April 2011

MACRAES PHASE III PROJECT

Groundwater Contaminant Transport Assessment - Deepdell Creek, North Branch Waikouaiti River and Murphys Creek Catchments

Submitted to:
Oceana Gold (New Zealand) Limited

REPORT

Report Number. 0978110-562 R006 vD

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Golder Associates

Executive Summary

Oceana Gold (New Zealand) Limited (OceanaGold) operates the Macraes Gold Project (MGP) located in east Otago, approximately 25 km west of Palmerston. The MGP consists of a series of opencast pits and an underground mine supported by ore processing facilities, waste storage areas and water management systems. As the result of a recent review of ore reserves, OceanaGold has concluded that mining operations at the Macraes Gold Project can be extended until 2020 through opencast mining of additional ore reserves.

New waste rock stacks (WRS's) and extensions to existing rock stacks are planned, increasing the total consented tonnage from 850 Mt to 1,180 Mt. A new WRS is planned substantially extending the existing Back Road WRS to the east of the Round Hill/Southern Pit locations. The existing Frasers East and Frasers West WRS's are to be expanded. A new linking rock stack between these two called Frasers South WRS and an extension to the north called Frasers North WRS are to be constructed.

OceanaGold is planning to decommission both of the existing tailings storage facilities (TSF's) by mid 2012 and commence using a new Top Tipperary TSF (TTTSF).

Golder Associates (NZ) Limited (Golder) has been retained by OceanaGold to undertake an evaluation of tailings water seepage and contaminant losses from the MGP, taking into account the planned changes to the mine site through to 2020.

Leachate from the TSF's and from the WRS's at the site is expected to discharge to:

- The drainage systems built into the TSF's
- The opencast pits
- The main creeks and tributary gullies close to the site, including those reaches managed through the use of sediment settling ponds and unmanaged sections

During the operational period of the mine, OceanaGold proposes to manage discharges from the TSF drainage systems built into the TSF's and water accumulating in most sediment settling ponds by returning the collected water to the ore processing system. Seepage water that cannot be collected by either the drains or the sediment settling ponds is expected to eventually discharge to the regional drainage system.

The Visual MODFLOW Pro software package was used to construct and calibrate the groundwater model. Modelling of contaminant mass transport within the groundwater system at the MGP has been undertaken to cover the period from 2010 through to the close of mining operations at the site at the start of 2020 followed by a 150 year post-closure period. Beyond that period, potential changes in the hydrogeological behaviour of the tailings material and climactic conditions are considered to limit the usefulness of predictive modelling.

The groundwater model used for this assessment has been based on an existing calibrated model simulating the groundwater system at the site through to 2010. Groundwater recharge rates applied to the TSF's are calibrated to ensure the water table within the tailings is at the tailings surface during the operational period of the mine. The regional recharge rate of 32 mm/year has also been applied to the rehabilitated WRS's and TSF's at the site.

Groundwater quality input parameters have been based on water quality data from the site environmental monitoring programme. These include leachate water quality representing TSF decant ponds, TSF drain discharges and WRS seepage. The water quality parameters simulated include the major ions, a range of metals, arsenic and cyanide$_{WAD}$.

Contaminant transport for each of the simulated contaminants with the exception of arsenic has been undertaken on the basis of conservative transport within the groundwater system. Arsenic transport has been modelled based on arsenic (III) being the main form of this element in the tailings seepage water. The adsorption parameters for arsenic (III) have been derived from testing of rock and soil samples from the site.

At this stage, arsenic and sulphate are considered to be the primary contaminants of concern with respect to compliance with existing and proposed consent limits for water quality. Mass loads for other contaminants entering the groundwater system from mine site operations have however also been calculated and reported.

The maximum simulated contaminant mass load in water discharging to natural water bodies at the Macraes Gold Project occurs after the site has been closed, with the exception of the tailings storage facility drainage systems. It requires a considerable period of time for contaminants to be transported through the groundwater system at the site to the receiving water bodies. Simulated groundwater discharge rates at the time of maximum discharge mass load and the associated average sulphate and arsenic concentrations are summarised in the table below.

Receiving Water	**Groundwater discharge rate** (1)	**Arsenic** (2)	**Sulphate** (2)
Deepdell Creek upstream from DC07	730	0.03	590
Deepdell Creek between DC07 and DC08	116	0.05	1,050
Murphys Creek upstream from MC100	180	0.03	380
North Branch Waikouaiti River upstream from NBWRRB	100	0.03	1,050

Notes: 1) Post-closure groundwater discharge rate: units of m^3/day.
2) Concentrations in groundwater discharges: units of g/m^3.

Drainage discharges from the combined MTI and SPI at closure were simulated to be approximately 1,800 m^3/day. The simulated TSF drainage systems are limited in detail and the drainage systems built into the uphill raises of the MTI and SPI are poorly represented. This lack of detail leads to understatement of drainage flows that may be expected at closure. Monitoring of drainage flows at the site indicates the total flows at closure are more likely to be in the order of 2,500 m^3/day (Golder 2011c).

Simulated MTI and SPI drain discharge rates indicate a decrease in flows of approximately 50% within a period of 10 years following closure. It is, however, likely that this decrease in drain discharges is conservative. An assessment of the rates at which MTI and SPI drain discharges have declined during inactive periods in the past, indicates discharges are likely to decline at a faster rate of between 50% and 90% within two years following closure (Golder 2011c).

It is expected that much of the stored tailings mass would become unsaturated during the 20 years following closure of a TSF. There is, however, considerable uncertainty with respect to the length of time required for the overall groundwater system to reach a steady state flow pattern. This uncertainty is partly due to the inherent variability of the hydrogeologic characteristics of the tailings mass. In addition, dynamic factors such as compaction of both the tailings mass and the underlying soils have not been taken into account in this projection.

Once the groundwater systems within the tailings storage facilities have reached a steady state following closure the contaminant loads in water subsequently lost from the tailings would be associated with the residual moisture content and ongoing recharge from rainfall. Further transport of contaminants from the tailings would mainly occur in response to significant rainfall events. These events would lead to pulses of seepage water travelling downward through the unsaturated tailings to the groundwater table. These pulses, averaged on a long term annual basis, are expected to be equivalent to the natural 32 mm/year groundwater recharge rate for the region.

ABBREVIATIONS

CTI	Concentrate Tailings Impoundment
EGL	Engineering Geology Limited
FF	Footwall Fault
FTI	Flotation Tailings Impoundment
HMSZ	Hyde Macraes Shear Zone
HWS	Hanging Wall Shear
ISS	Intrashear schist
K_d	Distribution coefficient
MGP	Macraes Gold Project
mRL	Metres above mean sea level
MTG	Maori Tommy Gully
MTI	Mixed Tailings Impoundment
ORC	Otago Regional Council
SPI	Southern Pit Tailings Impoundment, the combined SP10 and SP11
SP10	Southern Pit Tailings Impoundment SP10 currently incorporated in SP11
SP11	Southern Pit Tailings Impoundment SP11
TSF	Tailings Storage Facility
TTTSF	Top Tipperary Tailings Storage Facility
WRS	Waste rock stack

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1.0 INTRODUCTION

1.1 Background

Oceana Gold (New Zealand) Limited (OceanaGold) operates the Macraes Gold Project (MGP) located in east Otago, approximately 25 km west of Palmerston (Figure 1). The MGP consists of a series of opencast pits and an underground mine supported by ore processing facilities, waste storage areas and water management systems (Figure 1).

OceanaGold has an ongoing program of exploration drilling, ore reserves review and mine design optimisation. Consequently, operational pit designs are regularly updated. The performance of existing waste storage facilities and the requirement for additional waste storage capacity is also regularly reviewed. As the result of a recent review of ore reserves, OceanaGold has concluded that mining operations at the Macraes Gold Project can be extended until 2020 through opencast mining of additional ore reserves. These reserves include:

- Reserves located to the east of the existing Frasers Pit, to be accessed through expanding Frasers Pit.
- Reserves located to the east of the Round Hill Pit, which can be accessed through reclamation of tailings from within the current SP11 tailings storage facility (SP11) and removal of waste rock fill that has previously been stored in Round Hill Pit. The tailings are to be relocated to the existing Mixed Tailings Impoundment (MTI) and the new Top Tipperary tailings storage facility (TTTSF) to be constructed in the Tipperary Creek catchment.
- Reserves located to the east of Innes Mills Pit, to be accessed through removal of the waste rock previously stored in Innes Mills Pit and expanding the pit.

