporated into the uppermost layer of the confined aquifer succession. The 30 years of pit dewatering was modelled using the MODFLOW Drain Package, which simulates the interaction between the drains and groundwater, and calculates the water balance based on the drain base elevation, conductance and groundwater level in the local aquifer.

Model initial and boundary conditions were established based on the available measurements and maps by Kern (1997), Rockwater (2009) and URS (2007). Constant head boundaries (with the linear gradient option enabled) were set along the model boundaries. The model initial condition, that represents the pre-mining groundwater flow condition, was established with the model operating in steady-state mode and visual calibration against the interpreted groundwater level contour map provided by Rockwater (2009).

The solute transport model simulated the 3-dimensional movement of plumes of non-reactive solute to represent that which may be generated from in-pit co-disposal of ash and overburden rocks. The solute transport model incorporates hydrodynamic transport of solutes such as metals by simulating recharge infiltration, groundwater throughflow, and dispersion based on a pre-defined concentration gradient between the source area (pit) and local water table. The model delineated the potential concentrations of leachates within the model domain and timeframe based on the mining and dewatering schedules. The predictive result is then applied to the various analytes to provide an estimate of their concentrations based on the leachate concentrations provided in Table 3-4.

4.2   Conceptual Model

It is assumed that the disposal process has the potential to release dissolved salts and metals from the overburden rocks, evaporative salt generated from the evaporative ponds and coal combustion ash. It is expected that these dissolved components will be mobilised from interactions between rainfall recharge and groundwater throughflow. The leachate will flow from the backfilled areas to the mine (during operations) and final void (after operations). The concentration of a given solute will depend on the rates of rainfall infiltration, prevailing oxidisation environments, availability and exposure to incident waters, release rate, and groundwater throughflow rate. The location of the solute within the model will depend on the hydrodynamic variables listed in Section 4.1 and the time since its release and/or capture.

To estimate the dilution and dispersion behaviours of a number of analytes, the model assumes a nominal concentration of 100 mg/L of a non-reactive solute. In reality, many analytes are reactive and their concentration might decrease down-gradient due to physico-chemical processes that may promote adsorption and retardation of part of the plume. The rate of decay of solute release has been defined by the bulk leaching test work in similar settings in Collie, Western Australia. As indicated in Figure 4.2, the rate of decay in this setting is expected to result in the release of all soluble components within seven years. This approach assumes that there will be a decrease in direct exposure of the overburden backfill materials to rainfall infiltration as they are buried and consolidated.

The relationship between the modelled solute and each analyte is defined by the initial concentration determined during the geochemical testing as listed in Table 4.1 (after Terrenus Earth Science, 2008).

4.2.1   Conceptual Model Design for the 30-Year Mining Period

Key components of the solute transport model during the operational (mining) phase include:

- The framing of annual pit developments, including mining and backfilling of overburden rock materials and coal combustion ash is based on the 30-year mining schedule (Figure 4.1). As the transient model progresses the backfilled areas subject to recharge progressively increase.

- The backfilling of the mine pit starts in the second year and end in the 29th year of the mining operations. The modelling covers the whole mining operation period.

- A recharge rate of 3.78 mm/yr (sourced from the Rockwater Model, 2009) was applied to the model domain from the second year until the end of the simulations (to the 30th year, Figure 4.2).

- The model only represents solute transport in saturated aquifers, primarily the CCM. The solute concentration is applied progressively to the upper most water table in the backfill areas, also within the CCM.

- Simulations include non-reactive and non-absorptive solute transport. The result represents a worst case scenario.
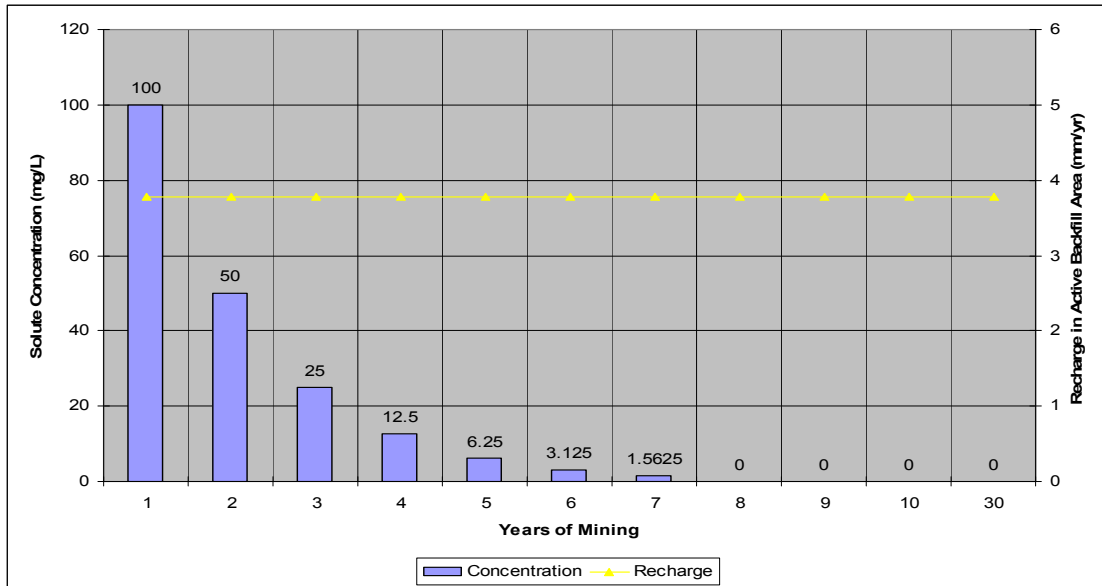
Figure 4.2 Recharge and Solute Concentration Decay Graphs

Table 4-1   Initial Metal Concentrations Used in the Model

Parameters	Concentration (mg/L)
Aluminium	0.62
Arsenic	0.09
Boron	1.9
Chromium	0.029
Copper	0.003
Molybdenum	0.11
Zinc	0.082

4.2.2 Conceptual Model Design for Mine Closure and Final Void

Key components of the solute transport model during the post operational (mine void) phase include:

- Groundwater level and water quality at the end of 30 years mining operation represents the conditions for the start of post mining simulations.

- The final void would be located at the northern end of the mined area.

- Rainfall recharge to the void is equal to the average annual rainfall (539 mm/yr), and evaporation from the void is 1,950 mm/yr, representing about 80% of the annual pan evaporation for this area. The evaporation from the final void will lead to the establishment of a hydraulic sink and elevated water concentrations of all chemical components that enter it.

- The solute transport model is used to assess water level and quality changes in the final void for 500 years after the mine closure. The selection of 500 years was determined through the model sensitivity analysis.

4.3 Numerical Model Set Up

The following processes and parameters were added to the flow model.

4.3.1 Modification of the Rockwater Flow Model

The following modifications have been made to the original Rockwater flow model based on the conceptual model, model setup and software requirement for solute transport modelling.

Merging of Layers 1 and 2.

This change was necessary to remove the numerical errors and visual artifacts caused by the dry cells in the mined area. Dry cells in the original flow model acted like a solute barrier that cause undesirable numerical instability and distortion of the solute plume. Hydraulic properties in layers 1 and 2 in the original model are the same. The layers are also horizontal with the exception of the layer boundary. Merging the two layers does not change hydraulic conductivity (except along the layer boundary) and does not have a significant effect on the hydraulics represented by the original model.

Removal of inactive cells in the western part of the model

The western part of the model domain (bound by the Peron Fault) comprises the Eneabba Formation. According to regional hydrogeological assessments (WRC, 1997; URS, 2006), the Eneabba Formation is a regional low-yield aquifer and hydraulically connected with the CCM in the east. Inactive cells in the Rockwater model were removed to allow for better simulation of groundwater flow and solute transport at local and regional scales. The Eneabba Formation was represented as a low-transmissivity aquifer in the modified model.

Change of hydraulic conductivity in the backfilled area

Higher hydraulic conductivity was used to represent the backfilled area, as the overburden rock is disturbed and not fully consolidated in the early years of backfilling.

4.3.2   Model set up for the 30-Year Mining Period

1) Recharge – A net recharge rate of 3.78 mm/yr was applied to the model domain throughout the modelling period.

2) MODFLOW Drain Package was used to simulate pit dewatering during the mining period.

3) Solute concentrations (100 mg/L) were assigned to the backfilled areas based on the decay graph using the Boundary Condition – Recharge-Concentration option.