The proposed opencast pit expansions are termed the Frasers Stage VI, Roundhill Extension, Southern Pit and Innes Mills Stage V.

New waste rock stacks (WRS's) and extensions to existing rock stacks are planned, increasing the total consented tonnage from 850 Mt to 1,180 Mt. A new WRS is planned substantially extending the existing Back Road WRS to the east of the Round Hill/Southern Pit locations. Frasers East and Frasers West Rock Stacks will be expanded with a new linking rock stack between these two called Frasers South Rock Stack and an extension to the north called Frasers North WRS to be constructed.

OceanaGold is planning to decommission both of the current tailings storage facilities (TSF's) by mid 2012 and commence using a new TSF. At this point it is likely that both existing TSF's will have remaining resource consent life, however in review of OceanaGold's new mining schedule and the pro's and con's of various options it is a more effective alternative to switch to the new facility.

OceanaGold is seeking to obtain resource consents covering:

- The construction and operation of the TTTSF
- The construction of the planned additional WRS's
- The expansion of the existing opencast pits

Golder Associates (NZ) Limited (Golder) has been retained by OceanaGold to undertake an evaluation of tailings water seepage and contaminant losses from the MGP, taking into account the planned changes to the mine site through to 2020. This report presents the results of studies undertaken to assess groundwater flow and contaminant mass transport budgets for the MGP, excluding the Tipperary Creek catchment and the effects of the TTTSF. A technical assessment of contaminant losses from the TTTSF has been documented separately (Golder 2011a). The outcomes of this evaluation are to be used in support of an Assessment of Environmental Effects.[1]

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[1] This report is provided subject to the conditions and limitations presented in Appendix A.

Golder Associates

TITLE SITE LAYOUT

MARCH 2011

PROJECT 0978110562

1

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1.2 Project Description

1.2.1 Main features

The main features of the project are:

- A new tailings storage facility, labelled the Top Tipperary Tailings Storage Facility (TTTSF), which is to be constructed in the upper Tipperary catchment basin. It would result in an increase of 51 Mt of total consented tailings storage capacity (from 81 Mt currently to 132 Mt).
- Reclamation of tailings from within the current SP11 tailings storage facility. The tailings are to be relocated to stacks within the footprints of the existing MTI and the new TTTSF.
- New WRS's and extensions to existing WRS's are to be constructed, increasing the total consented tonnage from 850 Mt to 1,180 Mt. A new WRS is planned substantially extending the existing Back Road WRS to the east of the Round Hill/Southern Pit locations. Frasers East and Frasers West WRS's are to be expanded. A new linking WRS between these two called Frasers South WRS is to be constructed and an extension to the north of Frasers East WRS called Frasers North WRS.
- Expansion of existing pits to include the Frasers Stage VI, Round Hill Extension, Southern Pit and Innes Mills Stage V.
- Surface water on the expanded mining infrastructure shall be managed with diversions and new silt control dams.
- A revised closure plan which will comprise three lakes formed from the pit excavations.

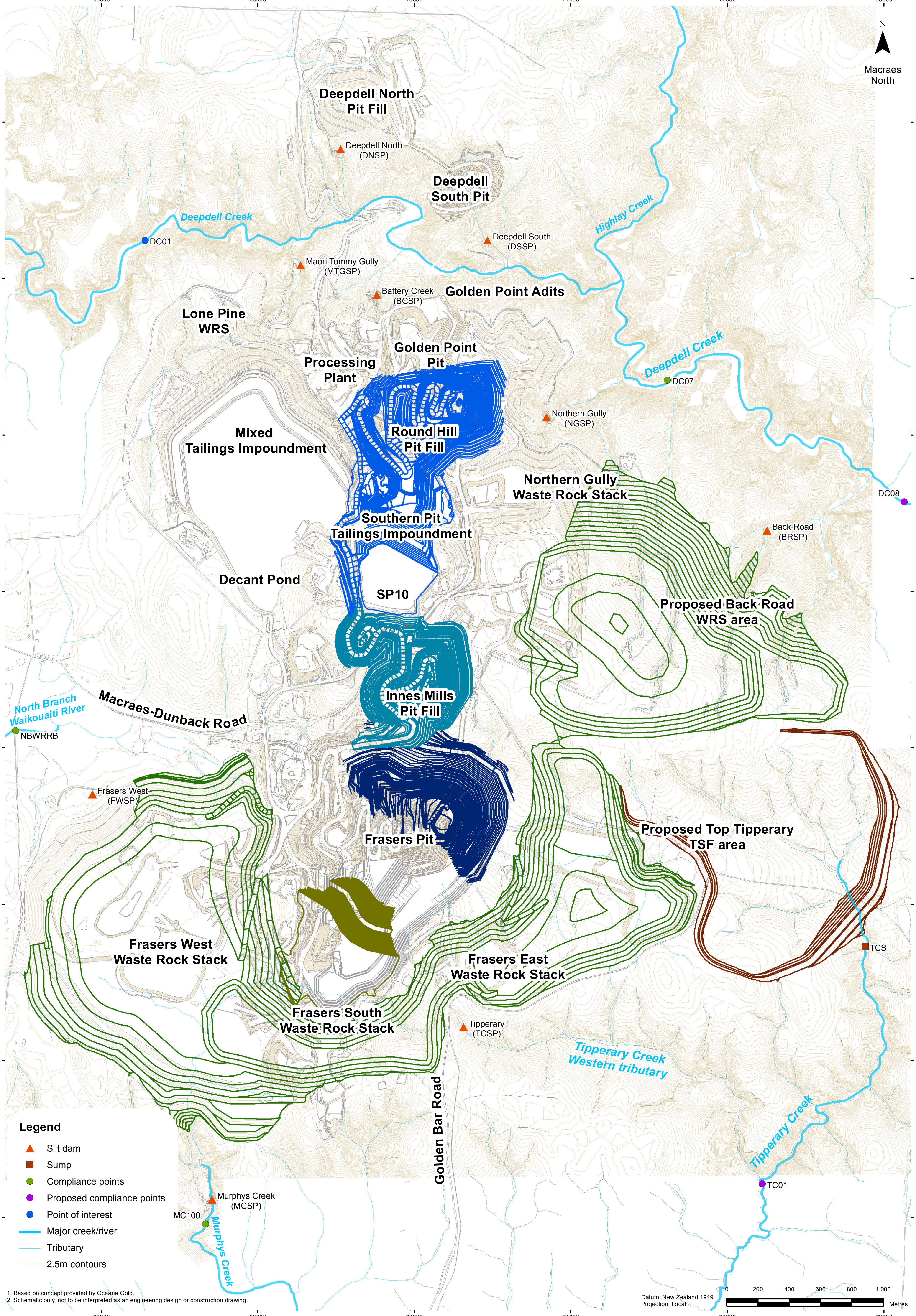
The processing rate is to be similar to current operations and the intensity of operations on site will be similar to present.

1.2.2 Tailings storage

At Macraes there are currently two active TSF's. These are the MTI and the SP11 impoundments (Figure 2). A previous TSF, the Southern Pit SP10 Impoundment (SP10), sits within the SP11 and has been inundated by tailings deposited in SP11.

OceanaGold needs capacity to store an additional 43.5 Mt of tailings to take the MGP from mid 2012 through until early 2020 at current processing rates. It is likely that both existing TSF's will have remaining resource consent life at mid 2012, however a review of OceanaGold's new mining schedule has indicated it is a more effective alternative to switch to a new facility. It is envisaged there will be one final tailings deposition phase into SP11 from circa January 2011 until June 2011, whilst a final consented upstream lift on the MTI is undertaken. Deposition is currently occurring in the MTI and a planned final deposition period is to occur from circa June 2011 to May 2012.

OceanaGold is planning to decommission both of the current TSF's by mid 2012 and commence using the new TTTSF. The final TTTSF footprint is 184 ha and the tailings storage capacity is to be 38,744,000 m^3. The crest height is planned to be 560 mRL and the operating height would be 70 m at the highest embankment point. Following decommissioning a process of closure and rehabilitation would commence on both existing TSF's. In the case of the SP11, tailings stored in the outer compartment (north of the internal SP10 wall), are to be mechanically re-handled once dry enough. The recovered tailings are to be placed as a reclaimed tailings stack (RTS) on top of the MTI and/or into the new TTTSF. The tailings stored in SP10 above the level of the SP10 embankment are also to be partially removed, so that SP10 is effectively reinstated as an existing, decommissioned TSF. The MTI and SP10 structures are to remain in place. The SP11 outer wall being removed as tailings are stripped down.



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TITLE MACRAES PHASE III PROJECT LAYOUT

MARCH 2011

PROJECT 0978110562

2

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The TTTSF is planned to be located in the headwaters of Tipperary Creek outside of the catchments of the North Branch of the Waikouaiti River (NBWR) and Deepdell Creek. It is proposed to pump tailings to the TTTSF via a pipeline from the processing plant and deposit the tailings from the TTTSF embankment. The footprint of the TTTSF is to gradually expand, reaching its maximum extent by about 2017. The rehabilitation plan at closure calls for TSF facilities to be fully capped with brown rock and topsoil and for pasture to be re-established.

1.2.3 Waste rock storage

There are currently two main WRS's in use at Macraes, the Frasers West WRS and Frasers East WRS. Other WRS's at Macraes include Deepdell, Western, Northern Gully South, Northern Gully North and Back Road.

There currently remains 149 Mt of consented capacity in the existing WRS's. Macraes Phase III calls for an increase in WRS capacity of 331 Mt. In summary this additional capacity is allocated to:

i) Back-Road 228 Mt

ii) Frasers South 50 Mt

iii) Frasers West 27 Mt

iv) Frasers East Extension 26 Mt

The Frasers East WRS, which is currently under construction, is to be expanded to include a northern addition to the current consented WRS (additional capacity of 26 Mt). This expansion, along with construction of the TTTSF, will necessitate the realignment of a 4.5 km stretch of the Macraes-Dunback road. Construction is expected to be completed in early 2013.

The existing Back Road WRS is to be expanded (additional capacity of 228 Mt), with construction to start early in 2013 and continue until 2019. The existing Back Road WRS is located on the eastern edge of Southern and Round Hill Pits. Its expansion is to incorporate material removed from these two pits along with some material from Innes Mills Pit. The Back Road WRS is to be wholly contained within the Deepdell Creek catchment. The foot-print of this WRS is to reach a maximum of 234 ha, with a maximum elevation of 650 mRL. This will result in a maximum WRS height of 65 m above natural topography.

The Frasers South WRS is to be located on the southern edge of the Frasers Pit, connecting the current Frasers East and Frasers West WRS's. It is to straddle the catchment divide of Murphys Creek and the NBWR. This WRS is to reach a maximum height of 590 mRL, or 45 m above natural topography

The average annual rate of waste rock deposition is planned to be circa 55 Mt, based on the following likely sequence of WRS construction.

i) Within the existing consented WRS footprints until mid-late 2012

ii) Back Road WRS from early-mid 2012 till 2015-16 then

iii) Return to the Frasers South, East and West extension WRS's

Rehabilitation of the completed WRS's is to be undertaken using mining equipment on a progressive basis. Once final profiles are achieved then a layer of brown rock (highly weathered schist) is to be placed over the fresh waste rock and track rolled. Over this is placed a layer of top-soil. Fertilising and seeding of grasses shall then be undertaken to return the ground to a pre-mining pasture.

1.2.4 Macraes Phase III schedule

A schedule of mining operations has been developed by OceanaGold which has been divided into four phases for the purposes of mine water management. These phases have not been specifically incorporated into the groundwater model, as it was not considered necessary to simulate each stage of the proposed mine development in detail.

The delays that are associated with contaminant transport in groundwater across the site mean these stages are of little value in terms of contaminant losses to surrounding receiving water bodies. As the outcomes of the groundwater model are also to be applied to the mine water management model, the operational stages associated with that model are summarised below.

There are no operations currently planned for the Deepdell North or South pits during the Macraes Phase III Project. Deepdell North pit has been backfilled and rehabilitated. Deepdell South pit is inactive and a lake is developing in this pit.

Stage 0: January 2010 to February 2012

- Stage 0 is representative of site conditions up until the commencement of the Macraes Phase III development at the end of February 2012. Frasers Pit is operational. Golden Point, Round Hill and Innes Mills pits are inactive. Round Hill and Innes Mills pits are full of waste rock. Frasers and Golden Point pits are being actively dewatered.
- Waste rock placement is to Frasers West and Frasers East WRS's.
- Tailings from the process plant are alternatively placed in both the MTI and the SP11 TSF. Seepages from the TSF's are collected by the impoundment drainage systems and returned to the process plant. Process plant water requirements exceed the volume of return water available. Make-up water is pumped from the Taieri River.
- Seepage water collecting in the Northern Gully and Maori Tommy Gully is either pumped back to the process plant or used for dust control.

Stage 1: March 2012 to December 2015

- Stage 1 incorporates most of the Frasers Stage Six pit expansion. Golden Point, Round Hill and Innes Mills pits are inactive. Round Hill and Innes Mills pits are full of waste rock. Frasers pit is being actively dewatered. Water levels in Golden Point pit are being actively managed.
- Waste rock placement is to Frasers East, Frasers North and Frasers South WRS's. Frasers West WRS is being rehabilitated.
- Tailings from the process plant are stored in the TTTSF. The MTI, SP10 and SP11 are inactive, are becoming dewatered and drain discharges are declining. Tailings from SP11 are recovered and dry stacked on the MTI or transported to the TTTSF. Seepages from the TSF's are collected by the impoundment drainage systems and returned to the process plant. Process plant water requirements exceed the volume of return water available. Make-up water is pumped from the Taieri River.
- Seepage water collecting in the Northern Gully and Maori Tommy Gully is either pumped back to the process plant or used for dust control.

Stage 2: January 2016 to December 2017

- Stage 2 incorporates the period of mining in Round Hill pit and the conclusion of mining in Frasers Pit. Innes Mills pit is inactive and full of waste rock. Round Hill pit is being actively dewatered. Water management for Frasers pit has ceased and a pit lake is starting to develop.
- Back Road WRS has become the active waste rock storage area. Frasers East, Frasers North and Frasers South WRS's are being rehabilitated.
- Tailings from the process plant are stored in the TTTSF. The MTI and SP10 are being rehabilitated and drain discharges are declining. The SP11 embankment has been removed. Seepages from the TSF's are collected by the impoundment drainage systems and returned to the process plant. Process plant water requirements exceed the volume of return water available. Make-up water is pumped from the Taieri River.
- Seepage water collecting in the Northern Gully and Maori Tommy Gully is either pumped back to the process plant or used for dust control.

Stage 3: January 2018 to December 2019

- Stage 3 incorporates the period of mining in Innes Mills pit. The waste rock fill is removed as part of this operation.
- Waste rock placement is to Back Road WRS and to the relatively small Frasers South in-pit stack.
- Tailings from the process plant are stored in the TTTSF. The MTI and SP10 are being rehabilitated and drain discharges are declining. Seepages from the TSF's are collected by the impoundment drainage systems and returned to the process plant. Process plant water requirements exceed the volume of return water available. Make-up water is pumped from the Taieri River.
- Water management for Round Hill pit has ceased and pit lakes are developing in both Round Hill and Frasers pits.

Stage 4: January 2020 to December 2169

- Stage 4 represents long term closure of the Macraes Phase III site.
- All mining operations have ceased. Pit lakes are developing in Frasers, Round Hill and Innes Mills pits.
- Rehabilitation of all WRS's is completed.
- Site management is undertaken to minimise and mitigate environmental effects. For an initial period following mine closure tailings seepage collected by impoundment drainage systems will be of high quantity and poor quality. Over time the quantity declines and the quality improves. Collected tailings drain discharges are to be initially pumped to Frasers Pit until they can be either passively managed or released to the environment without exceeding consent conditions.