4) Initial baseline concentration is 0.0001 mg/L. This small value was chosen to allow the model to predict the changes arising from the applied solute without background effects.

5) Longitudinal dispersivity is 10 m, and transverse dispersivity is 1 m.   This was based on the recommended values by Spitz and Moreno (1996) for similar geological materials and settings.

6) MODFLOW MT3DMS was used to simulate the non-reactive and non-absorptive solute transport. The result represents a worst case scenario that excludes retardation effects.

7) The recharge containing the nominal solute concentrations was applied to the uppermost active cell (water table).

8) Metal concentrations provided in Table 4.1 were used for the individual transport assessments using a discrete factor derived from the analyte concentration divided by the initial solute concentration.

9) The model was run for 30 years with yearly time steps.

10) The Warradarge Fault was represented with a horizontal barrier.

4.3.3   Model Set Up for the Mine Closure and Final Void

1) Recharge – A recharge rate of 3.78 mm/yr was applied to the model domain with the exception of the final mine void, where a recharge rate of 539 mm/yr equal to the full average annual rainfall was applied to the open water body.

2) Evaporation – the rate from the final void of 1,950 mm/yr was used representing the full average annual total.

3) No additional solute sources were included in the model.

4) The model was run 500 years to predict long term water level and water quality trend.

5) The same boundary conditions used in the operational model were applied to this model.

4.4   Modelling Results

4.4.1   The 30-Year (Operational) Mining Period

Predicted Non-Reactive Solute Transport Plume

The predictive solute transport modelling result for the 100 mg/L non-reactive solute is provided in Figures 4.3 to 4.9. The results show that.

- Elevated solute concentrations occurred in the uppermost model layer representing the superficial formations and CCM superficial aquifer.

- The highest concentrations follow the recently backfilled area behind the active pit floor. The concentration plume generally remained in the mined area as it is captured by the drawdown cone from pit dewatering. Dispersion effects to areas outside the pit were insignificant. Higher hydraulic conductivities in the backfilled mine area will promote this capture process. The capture zone effectively forms a hydraulic barrier to solute dispersion outside the immediate vicinity of the pit.

Predicted Non-Reactive Trace Metal Transport Plumes

The predicted metal transport plumes are provided in Figures 4.19 to 4.25. The raw results show that:

- The transport plumes form a similar distribution pattern to the non-reactive solute.

- As with the non-reactive solute, the highest concentrations also follow the recently backfilled areas with pit dewatering effectively capturing the plumes.

4.4.2　Model Results for the Mine Closure and Final Void

The predicted solute and metal concentration plumes are provided in Figures 4.10 to 4.18. The results show that:

- Water levels in the final void and groundwater levels in the local aquifers recover after mine closure (Figures 4.28 to 4.30). Due to high evaporation rate from the open water surface, the final mined void would form a groundwater sink, with the pit lake water level about 4 m below the baseline water table elevation (Figure 4.27) and creating a hydraulic capture zone.

- Drawdown impacts of the final void on the surrounding groundwater level will be limited to the area close to the void (Figures 4.29 and 4.30) and is not expected to have a large impact on the regional groundwater flow. At 500 years after the mine closure, residual drawdowns are predicted to extend up to 3 km from the final void. The capture zone is predicted to be elongated along the direction of regional flow.

- The residual drawdown cone will stabilise about 60 years after mine closure (Figure 4.27). At the same time, water in the void would also be saline (about 18,000 mg/L TDS) due to evaporation losses. Predicted soluble metal and salt concentrations in the final void will continue to increase over a long period of time (more than 500 years).

- The highest soluble metal concentrations occur within the final void. Steep concentration gradients occur immediately outside the final void (Figure 4.26).

- South of the void, outside the final void capture zone within the mined area, a second area of residual solute (derived only from the backfill waste) is present. This solute slowly dispersed into the local groundwater close to the pit, with an overall westerly movement down gradient along the regional flow path within the CCM.

Predicted concentration distribution maps for salinity and soluble metals for 100 and 500 years after mine closure are presented in Figures 4.31 to 4.48. Figures 4.49 to 4.57 show the concentrations in the final void continue to increase for the foreseeable future. The rate of change should decrease to a very small rate about 400 years after mine closure.

The estimated water salinity in the final void is illustrated in Figure 4.49. The pre-mining (baseline) groundwater salinity at the final void site was measured at 1,575 mg/L TDS. The leachate generated from the ash and overburden co-disposal is expected to contribute a salinity of 360 mg/L (Terrenus Earth Sciences, 2008). The long-term model prediction indicates that the water salinity in the final void will increase by evaporation of groundwater throughflow and the capture of the residual plume generated from the waste rock backfill. The model suggests that the salt derived from the co-disposed backfill forms a very small component of the salinity in the final void. This is primarily due to the comparative mass balances of salt derived from the groundwater throughflow and rainfall recharge.

It is predicted that the water in the final void will gradually become hypersaline, with salinity exceeding 100,000 mg/L TDS after about 400 years. However, the influence of the final void on the local groundwater quality is predicted to be limited to the area nearby the void where steep hydraulic gradients should prevail. The salt is expected to be captured by these processes operating in and around the final void.

The likely chloride and soluble metal concentrations in the final void after 500 years of mine closure are illustrated in Figures 4.50 to 4.57. The trend is similar to that for the salinity since the water is predominantly of the sodium chloride type. The pre-mining (baseline) metal concentrations in the groundwater at the final void site are already higher than that what is expected from the leachate at the end of mining operations (Table 4.1). The predicted metal concentrations in the final void after 500 years of mine closure are outlined in Table 4.2. Similar processes to those described for salinity (above) are expected to effectively capture soluble metals derived from both the backfill and natural groundwater.

Table 4-2   Simulated Metals Concentrations in the Final Void 500 Years after Mine Closure