1.3 Scope of Work

The scope of work for this study is to:

1) Assess the potential rate of contaminant losses to groundwater from the MGP site, including the existing TSF's and the existing and planned WRS's.

2) Evaluate potential contaminant mass loads discharging from the MGP to Deepdell Creek, the NBWR, Murphy's Creek and their tributaries.

The assessment of contaminant losses to groundwater and discharges to Tipperary Creek and Cranky Jims Creek has been documented in a separate report (Golder 2011a).

The above objectives have been fulfilled through the construction of a 3D groundwater flow and contaminant transport model to simulate seepage flows across the site and the potential transport of contaminants lost from the MGP as a whole. The model is based on similar models of the MGP site groundwater system developed by Kingett Mitchell Ltd (Kingett Mitchell) to support past resource consent applications. The model simulates an eight year facility operational period followed by a 150 year post-closure period.

1.4 Previous Studies

A range of groundwater evaluations at the MGP site have been undertaken in support of mining feasibility studies and subsequently in support of applications for resource consents to authorise various operations at the site. These past studies have been summarised in a report by Kingett Mitchell (2005a).

Prior to 2002 assessments of groundwater mass transport budgets at the mine site concentrated on specific questions and the requirements of individual consents for the mine site. Increasing mine site complexity, expanding tailings storage requirements and concerns about the cumulative contaminant loading to regional drainage systems led to a shift from analytical groundwater flow path calculations to digital modelling of groundwater movement across the entire site. As the MGP has increased in area, digital groundwater simulations have become progressively larger and more complex (Kingett Mitchell 2002, 2005a).

The outcomes from groundwater and mass transport models developed by Kingett Mitchell for the MGP site have been used in support of several applications by OceanaGold for resource consents since 2002. The model outcomes have been accepted by the Otago Regional Council. Although it was possible to extend the most recent site-wide groundwater model (Kingett Mitchell 2005a) to include the TTTSF facility, it was considered more efficient to simulate details of the TTTSF embankment in a separate model. This model has been documented in a separate report (Golder 2011a).

Previous assessments have also been undertaken of the groundwater system at Macraes as it relates to the construction of the Frasers East WRS (Kingett Mitchell 2005b) and the Frasers Underground Mine (Kingett Mitchell 2006).

The modelling undertaken during previous studies has been taken into account in the current study. In particular, the structure of the groundwater model and the values applied to input parameters have been derived from these previous studies. Where additional calibration of hydrogeological parameters has been undertaken, the process is discussed in the appropriate sections.

1.5 Supporting Studies

The site wide groundwater and contaminant transport assessment documented in this report refers to several supporting studies undertaken to compliment or support this one. These studies include:

- An assessment of groundwater flows and contaminant transport within the Tipperary Creek catchment related to the planned construction of the TTTSF (Golder 2011a).
- An assessment of potential seepage losses and flows through the historical underground workings located in the pit wall between Golden Point Pit and Deepdell Creek during filling of the pit lake. This study was undertaken to assess the effect lining the northern pit wall would have on reducing water loss through the underground workings (Golder 2011b).

- An assessment of the rates of decline in TSF drainage flow discharges following the close in operations at the TSF. The existing TSF's at the MGP have been alternately allowed to remain dormant for extended periods over the past six years and drainage discharge flows have decreased each time. This study was undertaken to assess how rapidly the discharge flows could be expected to decline based on environmental monitoring data from the MGP site (Golder 2011c).

1.6 Report Structure

In addition to the introductory Section 1, this report contains the following sections:

- Section 2 summarises the hydrogeology of the site.
- Section 3 summarises the quality of groundwater, natural surface water, tailings decant and pore water and waste rock seepage water at the site. In addition, the attenuation of contaminants during transport within the groundwater system is discussed in this section.
- Section 4 summarises the conceptual groundwater model of the site.
- Section 5 summarises the translation of the conceptual model into a numerical model. In addition this section outlines important aspects of the numerical model and the calibration and validation process.
- Section 6 presents the contaminant discharge projections derived from the groundwater model.
- Section 7 summarises this report and presents the conclusions of the investigation.
- A list of referenced reports and documents referred to in this document is provided in Section 8.

2.0 HYDROGEOLOGY

2.1 Topography and Drainage Pattern

The topography of the Macraes Flat area is that of an ancient erosional surface, or peneplain, which has been bisected by Deepdell Creek, the NBWR, Tipperary Creek and Murphys Creek. Deepdell Creek, which flows toward the northeast, and Tipperary Creek, which flows toward the southeast, both discharge into the Shag River. The NBWR flows toward the southwest past Macraes Flat, then turns to flow toward the southeast. Murphys Creek initially flows toward the southwest then turns toward the south to discharge into the NBWR (Figure 1).

The groundwater study area mainly covers sections of the Deepdell Creek and the NBWR catchments, however small sections of the Murphys Creek and Tipperary Creek catchments are also included.

Deepdell Creek and its tributaries, such as Maori Tommy Gully (MTG), Battery Creek and Northern Gully are deeply incised into the peneplain surface, with steep valley slopes and a narrow or no alluvial terrace. Tributary streams generally have steep gradients. The upper reaches of many of the steep tributary gullies have developed along lines of structural weakness in the basement schist.

The NBWR upstream from Macraes Flat has, in contrast, created a broad shallow valley. Through deposition of sediment in the valley bottom alluvial flats have developed up to 500 m wide that form a continuous feature between Glendale Station and Macraes Flat (Figure 1). The headwaters of Tipperary Creek around Glendale Station and the headwaters of Murphys Creek are both characterised by relatively gentle valley slopes. Neither of these creeks has however developed broad alluvial flats.

Mining operations since 1990 have created a series of opencast pits along the line of the Hyde Macraes Shear Zone (HMSZ) with associated waste rock stacks, tailings impoundments, water supply and silt ponds and various other subsidiary structures (Figure 1).

2.2 Geology

2.2.1 Introduction

The eastern area of Otago is underlain principally by Mesozoic age schist of the Torlesse Terrane (Forsyth 2001). Weathering and erosion over a long period formed the distinctive low relief of the Otago peneplain. Deposition of alluvium, rich in quartz gravel occurred in east Otago during the Eocene (Hogburn Formation) and Miocene (Manuherikia Group). Miocene age volcanics were also widespread. Post-Miocene tectonic deformation and erosion has removed most of the Tertiary age deposits, along with an unknown thickness of schist. The resulting landscape in the Macraes area comprises widespread outcrops of schist and thin cover soils (Figure 3).

2.2.2 Schist

The schist, being a crystalline metamorphic rock, has effectively no primary or intergranular porosity or permeability, except where weathered. Secondary porosity and permeability in the form of fractures and faults provide the major groundwater seepage routes below the surficial, strongly weathered zone.

It is considered that hydraulic conductivity of the schist increases upward through the schist rock mass due to the increasing intensity of weathering and reducing overburden pressures. Similar trends or decreasing rock mass permeability with depth have been recorded with respect to fractured crystalline rocks in other areas of the world (e.g., Masset & Loew 2010). This trend has been incorporated in several groundwater models of the MGP site (Kingett Mitchell 2002, 2005a) and is based primarily on an assessment of hydraulic conductivity variation with depth for the Maori Tommy Gully area (GCNZ 1988).

2.2.3 Alluvium and colluvium

Exploratory and geotechnical drilling and landform comparison indicates that a thin layer of loess covers much of the MGP area. The loess soils comprise a very stiff, light yellow grey silt, sandy silt or silty fine sand. Subsurface investigations identified a surficial cover of loess, colluviums and topsoil. Geotechnical investigations in the area of the proposed Back Road WRS and the TTTSF typically exposed 0.2 m to 0.4 m of soil materials, with a maximum identified thickness of 1.8 m (EGL 2010; Golder 2011d).

Along the catchment divides occasional small bogs have developed on top of the loess. These bogs are generally located in slight depressions where run-off is concentrated and shallow drainage may be impeded by accumulated dust and decomposing vegetation.

Colluvium has accumulated at the base of steep slopes around the MGP site. Colluvium mainly comprises fine angular schist gravel in a sandy or silty matrix, with the matrix mainly derived from reworked loess.

The alluvial fill in the creek valleys across the site is generally not considered to have a substantial effect on the regional groundwater flow regime. The fill is neither voluminous enough nor covers a sufficient area to act as an aquifer or aquitard at the scale represented in the groundwater model. In the NBWR, the alluvial valley fill acts as a thin localised aquifer, parallel to the river. Contaminants leached from the Frasers West WRS are apparently being transported within this alluvium toward the NBWR (Golder 2011e). On the scale of the site wide model however this contaminant transport route is not considered to be sufficiently different in hydraulic characteristics from the weathered schist to justify simulating this material separately in the model.

1. SOURCE: Institute of Geological and Nuclear Sciences Geological Map 7. Copyright Reserved

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The small thicknesses of loess and colluvium have not been specifically simulated in the groundwater flow model as this material is not expected to have a major effect on groundwater flow routes and rates at the mine site. Groundwater that seeps along the schist/colluvium interface appears either to surface rapidly as ephemeral springs in discharge areas or to be lost to evaporation.

The colluvium and loess do however have a significant effect on the transport of arsenic by groundwater due to the adsorption of arsenic onto the soils and oxidised schist. Arsenic adsorption parameters applicable to these materials have been incorporated in the upper layer of the contaminant transport simulation (refer Section 6.4).

2.2.4 Geological structures

Foliation

The schist bedrock at the MGP site is characterised by eastward dipping foliation and foliation parallel fractures. These foliations typically dip about 15° to 30° towards the east or south east. Foliation orientations rotate approaching major faults in the area, such as Macraes Fault (Golder 2011f).

Discontinuities observed in the schist comprise mainly foliation partings. In addition to the foliation parallel discontinuities, several major structural features or sets have been documented from the MGP site.

Hyde Macraes Shear Zone

The Hyde Macraes Shear Zone (HMSZ) consists of three major physical components:

- The Hanging Wall Shear (HWS)
- The Intrashear Schist (ISS)
- The Footwall Fault (FF)

The position of both the HWS and the FF has been defined through interpolation of intersect data from drilling programs. As the ore mineralization is focused on the HWS and on the immediately underlying ISS, fewer drillhole intersects are available for the FF and consequently geometric control on the FF is less well defined.

Within both the HWS and the FF the structure of the host schist has been completely disrupted, with no well defined fracture sets apparent. Both the HWS and the FF are expected to be characterised by greater hydraulic conductivity parallel to the respective structures than perpendicular to them. This is due to the presence of fault gouge materials forming layers within these two structures. In contrast, review of the structural features of the ISS indicates it is likely to display similar hydraulic conductivity characteristics to the host schist.

Structural deformation combined with mineralisation may have changed the ratios between vertical and horizontal hydraulic conductivities within this material when compared to schist outside the HMSZ. The general reduction in hydraulic conductivity with depth is however expected to follow a similar trend to that identified from schist outside the HMSZ.

Major faults

In addition to the HMSZ, three major northeast – southwest trending tectonic structures intersect the MGP area. These structures are:

- The Deepdell Fault, which has primarily been defined through an offset of the HMSZ indicated from interpretation of exploratory drillhole logs. This fault is aligned with the valley floor of Deepdell Creek in the area of the MGP.

- The Macraes Fault, which intersects the northern end of Frasers Pit and forms the northern boundary to Frasers Underground. This fault is inferred to affect a zone approximately 700 m wide to the north of Tipperary Creek. An assessment of the Macraes Fault did not find evidence of deformation associated with movement of the Macraes Fault during the last 12,000 to 15,000 years (Golder 2011f). The schist observed in investigation trenches close to the Macraes Fault is generally highly fractured and contains a significant proportion of gouge material.
- A third fault has been interpreted to be aligned with Murphys Creek immediately to the south of Frasers Pit.

Secondary faults and joints

North to northwest striking high angle faulting has been identified through interpretation of drillhole data, evaluation of aerial photograph lineaments and direct mapping of outcrops during investigation of potential tailings embankment locations (EGL 2010; Kingett Mitchell 2005a; Golder 2011d).

Joints in the TTTSF area are typically steeply dipping, have a rough surface and are planar to undulating. The most common strike is approximately southeast.

2.3 Weathering

The intensity of chemical weathering of the schist rock mass decreases with increasing depth. Geotechnical investigation of the TTTSF and Back Road WRS areas has indicated that moderate weathering of the schist as indicated from drillhole cores generally does not extend past a depth of about 5 m, while slight weathering has been identified in drill cores to a depth of about 35 m (EGL 2010; Golder 2011d). From a geochemical viewpoint, rock that is considered to be highly and moderately weathered extends to greater depths. This is significant in terms of the adsorption of arsenic onto the weathered surfaces of discontinuities in the schist rock.

In assessing the transport of contaminants beneath and down-gradient from the MGP, the highly geochemically weathered zone is considered to extend downward for between 5 and 10 m. Below that depth the degree of weathering decreases to moderate and slight geochemical weathering of the rock faces.

2.4 Climate and Groundwater Recharge

An assessment of long term average rainfall for the site based on rainfall records from the Glendale and Deepdell monitoring stations indicated the average annual rainfall across the site was approximately 607 mm/year (Kingett Mitchell 2005a). Although there is a slight seasonality to the rainfall pattern (Figure 4), this variation is not considered to be significant for groundwater recharge assessment purposes. An updated assessment of the annual rainfall (Golder 2011g) indicates the Glendale and Golden Point sites receive average annual rainfall totals of around 628 mm and 659 mm, respectively. The Deepdell site receives around 518 mm annually. This latter value has decreased from the value applied in the 2005 assessment.

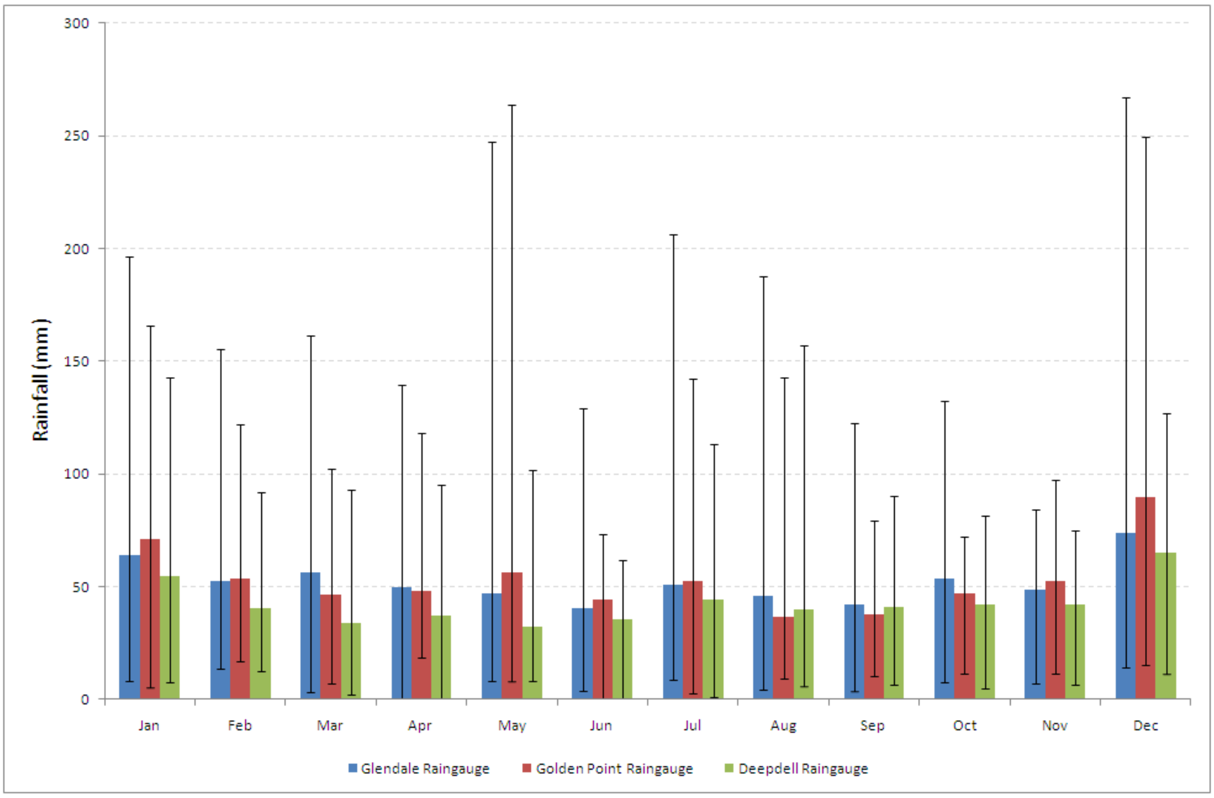


Figure 4: Monthly rainfall summaries for Deepdell, Glendale and Golden Point rain gauges.