Parameters	Concentration (mg/L)
Aluminium	38.3
Arsenic	0.1
Boron	19.5
Chromium	0.3
Copper	1.3
Molybdenum	142.9
Zinc	11.7
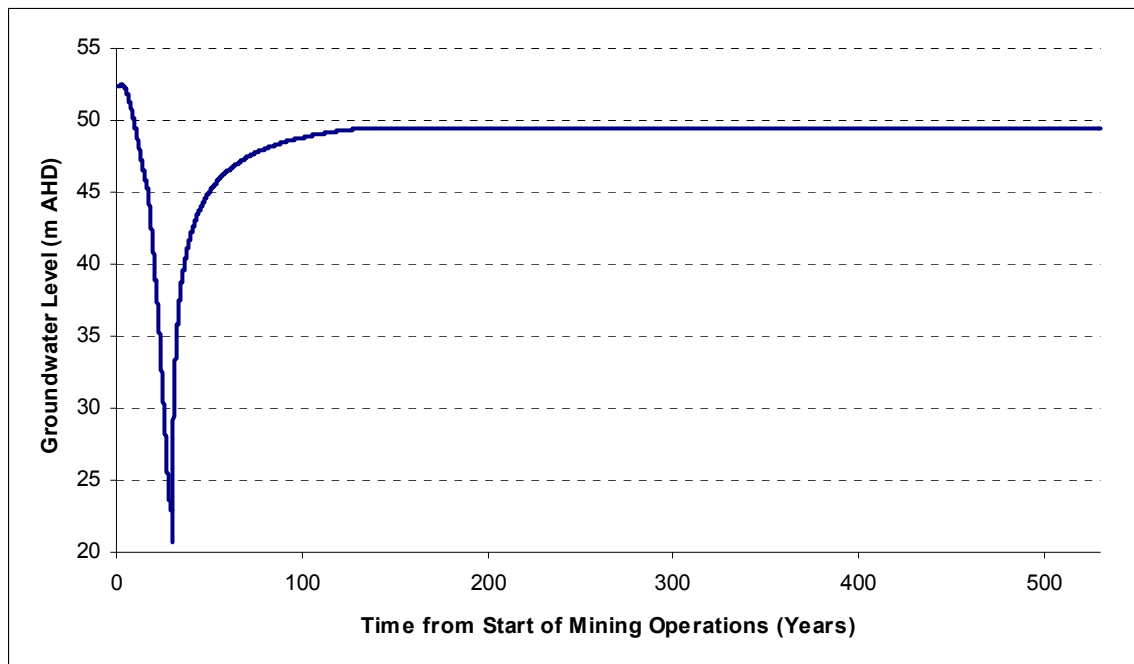
Figure 4.27 Simulated Groundwater Levels in the Final Void
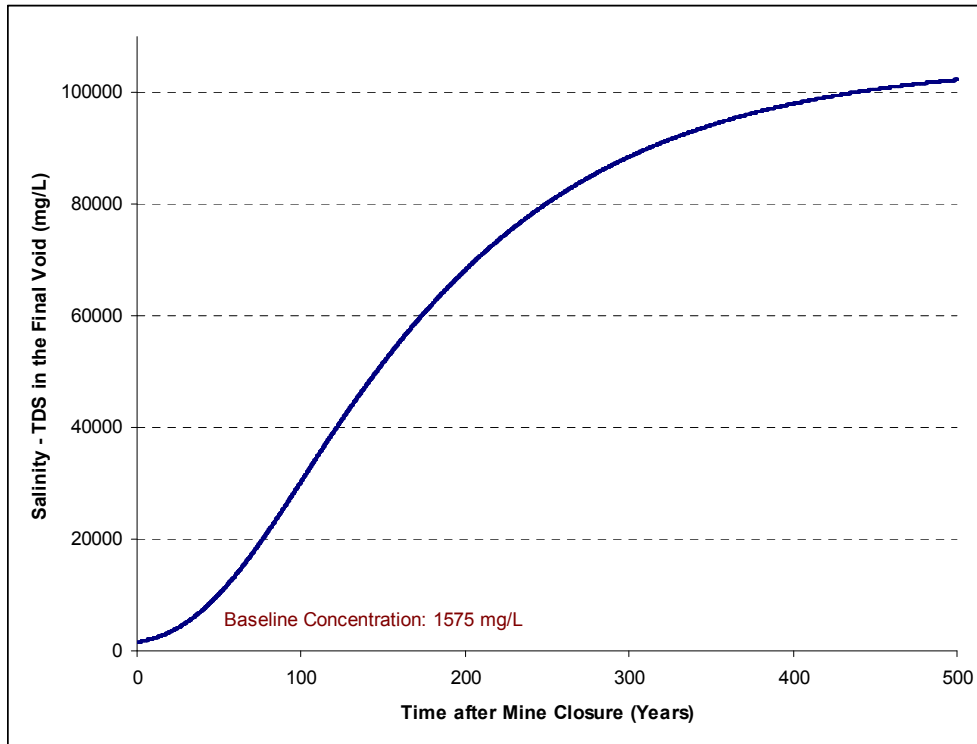
Figure 4.49 Simulated Salinity Concentrations in the Final Void
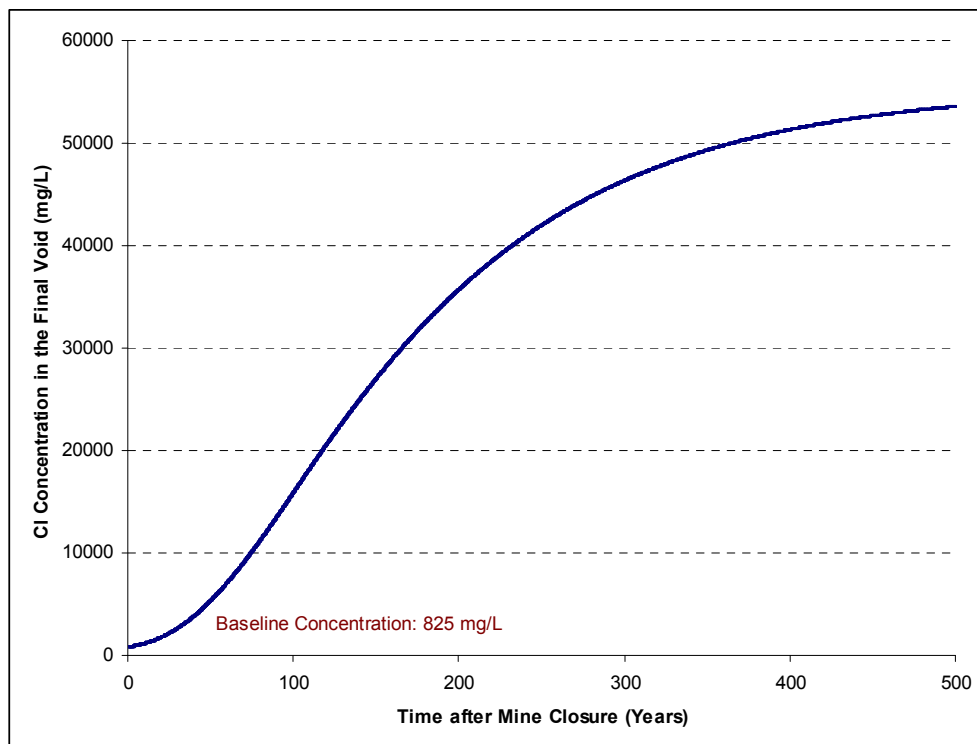
Figure 4.50 Simulated Chloride Concentrations in the Final Void
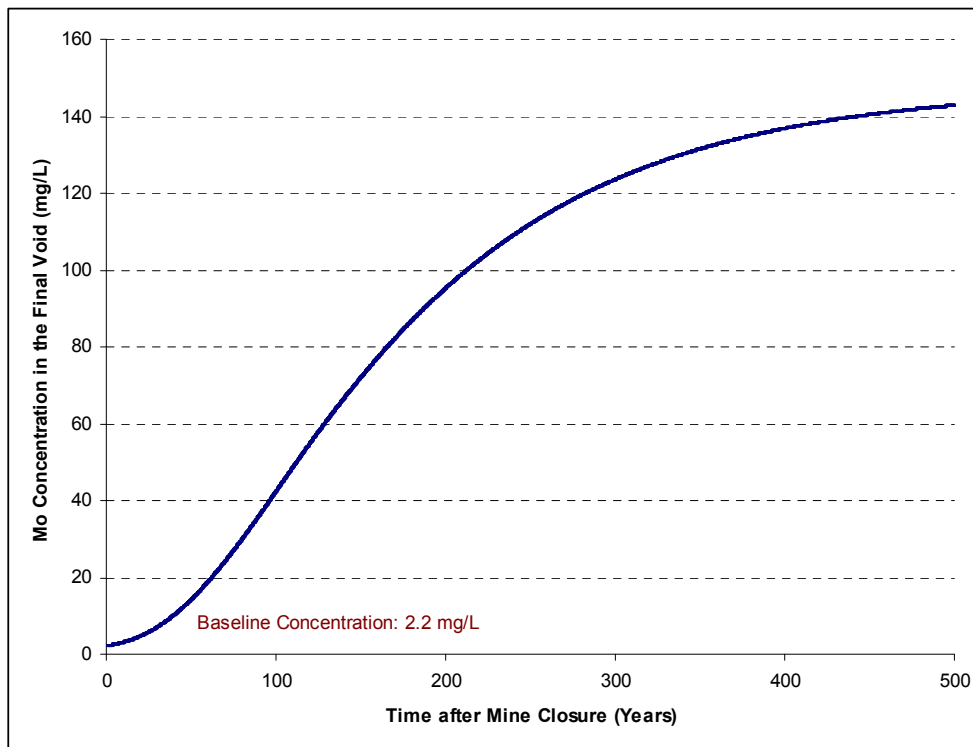
Figure 4.51 Simulated Molybdenum Concentrations in the Final Void