An evaluation of the Deepdell catchment water balance performed by Kingett Mitchell (2005a, Appendix 3) has been based on:

- An average annual rainfall of 607 mm/year across the whole of the MGP site
- An average annual evaporation of 1,092 mm
- An average annual open water evaporation of 764 mm

The calculated regional groundwater recharge rate was approximately 32 mm/year. As the annual rainfall across the site may be slightly less than what was calculated for this recharge assessment, it is possible the outcome slightly overstates the annual recharge rate. This recharge value has however been used in previous groundwater flow and mass transport modelling of the MGP (Kingett Mitchell 2005a) and is retained for this purpose in the current study.

The runoff characteristics of the NBWR catchment probably differ from those of the other catchments intersecting the study area, given the lower topographic and stream gradients in this catchment. For the purposes of this study it has however been assumed that the infiltration characteristics are the same for all catchments within the model area. This assumption has been incorporated in previous mine site studies relating to the rate of water rise in Frasers Pit following closure (WWC 2001, Kingett Mitchell 2005a).

2.5 Rock Mass Permeability

It is commonly reported that hydraulic conductivity increases upward through the schist rock mass due to the increasing intensity of weathering. This trend has been incorporated in several groundwater seepage models (Kingett Mitchell 2002, WWC 1996, 2001). This conclusion is based on an assessment of conductivity variation with depth for the Maori Tommy Gully area (GCNZ 1988).

Later hydraulic test data does not necessarily support this assessment. However, the variable nature of the rock mass on a small scale has a strong influence on packer test data in particular (Kingett Mitchell 2005a, Appendix 4; Golder 2009). The results of packer tests performed in the area of the planned TTTSF (Golder 2011a) and the Back Road WRS (EGL 2010) indicate the permeability of the rock mass across the MGP does not differ substantially across the site (Appendix B).

The hydraulic conductivity applied to previous MODFLOW models of the MGP site has been anisotropic, with a higher value applied in the north-south direction than in the east-west direction (refer Section 6.3). This anisotropy has been applied to simulate the presence of minor faults and near vertical fractures aligned approximately north-south across the site as well as to place an emphasis on the low dip of the schist foliations toward the east. The eastward dip of the foliation in the Back Road WRS area is similar to that across the remainder of the TTTSF site (EGL 2010). Minor north-south trending faults and fractures have been mapped in the Back Road and TTTSF areas (EGL 2010, Golder 2011d,) and it is expected that these features will prove to be ubiquitous across the site.

The hydraulic testing performed to date at the MGP site indicates that the values for rock mass permeability previously applied to the calibrated groundwater model remain reasonable. As such, the values for hydraulic conductivity applied to the previous model have been retained for the current modelling project (refer Section 6.3).

3.0 MINING RELATED STRUCTURES

3.1 Tailings Storage Facilities

3.1.1 Embankment design and hydraulic performance.

Each of the embankments constructed at the MGP incorporate:

- An embankment body constructed from waste rock.
- A drainage system to ensure the embankment body is maintained in an almost completely unsaturated state for stability purposes.
- A low permeability Zone A which forms the upstream face of the embankment and limits the rate of seepage into the embankment body.
- Upstream embankment raises constructed on top of stored tailings material, each of which has its own Zone A and drainage system (Figure 5). For the lifts between 533 and 545 mRL, the A zone was not incorporated within the upstream embankment raises.

The drainage systems installed in both the MTI and the SPI consist of:

- Drains installed in the main embankment, including chimney drains and basal collector drains.
- Drains installed on the floor of the impoundment prior to deposition of the tailings, termed underdrains. The underdrains combined with the drains in the main embankment are collectively referred to as the key drains.
- Mattress and collector drains constructed in the upstream embankment raises constructed on top of stored tailings material.

Geotechnical monitoring of pore water pressures within the main embankments has demonstrated that the embankments have essentially remained unsaturated to date. This status reflects the efficiency of the combined Zone A and chimney drain system at limiting seepage losses into the embankments.

3.1.2 Drainage system records

Drain discharge records for the MTI (Golder 2011c) indicate that:

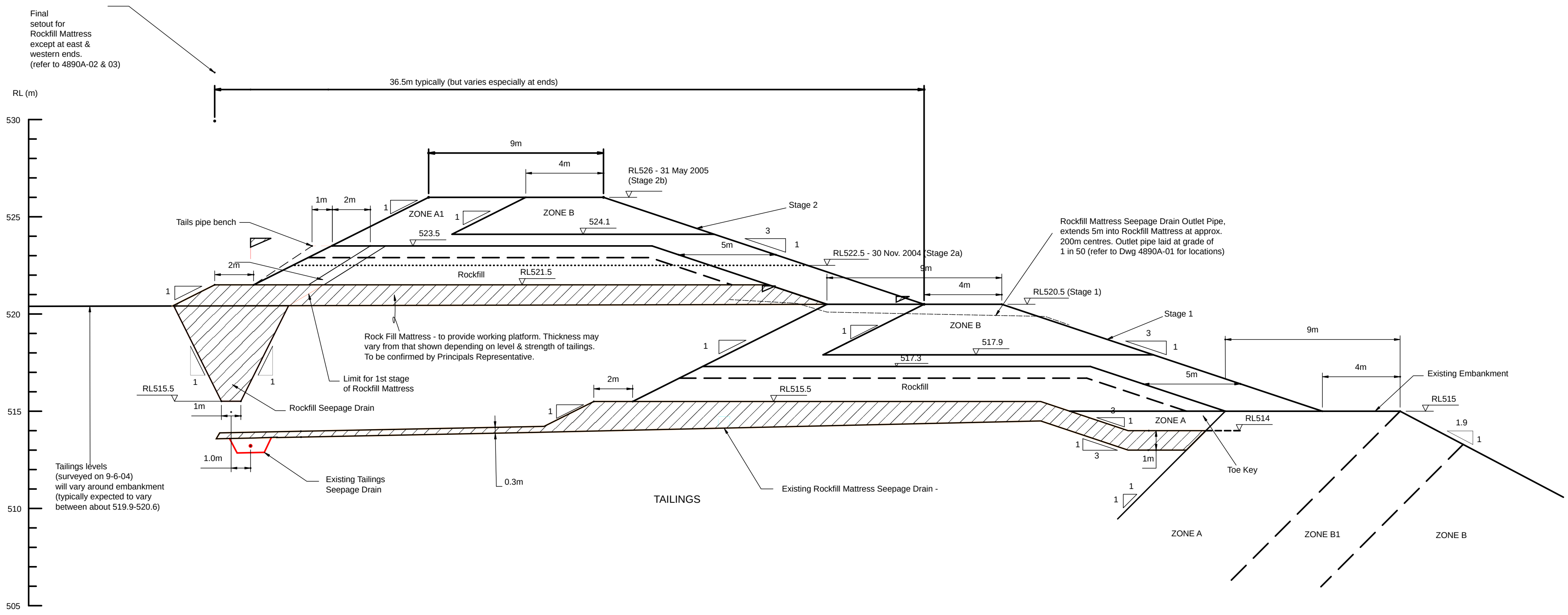
- Discharge flows from the drainage systems in the main embankment and the underdrains have typically been within a range of 700 m^3/day to 1,000 m^3/day since 2005.
- Discharge flows through the drainage systems in the upstream embankment raises have varied greatly depending on the activity status of the TSF. The flow range has been from being essentially dry through to 1,000 m^3/day.
- The discharge flow responses recorded from the upper drainage system are much stronger than those recorded from the key drains during corresponding periods, apparently due to their shallower depth of burial. The shallower drainage systems installed in the upstream raises of the MTI embankment appear to be limiting downward pore water seepage in areas close to the embankment.