Figure 4.52 Simulated Copper Concentrations in the Final Void

Figure 4.53 Simulated Boron Concentrations in the Final Void

Figure 4.54 Simulated Aluminium Concentrations in the Final Void

Figure 4.55 Simulated Zinc Concentrations in the Final Void

Figure 4.56 Simulated Arsenic Concentrations in the Final Void

Figure 4.57 Simulated Chromium Concentrations in the Final Void

5.1   Methodology for Salt Co-Disposal Modelling

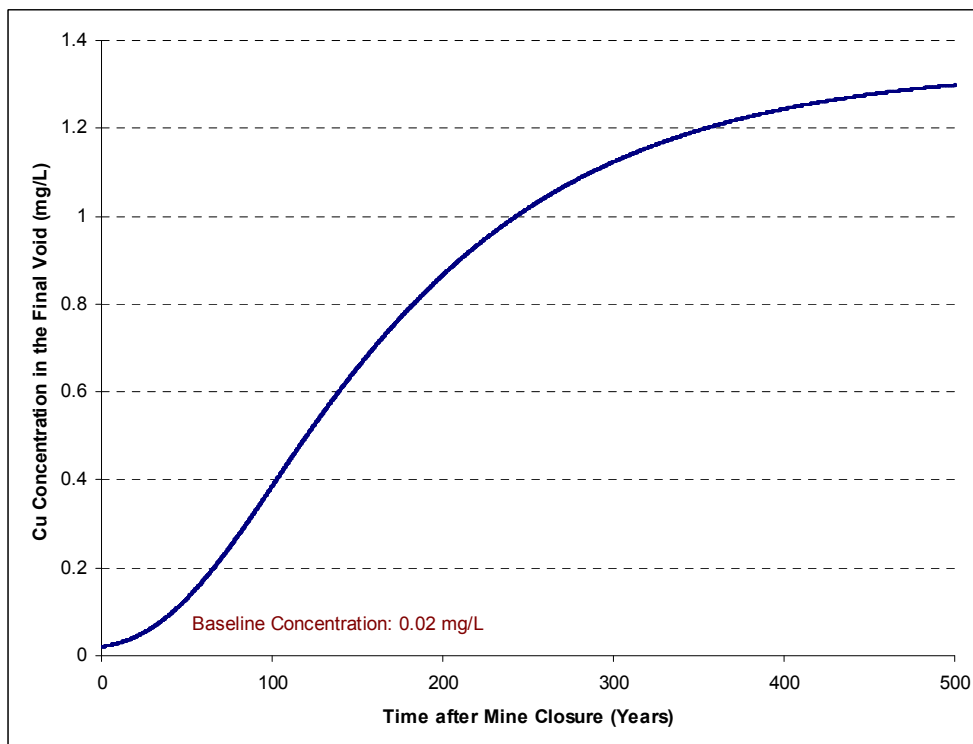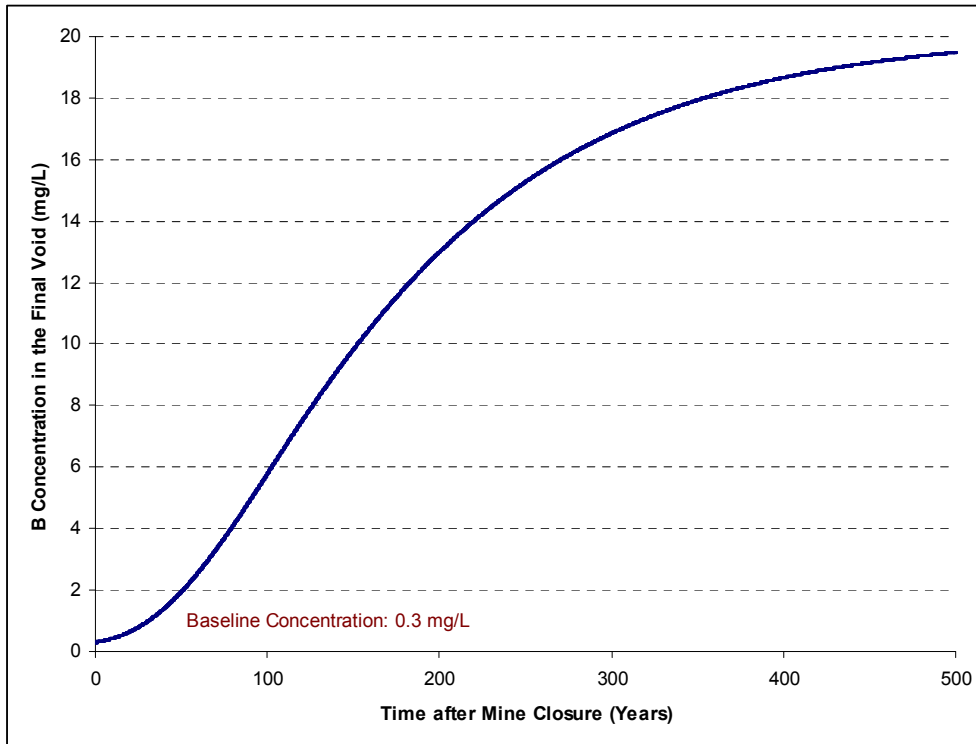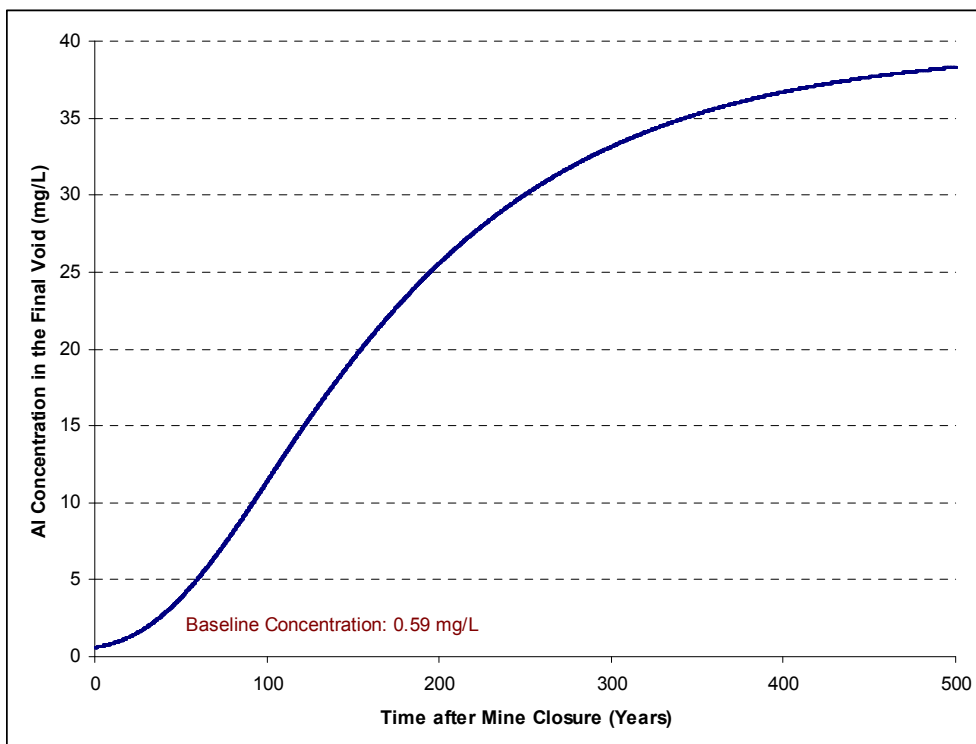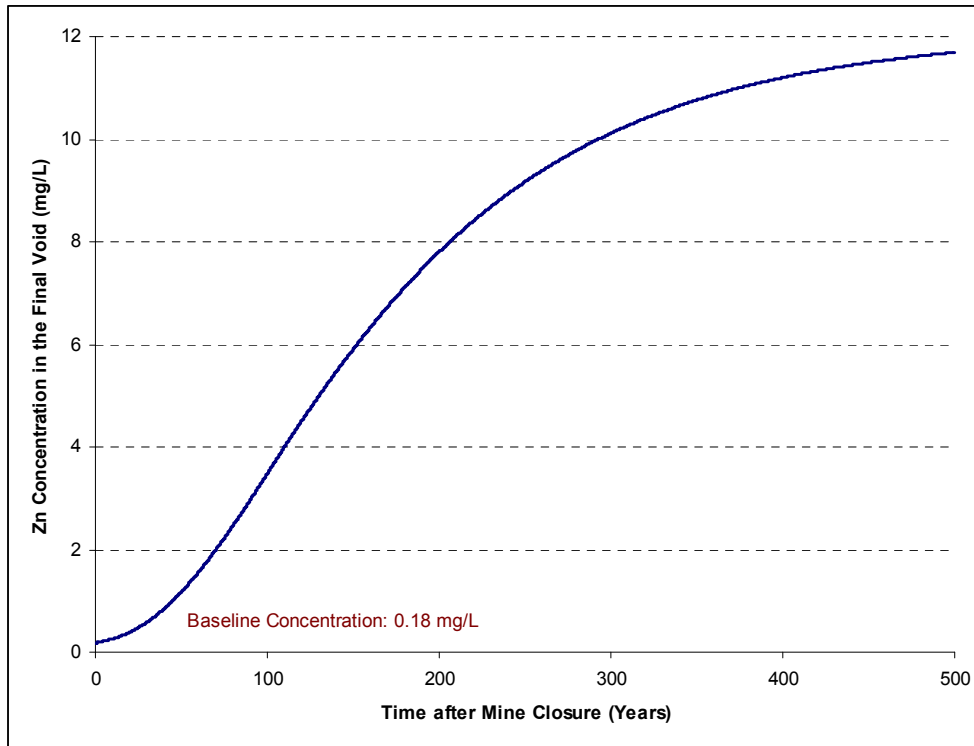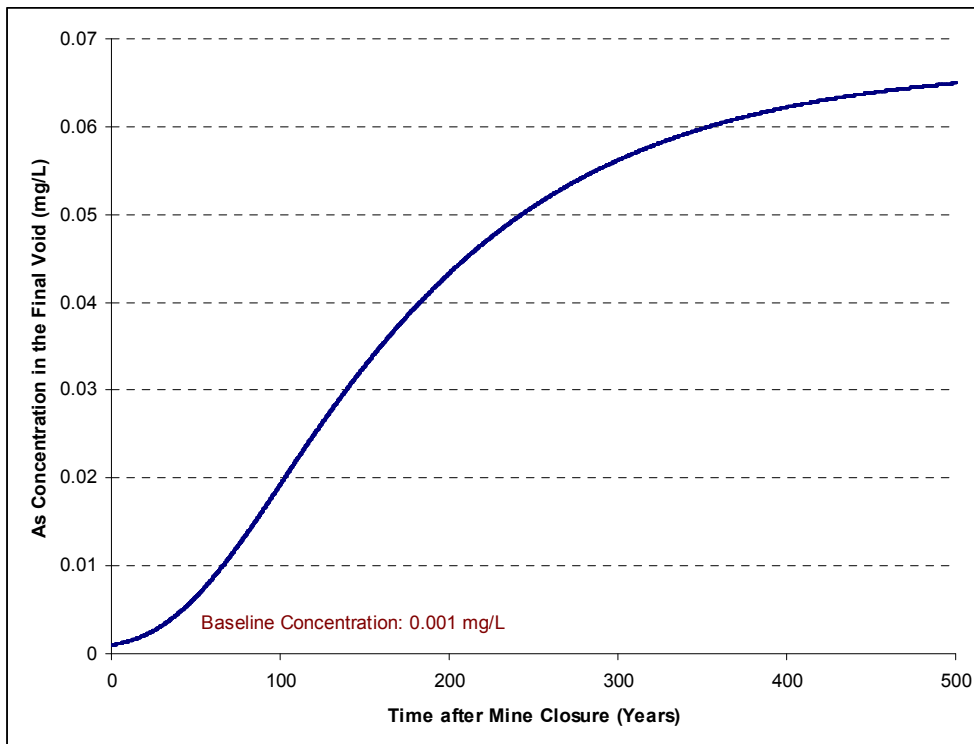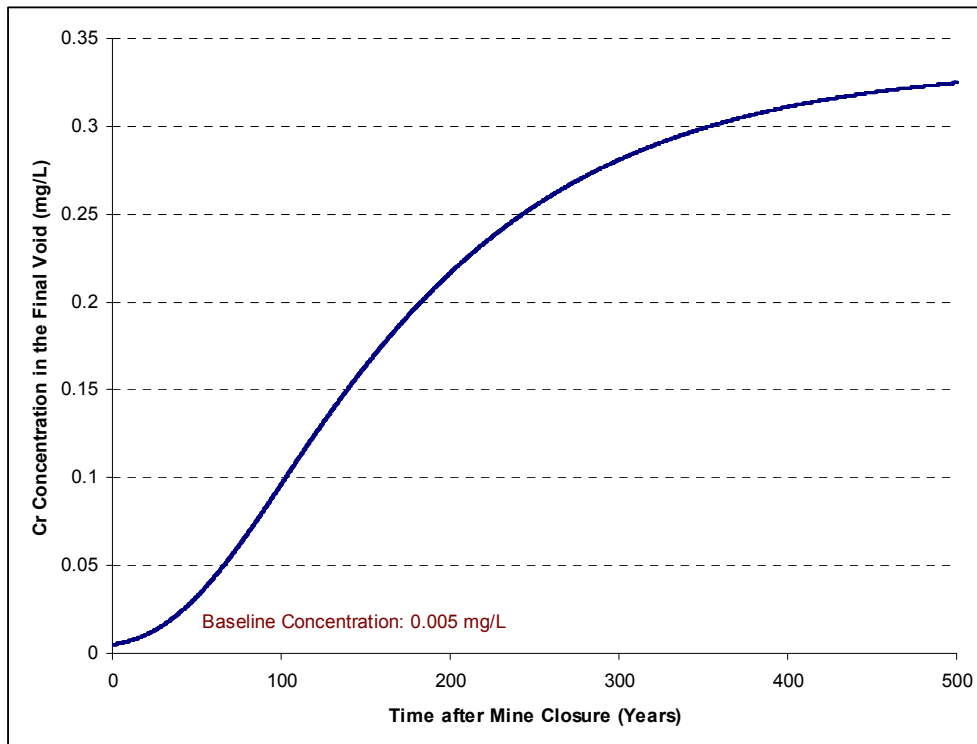It is estimated that the Coolimba Power Station will generate about 19,829 tonnes of salts per year, of which sodium chloride will be the predominant salt (15,365 tonnes). Solute transport modelling of the salt co-disposal takes into account the co-disposal of evaporative salt, the waste rock (overburden) and coal combustion ash.