Drain discharge records for the SPI (Golder 2011c) indicate that:

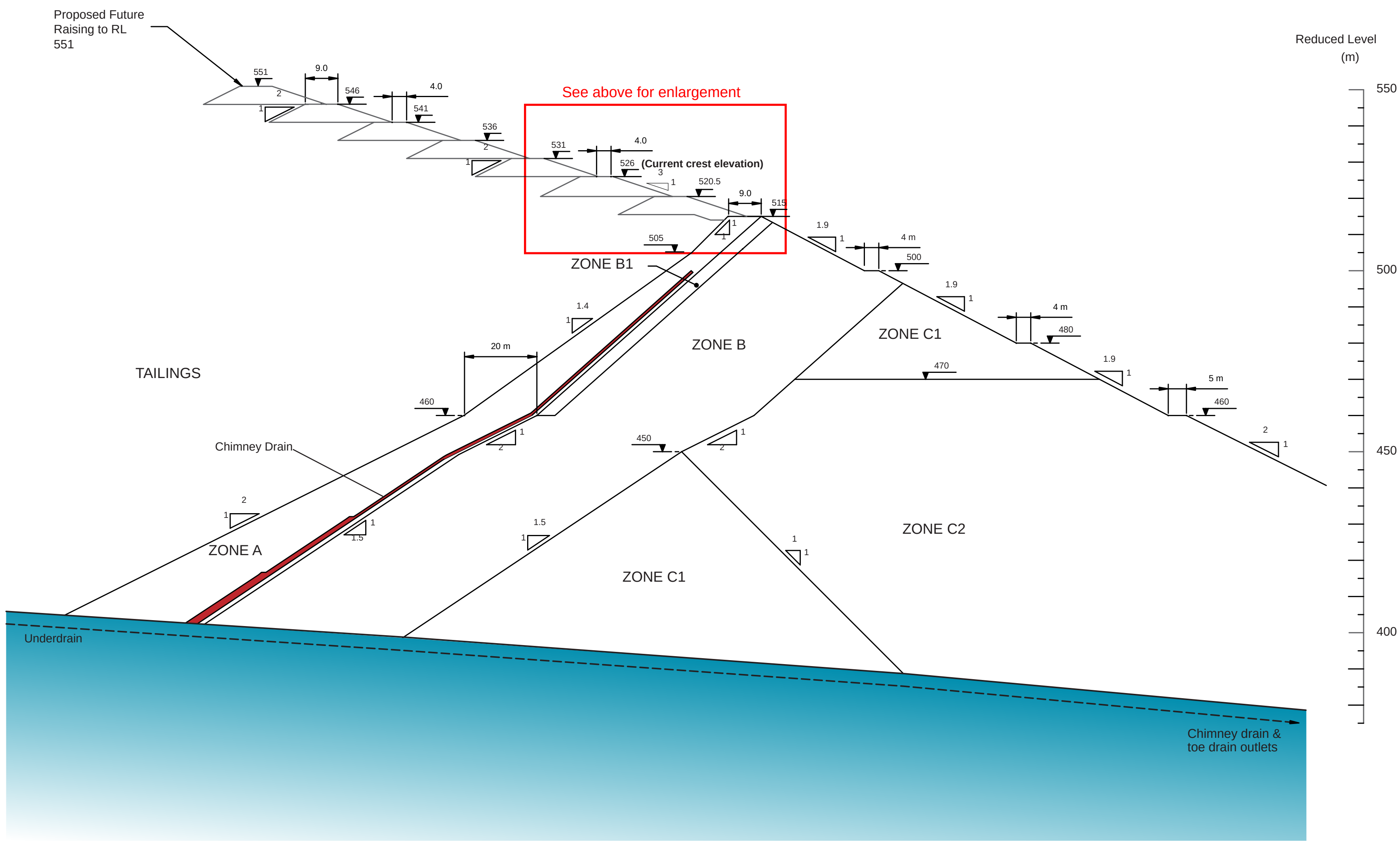
- Total discharge flows from the drainage systems in the main embankments and the underdrains have typically been within a range of 200 m^3/day to 1,500 m^3/day.
- Discharge flows through the drainage systems in the upstream embankment raises have also varied greatly depending on the activity status of the TSF. The flow range has been from being essentially dry through to 600 m^3/day.

3.2 Waste Rock Stacks

Within the Deepdell Creek catchment WRS's have been constructed to cover almost all of the original catchment area of Northern Gully and a small area of the MTG above the Lone Pine water supply impoundment (Figure 2). Round Hill Pit has essentially been entirely backfilled with waste rock, as has the base of Golden Point Pit. To the north of Deepdell Creek, the Deepdell WRS was constructed to store waste derived from the Deepdell North and Deepdell South pits. Waste rock from Deepdell South has also been used to backfill Deepdell North pit.



TYPICAL CROSS SECTION OF MIXED TAILS EMBANKMENT UPHILL RAISES



Note: Drainage details are not shown on this section

1. Base drawing produced and supplied by Engineering Geology Limited, 2005. Schematic only, not to be interpreted as an engineering design or construction drawing

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The Frasers West WRS has been constructed to the west of Frasers Pit, straddling the catchment divide between the NBWR and Murphys Creek. During the past two years construction work has also been initiated at the Frasers East WRS which is planned to fill much of the remaining NBWR catchment upstream from Frasers Pit.

Waste rock is stored in Innes Mills Pit, forming the southern boundary of the Southern Pit Tailings Impoundment (SPI) and also supporting the realigned Macraes Dunback road. In addition, the southern limit of the Northern Gully waste rock stack forms the catchment boundary between the NBWR and Deepdell Creek.

OceanaGold plans to complete construction of the Frasers East WRS to a larger size than has been previously consented, together with the Frasers South WRS. Waste rock storage operations are then planned to shift to the Back Road WRS. A relatively small WRS is also planned as backfill for the southern section of Frasers Pit.

Placement of most waste rock stacks has been by end dumping in 10 – 20 m lifts. The waste rock is generally poorly sorted with particle sizes ranging from clay to boulders. Physical sorting of the material during the dumping process, together with physical abrasion from vehicular traffic, is likely to produce a particle size grading in each lift, with the coarsest material near the base of the lift. Compaction of the upper material due to vehicle movement across the lift results in a low permeability cap at the surface of each lift.

Where waste rock disposal in an opencast pit is based on the creation of a single large lift material sorting would tend to be more pronounced and the generation of low permeability layers within the dump restricted to the top of the dump.

The low permeability cap generated by traffic across a WRS prior to closure combined with the reinstatement of a soil horizon during rehabilitation is expected to result in infiltration rates similar to the background regional rates of rainwater infiltration to groundwater. To date very little seepage has been observed discharging from the toes of the WRS's at the MGP. The underdrains installed beneath the Northern Gully WRS also have very low discharge rates (D. Clarke, OceanaGold, pers comm.). These observations tend to support the expectation of low infiltration rates.

The sheer size of a WRS results in a considerable delay between its construction and the development of a groundwater system within the base of the stack. It may require decades for the groundwater within a WRS to develop to the long term steady state flow system.

3.3 Silt Ponds

Several silt ponds have been constructed for environmental protection purposes at the MGP site (Figure 2). They mainly take the form of small dams across gullies with ephemeral surface water discharges.

In most cases the water flows discharging to silt ponds are dominated by short term surface water flows supported by shallow groundwater flows. Most of the silt ponds dry out periodically due to the lack of a deeper groundwater base flow entering the gully upstream from the pond.

The Maori Tommy Gully, Northern Gully, Murphys Creek and Deepdell North silt ponds may have permanent standing due to the presence of waste rock stacks or a TSF in the gullies upstream from these silt ponds. Although the seepage discharges from the WRS's are small, these discharges could support a silt pond through the summer period.

Maori Tommy Gully

The silt dam across MTG is located a short distance below the toe of the MTI. The tailings impoundment has restricted the catchment area for the silt pond to a fraction of the size of the former MTG catchment and there is effectively no surface flow in MTG below the silt pond.