The solute transport modelling is based on the general information of solubility and leachate forming processes with emphasis on sodium chloride salt. The solute transport simulation does not take into account density-driven effects. Therefore, the model only represents a primitive and simplistic simulation.

The evaporative salt co-disposal will substantially elevate the metals concentrations, likely to above all guideline values. It needs to be recognised that we have not assessed this aspect in the report, due to lack of geochemical data.

5.1.1   Evaporative Salt Disposal

Evaporative salts are proposed to be removed from the evaporation ponds and co-disposed in the advancing mine back-fill faces. Frequency of salt co-disposal is generally expected to occur in four-year campaigns.

5.1.2   Mass Balance

Sodium chloride salt is of high solubility and can enter the groundwater system with the rainfall infiltration and dissolution once co-disposal settings are immersed beneath the water table (as groundwater levels recover after mining). The NaCl solubility curve is shown in Figure 5.1, with a fully saturated concentration of about 320,000 mg/L. Based on a four-year disposal campaign, the co-disposed salt may form a fully saturated concentration boundary, at the salt disposal points, for about 756 years.

5.1.3   Model Set Up

The model set up for the 30-year mining period and for mine closure and final void is similar to the ash co-disposal simulations described in Sections 4.2 and 4.3, with the following additional considerations:

- The simulation is for the mining operation (30 years) where the salt is disposed every 4 years.

- Recharge Concentration Boundary condition is used to represent the saturated salt solution (320,000 mg/L TDS) entering the aquifer with the rainfall recharge (3.78 mm/yr). The 4-year salt disposal schedule is shown in Figure 5.2.

- The baseline groundwater salinity is 2,000 mg/L.

- Decay of concentration is not considered.

- Density driven effects are not considered.

Section 5	Predicted Impacts of Salt Co-Disposal on Groundwater

5-2

5.2 Modelling Results

5.2.1 Model Results for the 30-Year Mining Period

The following results were produced by the 30-year mining dewatering simulation.

- Salinity level in the salt disposal areas is higher than the surrounding groundwater salinity, and increases (Figures 5.3 to 5.4).

- There is a strong dilution effect. The mined area is a local sink due to dewatering and preferential flow. As a result, the local groundwater flows towards the mined area, mixes with the hypersaline leachate and dilutes the hypersaline leachate salinity concentrations.

- Although the simulated leachate concentration is 320,000 mg/L, the loading is comparatively small (about 564 tpa, Table 5-1) due to the low flux or infiltration rate. So the effect is local during the mining period.

- Groundwater flow within the mined area is northerly, aligned to the mine path. The groundwater flow velocity is low as the CCM is of comparatively low transmissivity. This results in low dispersion of the salt (Figure 5.5).

5.2.2 Model Results for Post Mining and Final Void

The following results were produced by the 500-year post mining simulation.

- Salinity concentrations beneath the salt co-disposal areas increase continually (Figures 5.6 to 5.7) and become hypersaline after only a few years. The effect of this hypersaline salinity on the local groundwater flow is poorly defined by the current model. Density driven solute transport modelling would be required to provide a reasonable assessment.

- The plume is predominantly located in the mined area and moves towards the final void.

- The final void would host crystalline salt. The final void has an estimated capacity to contain between 27 Mt to 51 Mt of crystalline salt, based on the salt bulk density (2.165 g/cm^3) and the estimated total volume of the final void. The annual NaCl salt production from the evaporation ponds is 15,438 tonnes/yr), about 0.03% to 0.06% of the final void capability.

- The long-term salinity trend in the final void is illustrated in Figure 5.8, Salinity concentrations increases much more rapidly for the salt co-disposal option than for the ash co-disposal alone.

In summary, the result from the simplistic solute transport simulation of the salt, overburden and ash co-disposal suggests that the salt disposal option may produce significant salt accumulations in the groundwater system in the mined area, in particular in the final void. The system does not reach a steady state condition during the 500-year simulation period.

Predicted Impacts of Salt Co-Disposal on Groundwater

It is noted that this work is based on non-reactive and non-density driven simulations. The parameters used in the current modelling may only provide an order-of-magnitude estimate, due to the lack of geochemical testing data and absence of a density-driven modelling platform.
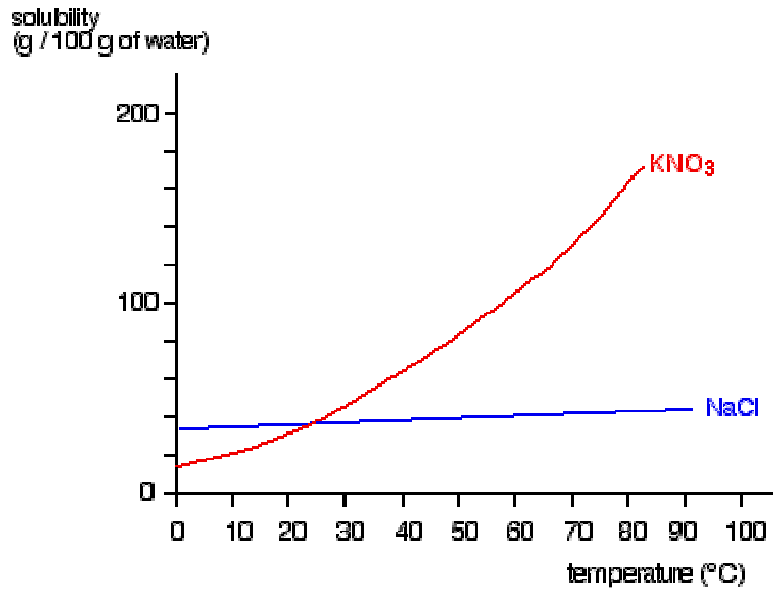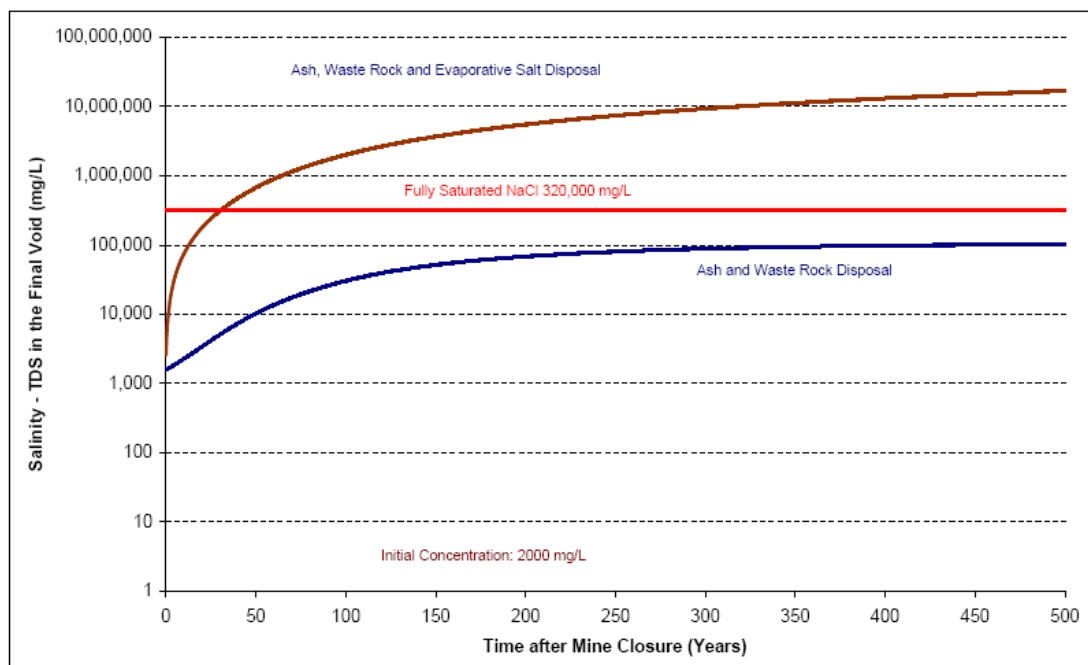

**Figure 5.1 Solubility Diagram for Sodium Chloride
(http://www.chemguide.co.uk/physical/phaseeqia/saltsoln.html)**

Figure 5.8   Predicted Salinity Trend for the Final Void

5.2.3   Intuitive Summary of Model Results

In fundamental terms, the model applied to evaluate the salt is primitive and simplistic. In actual fact, the salt co-disposal issue is complex.

The predictive modelling results, both for ash and salt co-disposal suggest the final void will form a long-term hydraulic sink. In actual fact it may not. Actual groundwater flow, within the aquifer and void will be density-driven, given increased salt loadings. Flow in the void would also be driven by thermal gradients, with turn-over and full mixing of the void waters each year. Similar conditions may dictate groundwater flow along flow paths within the backfill domains.

The modelling shows the final pit lake water levels would be about 4 m below the natural water table elevation. The water column in the final void may be up to 110 m (120 m deep pit, 5 m to pre-mining water table and 5 m drawdown) in depth. Under natural conditions, the water column would be about 115 m in depth through the same profile. In this or similar settings the natural water column (115 m, with SG 1.002) will be equalised by a 110 m water column of SG 1.047. That is, once the salinity plume and final void attain an effective SG 1.047, the local and regional flow systems would be hydraulically balanced. Under this condition the backfill flow paths and final void would tend to become throughflow systems. Groundwater of SG 1.047 reflects salinity of about 50,000 mg/L.

This assessment assumes that the water column has a base at the base of the pit. The assessment may vary if strong density stratification occurs, with the salt plumes being transported vertically downward within the CCM.

In summary, intuitively, the flow paths in the backfill would equalise with the regional water table and promote throughflow. This would probably occur progressively as the backfill is re-wet when the water table locally recovers and the salt-laden water column weights equalise with those in the perimeter areas of the pit. Similar would occur in the final void, with the flow system forming a throughflow system. This may occur in the short-term, during and/or after cessation of mining.

The implication of a throughflow system is that the co-disposed salts would be mobilised and transported downstream beyond the mined environment. It is not apparent at what salinity concentrations the salt plumes would be manifest or where they would be discharged. Presumably once the salt enters the wider throughflow environment, the salt concentrations would be increasingly diluted by mixing with natural groundwater. The degree of mixing will depend on the extent of the lateral and vertical fronts over which the plume is expressed, the uniformity of flow paths and hydrostratigraphy.

Section 5   Predicted Impacts of Salt Co-Disposal on Groundwater

Table 5-1   Evaporative Salt – Disposal Option and Mass Balance

Recharge Rate (mm/yr)	3.78
Estimated Yearly Mined Area (m^2), Based on Total Footprint of Disturbance (1,400 ha)	466,667
Volume of Recharge Entering the Void (m^3/yr)	1,764
Annual Evaporative Salt (NaCl, tonnes/yr)	15,438
NaCl Salt to be Disposed Every 4 Years (tonnes)	61,752
Salt Concentration (mg/L) based on 4-year Disposal Schedule	24,197,492
Saturated NaCl Salt Concentration (mg/L)	320,000
Annual NaCl Salt Entering the Watertable (tonnes/yr)	564
Residual Salt (tonnes, based on 4-Year Disposal Schedule)	61,188
Salt Needed to Maintain a Fully Saturated Concentration (tonnes/yr)	564
Years of Decay to Reach the Concentration Threshold (320,000 mg/L)	109
Density of NaCl Salt (tonnes/m^3)	2.165
Estimated Volume of the Final Void (m^3 x 1,000,000)	12.5 - 23.5
Amount of Salt Needed to Fill Up the Final Void (tonnes)	26,870,586 - 50,516,702
% Annual Salt Production / Final Void Capacity	0.03 % - 0.06 %

Section 6	Conclusion and Recommendation

The assessment of ash co-disposal impacts on groundwater for the Central West Coal and Coolimba Power Station Projects at Eneabba was based on the baseline groundwater quality assessment and solute transport modelling. The study provided short-term (30 years) and long-term (500 years) predictions on groundwater level and water quality changes during the 30-year mining period, and for closure and final void. The study derived the following conclusions.

6.1   Baseline Groundwater Quality

Groundwater quality at the project site varies considerably with location and depth. Local groundwater in both the shallow superficial formations and deep confined aquifers such as the Cattamarra Coal Measures (CCM) are generally fresher in the eastern part of the study area than to the west. Groundwater chemistry maps (including salinity, pH and trace metals) have been produced for groundwater in the superficial and deep confined aquifers. The groundwater salinity in the superficial formations changes generally from fresh/brackish in the east to very saline towards the west. Local hypersaline areas are present where the water table is close to the surface and subjected to strong evapotranspiration processes. These are similar processes to those expected to occur in the final void.

The groundwater salinity in the deep confined aquifers is generally fresh/brackish in areas to the east, north and south of the mine area, and increases to the west and north-west. There is a high salinity zone in both the superficial and deep confined aquifers extending from the west of the mine area to the coast. This high salinity zone may be correlated with the discharge of saline groundwater from the Eneabba Formation and the Cattamarra Coal Measures, possibly coinciding with low-lying catchment areas that have shallow water table with high rates of evapotranspiration. Groundwater pH values in the Project area range from 6 to 8 and are neutral on average.

Baseline concentration values of Aluminium (Al), Copper (Cu), Molybdenum (Mo), and Zinc (Zn) in both the shallow and deep aquifers generally exceeded guideline values applied to various end users including aquatic ecosystems, terrestrial groundwater dependent ecosystems as well as that required for primary production (stock) and recreational purposes. Application of the various guidelines to the natural and modified hydrochemistry is beyond the scope of this study. Applicable guidelines should be considered according to their level of interaction with this Project, taking into account the existing baseline as well as secondary and cumulative impacts from other natural and man-made processes in the region.

6.2   Hydrochemistry of Leachate from the Co-Disposed Waste

Leaching tests found that the salinity of the leachate from the waste rock ranges from 92 mg/L to 604 mg/L, pH from 5.5 to 7.0. The soluble metal concentrations are all below the detection limits.

Salinity of the leachate from the coal combustion ash ranges from 505 µS/cm to 558 µS/cm, pH value ranges from 7.1 to 8.2. Solute metal concentrations in leachate from the coal combustion ash are expected to be higher than from the waste rock (Terrenus Earth Sciences, 2008). The metals of concern are Al, As, B, Cr, Cu, Mo and Zn. Concentrations for other metals are below the detection limits. In summary, salinity and metal concentrations of the leachate from the waste rock and coal combustion ash are much lower than the baseline groundwater. Potential impacts of these metals from the coal ash disposal on the groundwater system were evaluated with a solute transport model.

Multi-element tests by Terrenus Earth Science (2008) indicated that nutrient and organic matter concentrations in ash solids are low, and in most cases, below the detection limits. Leaching tests indicated that nutrient and organic matter concentrations in ash leachate are low and below the relevant guidelines.

6.3   Assessment of Ash Co-Disposal Impacts on Groundwater Quality

Solute transport modelling provided the following results.

6.3.1   The 30-year mining operation period.

Solute concentration plumes occurred in the CCM. These plumes should remain primarily within the CCM due to operational drawdowns (dewatering) and post closure residual drawdowns (close to the final void).

The highest concentration of backfill-derived solute is always located close to the recently backfilled areas. The concentration plume derived from the backfill waste is expected to be retained in the mined area and captured by pit dewatering (during operations) and the final void (after closure). Dispersion effects to areas outside the pit were insignificant. A capture zone centred on the final void will form a hydraulic barrier to solute dispersion to the regional groundwater system. The mined area will probably form a preferential flowpath for groundwater throughflow as the backfill will be more permeable than the surrounding undisturbed CCM. This will tend to guide groundwater and rainfall recharge to the final void that will then be subjected to evapo-concentration processes.

The model also predicted that a large part of residual plume to the south of the mined area will remain within the mined area and may be transported very slowly by the north-westerly groundwater flow. However, the salt and metal concentrations in the residual plume, as well as the groundwater flow rate are expected to be very low. As a result, the change that this residual solute will impart on the local groundwater system is probably also very small.

Section 6   Conclusion and Recommendation

It is estimated that the Power Station will generate about 20,000 tones of salts per year, of which sodium chloride will be the predominant salt. The salts, if dissolved by rainfall infiltration or groundwater, will entre the local groundwater system and cause the groundwater salinity to increase. However, it is thought that the salt plume would be largely captured through pit dewatering.

Impact of solute transport from the ash co-disposal on the superficial aquifer was thought to be insignificant, as the leachate concentrations (including salts and metals) entering the groundwater system are lower than that in the local groundwater. The pit dewatering would capture the solute plume during the mining period.

6.3.2   Predictions of water level recovery and water quality for mine closure and the final void

Water level in the final void and groundwater level around the final void recovered after the mining. Due to high evaporation rate from the open surface water, the final mined void will form a groundwater sink and be the local focus of groundwater flow. It was predicted that the final water level in the void is within 4 m of the original watertable. The residual drawdown cone should stabilise after 60 years of mine closure.

Drawdown impacts of the final void on the surrounding groundwater level should be limited to the area nearby the void and not significantly impact the regional groundwater flow system in the CCM.

The final void is expected to form a capture zone where metals and salt in the concentration plume generated from the ash co-disposal will be transported by groundwater flow to the final void.

At the same time, the void will continue to receive the groundwater seepage, which brings salt and dissolved metals from the surrounding areas. It is predicted that the salt concentration will be built-up progressively in the void, due to evaporation of water from the void.

The water quality in the final void will be determined by a cumulative process that comprises capturing of the residual solute plume generated from the ash co-disposal and concentrating of natural salts through evaporation. Long term salinity and metal concentrations were determined by water and mass balance calculations on outputs from the flow and solute transport models. It was found that the pre-mining groundwater salinity at the final void site was at 1,575 mg/L. This salinity level was reduced slightly when the leachate plume (dominated by rainfall recharge) reached the site. Water salinity in the final void increased gradually over the long term and reaches super-saline (with TDS over 100,000 mg/L) after 500 years of the mine closure. This trend is likely to continue until the water reaches its maximum concentration as determined by salt saturation indices, probably in excess of 300,000 mg/L TDS.

The influence of the final void on the local groundwater level and water quality is considered to be small and localised. Salt and metal concentration plumes should be confined within the mined area and continually captured by the final void.

Movement of solute, especially metals, can be affected by geochemical reactions, including absorption and cation exchange where metals are bound to aquifer materials. It is noted that the above solute transport modelling is based on non-reactive solute transport and the model result therefore represents a worst case scenario.

6.4 Assessment of Salt Co-Disposal Impacts on Groundwater Quality

It is estimated that the Coolimba Power Station will generate about 19,829 tonnes of salts from the evaporation ponds per year, of which sodium chloride will be the predominant salt (15,365 tonnes). Potential impacts of salt co-disposal were simulated based on the co-disposal of evaporative salt, the waste rock (overburden) and coal combustion ash.

The following results were produced by the 30-year mining dewatering simulation of the salt co-disposal option which includes disposal of evaporative salt (every four years), and waste rock and coal combustion ash (every year).

- Salinity level in the salt disposal areas is higher than the surrounding groundwater salinity, and increases.

- There is a strong dilution effect. The mined area is a local sink due to dewatering and preferential flow. As a result, the local groundwater flows towards the mined area, mixes with the hypersaline leachate and dilutes the hypersaline leachate salinity concentrations.

- Although the simulated leachate concentration is 320,000 mg/L, the loading is comparatively small due to the low flux or infiltration rate. So the effect is local during the mining period.

- Groundwater flow within the mined area is northerly, aligned to the mine path. The groundwater flow velocity is low as the CCM is of comparatively low transmissivity. This results in low dispersion of the salt.

The following results were produced by the 500-year post mining simulation.

- Salinity concentrations beneath the salt co-disposal areas increase continually and become hypersaline after only a few years. The effect of this hypersaline salinity on the local groundwater flow is poorly defined by the current model. Density driven solute transport modelling would be required to provide a reasonable assessment.

- The plume is predominantly located in the mined area and moves towards the final void.

- The final void would host crystalline salt. The final void has an estimated capacity to contain between 27 Mt to 51 Mt of crystalline salt. The annual NaCl salt production from the evaporation ponds is 15,438 tonnes/yr), about 0.03% to 0.06% of the final void capability.

- The long-term salinity concentration increases much more rapidly for the salt co-disposal option than for the ash co-disposal alone.

In summary, the result from the simplistic solute transport simulation of the salt, overburden and ash co-disposal suggests that the salt disposal option may produce significant salt accumulations in the groundwater system in the mined area, in particular in the final void. The system does not reach a steady state condition during the 500-year simulation period.

Section 6   Conclusion and Recommendation

It is noted that this work is based on non-reactive and non-density driven simulations. The parameters used in the current modelling may only provide an order-of-magnitude estimate, due to the lack of geochemical testing data and absence of a density-driven modelling platform.

The evaporative salt co-disposal will substantially elevate the metals concentrations, likely to above all guideline values. It needs to be recognised that we have not assessed this aspect in the report, due to lack of geochemical data.

It is recommended that a groundwater monitoring program be developed to monitoring the solute plume movement and capture by the final mine void. The model also needs to be refined to provide necessary absolute water level and water quality predictions for the mining operation.

It is also recommended that appropriate methods and schedules for the disposal of evaporative salts generated from the evaporative ponds of the Power Station need to be established to minimise the impact on the groundwater system.

Water quality criteria need to be developed for the major groundwater users in the area.

Section 7   References

ANZECC (2000a). Australian and New Zealand Guidelines for Fresh and Marine Water Quality: Aquatic Ecosystems. Australian and New Zealand Environment Conservation Council (ANZECC) and Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ). Canberra, ACT.

ANZECC (2000b). Australian and New Zealand Guidelines for Freshwater and Marine Water Quality: Livestock Drinking Water. Australian and New Zealand Environment Conservation Council (ANZECC) and Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ). Canberra, ACT.

Commander, D.P. (1978), Hydrogeology of the Eneabba Borehole Line, Western Australia Geological Survey, Annual Report 1977, p13-19.

Commander, D.P. (1980), Assessment of groundwater abstraction from mineral sand processing on Eneabba, WA geological Survey Hydrogeology Report 2147 (unpublished).

Commander, D.P. (1981), The hydrogeology of the Eneabba area, WA, University of Western Australia, MSc thesis (unpublished).

http://www.chemguide.co.uk/physical/phaseeqia/saltsoln.html on Salt Solubility.

Mattiske (2005). Progress Report – Aviva Project Area; Flora and Vegetation. December 2005.

MINSERVE (2005). Central West Coal Project – Perth Basin Western Australia. Scoping Study – Mining and Coal Preparation. January 2005.

NACC (Northern Agricultural Catchments Council Inc.) (2002). West Midlands Hydrology Project, Stage One Report: The Impacts of Hydrological Issues on Biodiversity and Agriculture in the West Midlands Region. January 2002.

Rockwater Pty Ltd. (1990). Water Supply Bore Completion Report (EWP 1 – EWP 8 [now EWP 21 –EWP 28]). May 1990. Report reference 8.7.1/90/1JP/R22.

Rockwater Pty Ltd. (1992). Bore Completion Report – Production Bores EWP 29 and 30 and Monitoring Bores EWM 22 - 25. January 1992. Report reference 8.7.1/92/1 R35/JP.

Rockwater Pty Ltd. (1997). Investigation of the Effects of Groundwater Production and Drawdown on Vegetation at Eneabba West. October 1997. Report reference 8.7.6/97/1.

Rockwater Pty Ltd. (2002). Eneabba West Annual Aquifer Review: Groundwater Monitoring for Period January to December 2001. February 2002.

Rockwater Pty Ltd. (2003). Eneabba West Annual Aquifer Review: Groundwater Monitoring for Period January to December 2002. June 2003.

Rockwater Pty Ltd. (2003). Eneabba East Mine: Results of Groundwater Monitoring January to December 2002. November 2003.

Rockwater Pty Ltd. (2008). Central West Coal Project: Pit-dewatering Assessment and Bore Completion Report. June 2008.

Section 7   References

Spitz, K and Moreno, J. (1996). APractical Guide to Groundwater and Solute Transport Modelling, Publication by John Wiley & Sons, Inc.

Stewart W., Miller S., Thomas J. and Smart R. (2003) Evaluation of the Effects of Organic Matter on the Net Acid Generation (NAG) Test. Proceedings of the 6th International Conference on Acid Rock Drainage, 14-17 July, 2003, Cairns, Australia.

Terrenus Earth Sciences (2008) Central West Coal Project and Coolimba Power Station Project – Geochemical Assessment of Overburden, Potential Coal Reject and Coal Combustion Ash, Report prepared for URS Australia.

URS (2005) Central West Iron Project Monitoring Well Installation. December 2005. Report reference 42906005/621-F7405.0.

URS (2006) An Elevation of Groundwater and Surface Water Characteristics for the Central West Iron Project Pre-Feasibility Study.

URS (2008) Predicted Impacts of Ash Co-Disposal on Groundwater for Griffin Mine.

Water and Rivers Commission. (1997) Hydrogeology of the Coastal Plain between Cervantes and Leeman, Perth Basin. Report No HG3. Hydrogeology Record Series.

Woodman Environmental Consulting Pty Ltd (2003) Assessment of the Health of Eucalyptus Camaldulensis at Lake Indoon. Prepared for Illuka Resources Limited Eneabba Operations. August 2003.

Section 8   Limitations

URS Australia Pty Ltd (URS) has prepared this report in accordance with the usual care and thoroughness of the consulting profession for the use of Aviva Corporation Limited and only those third parties who have been authorised in writing by URS to rely on the report. It is based on generally accepted practices and standards at the time it was prepared. No other warranty, expressed or implied, is made as to the professional advice included in this report. It is prepared in accordance with the scope of work and for the purpose outlined in the Proposal dated 8 May, 2008.

The methodology adopted and sources of information used by URS are outlined in this report. URS has made no independent verification of this information beyond the agreed scope of works and URS assumes no responsibility for any inaccuracies or omissions. No indications were found during our investigations that information contained in this report as provided to URS was false.

This report was prepared between August 2008 and March 2009 and is based on the available data, especially the groundwater modelling report by Rockwater (2008) and geochemical assessment by Terrenus Earth Sciences (2008) at the time of preparation. Evaporative salt data, disposal method and schedule are supplied by Aviva (2009). URS disclaims responsibility for any changes that may have occurred after this time.

This report should be read in full. No responsibility is accepted for use of any part of this report in any other context or for any other purpose or by third parties. This report does not purport to give legal advice.

This report contains information obtained by inspection, sampling, testing or other means of investigation. This information is directly relevant only to the points in the ground where they were obtained at the time of the assessment. The borehole logs indicate the inferred ground conditions only at the specific locations tested. The precision with which conditions are indicated depends largely on the frequency and method of sampling, and the uniformity of conditions as constrained by the project budget limitations. The behaviour of groundwater and some aspects of contaminants in soil and groundwater are complex. Our conclusions are based upon the analytical data presented in this report and our experience. Future advances in regard to the understanding of chemicals and their behaviour, and changes in regulations affecting their management, could impact on our conclusions and recommendations regarding their potential presence on this site.

Where conditions encountered at the site are subsequently found to differ significantly from those anticipated in this report, URS must be notified of any such findings and be provided with an opportunity to review the recommendations of this report.

Whilst to the best of our knowledge information contained in this report is accurate at the date of issue, subsurface conditions, including groundwater levels can change in a limited time. Therefore this document and the information contained herein should only be regarded as valid at the time of the investigation unless otherwise explicitly stated in this report.