Northern Gully

The silt dam across Northern Gully captures most of the surface runoff from the Northern Gully waste rock stacks and a large fraction of the groundwater seeping through these stacks. Underdrains installed in the valley inverts beneath the Northern Gully waste rock stack contribute to the base flow component of the flow at the silt dam.

The extent of the surface catchment for this silt pond has changed over time due to the disposal of waste rock across most of the catchment area. As the crests of the waste rock stacks exceed the pre-existing catchment boundary elevations, these boundaries have consequently shifted and the catchment area for the Northern Gully silt dam has decreased.

Murphys Creek

Practically the entire Murphys Creek catchment upstream from the silt pond has been taken up by the Frasers West WRS. As the WRS has expanded over time the balance of water flowing into the silt dam has shifted from natural run-off to run-off from the WRS combined with groundwater seepage from the WRS. The quality of water in Murphys Creek silt pond is declining in response to the establishment of the WRS (Golder 2011e).

Deepdell North

Deepdell North silt pond is located downstream from the former Deepdell North pit. The pit has been backfilled and rehabilitated. It is not clear whether the declining water quality in the silt pond (Golder 2010e) is a consequence of surface run-off or whether waste rock in the former pit has become saturated and is now overflowing into the gully. Based on low WRS discharges observed elsewhere at the site, simple infiltration of rainwater through the WRS to the pit is unlikely to have resulted in the pit filling up in the period since the WRS was established.

Back Road WRS silt dams

OceanaGold plans to establish silt dams or smaller water retention structures in the gullies downstream from the Back Road WRS. The deeper sections of these gullies may contain almost permanent flows supported by groundwater discharges. The construction of a WRS upstream is unlikely to have a major effect on these flows over the short period. Over the longer term the flows at these structures may increase slightly due to the removal of evaporative losses from the upstream gully sections due to infilling with waste rock.

3.4 Underground Workings

Frasers Underground

The Frasers Underground workings have not been incorporated in this groundwater assessment. These workings are located at depth and are not considered to have a significant effect on the overall groundwater flow system at the site.

Although ongoing operations at Frasers Underground are expected to lead to significant caving of overburden and dewatering of the rock mass above the workings, this is not considered to have a substantial effect on the long term water balance for the site. Much of the Frasers Underground is to be covered by the Frasers East WRS. Following closure of the MGP any future groundwater flows into Frasers Underground will eventually discharge to the Frasers pit lake.

Golden Point historical underground workings

The Golden Point historical underground workings form a hydraulic connection between Golden Point and Round Hill pits and Deepdell Creek. This connection is not active when the water level in Golden Point pit is at an elevation below about 340 mRL. This would be the case for the entire planned operational period of the mine. Following closure of the mine, water levels in the pit are expected to rise. The potential for seepage losses through the underground workings if pit lake levels rise above 340 mRL, have been independently assessed and documented in a separate report (Golder 2011b).

4.0 WATER QUALITY

4.1 Introduction

Process water from the ore processing plant is to be mixed with mine tailings and deposited sub-aerially as a slurry on the surface of the TSF's. During operations there is a permanent standing pool of water covering part of the tailings surface, which is recycled to the plant or lost through evaporation or seepage into the tailings. This standing pool of water is referred to as the decant pond. Samples of decant water and water discharging from the TSF embankment drainage systems are collected and analysed on a regular basis. The MGP water management system includes recycling of as much decant water as possible to the ore processing plant.

As the tailings solids deposit out of the slurry in the TSF, decant water keeps the tailings deposit fully saturated as long as deposition continues. Monitoring of pore water pressures within tailings stored in the existing TSF's indicates there is generally a downward hydraulic gradient through the tailings mass (Kingett Mitchell 2005a).

As the decant water seeps through the stored tailings, this water interacts with tailings material. The pore water quality is altered due to dissolution and precipitation reactions that occur within the tailings mass. In addition, oxygen in the pore water reacts with the tailings minerals. As parts of the tailings deposit drain and consolidate between episodes of deposition, the tailings become unsaturated, allowing more oxygen to penetrate the deposit to react with tailings minerals. Consequently, the geochemical conditions in the tailings mass become more reducing with depth. These interactions and changes mean that the seepage water quality at the base of the tailings mass differs from the decant water quality.

Groundwater monitoring wells have been installed around each of the existing tailings storage areas. Specifically, the water quality in the groundwater system down-gradient from the MTI is intensively monitored. Some of the ions present in tailings seepage water are considered to be conservatively transported. Other ions may be delayed by being precipitated, adsorbed or broken down during their transport through the groundwater system at the MGP. The observed breakthrough curves at monitoring wells have been evaluated for a range of parameters. This analysis can support an assessment of the degree to which contaminant loads are reduced within the groundwater system prior to discharge to receiving surface waters at greater distance from the tailings storage than the monitor bores.

4.2 Tailings Decant Water

4.2.1 Introduction

Tailings decant water samples have been obtained and analysed on a regular basis from each of the tailings storage facilities since operations at the MGP began in 1991. Tailings decant water quality is influenced by a number of factors, including:

- The geochemical composition of the ore being processed in the plant

- The processing conditions in the plant
- Tailings deposition schedules for the MTI and the SPI
- Environmental factors, such as concentration through evaporation or dilution through precipitation
- Geochemical processes occurring in the plant and decant pond

Over the past 20 years, a number of operational changes have occurred that have influenced the quality of the decant pond water. These changes include:

- The introduction of a pressure oxidation stage to ore processing at the plant in 1999
- An increase to full plant capacity in 2006
- The introduction of Reefton ore to the plant in 2007
- Ongoing optimisation of the gold extraction processes

Prior to 1993 two tailings storage areas were operational. These areas consisted of:

- The Concentrate Tailings Impoundment (CTI), where tailings from the concentrate process stream were stored
- The larger Flotation Tailings Impoundment (FTI), where tailings from the flotation process stream were stored

Flotation tailings are produced through the initial separation of high gold content minerals from the low gold content ore by a flotation process. Low gold ore is processed through a froth flotation cycle and the resulting waste material is referred to as flotation tailings. High gold concentrate produced from the flotation circuit is processed by pressure oxidation and multiple cyanide leaches, with the resulting waste product referred to as concentrate tailings.

Following 1993 the two tailings streams were combined. Since 1993 they have been mixed with a short period of separation in 1998/99, immediately prior to implementation of pressure oxidation. Since then the mixed tailings have been stored in the MTI and the SPI (Table 1). Tailings stored in the CTI have been excavated and processed. The storage space that was made available has since been incorporated in the MTI.

Table 1: MGP tailings deposition schedule.

Period start	Period end	Active TSF
10 February 1992	1993	FTI, CTI
1993	7 February 2002	MTI
7 February 2002	27 May 2003	SP10
27 May 2003	18 May 2004	MTI
18 May 2004	25 November 2004	SP10
25 November 2004	22 March 2006	MTI
22 March 2006	13 December 2007	SP11
13 December 2007	20 May 2009	MTI
20 May 2009	13 February 2010	SP11
13 February 2010	Present (December 2010)	MTI

4.2.2 Flotation Tailings Impoundment

Prior to 1999, the quality of the FTI decant water (Golder 2011e), was characterised by:

- Sulphate concentrations below 1,800 g/m^3. Due to the limited oxidation of sulphides during the ore treatment process
- The addition of lime to maintain a high pH in the tailings water, resulting in a pH of above 8.0. Calcium concentrations in the FTI decant water were generally below 150 g/m^3
- Highly variable weak acid dissociable cyanide (cyanide$_{WAD}$) concentrations, although for much of this period the concentrations detected in decant water were between 0.05 g/m^3 and 2 g/m^3
- High total arsenic concentrations, generally within the range of 2 g/m^3 to 20 g/m^3.
- Total iron concentrations were generally below 10 g/m^3

The water quality associated with the FTI decant ponds is primarily relevant to the models simulating the early period in the mine life. It is not expected that this quality of decant water will appear in future decant water at the site.

4.2.3 Mixed and Southern Pit tailings impoundments

The CTI was used for the storage of concentrate tailings until 1999 although deposition ceased in 1993, with a short period of deposition recommencing prior to the introduction of pressure oxidation in 1999. The concentrates were subsequently removed for reprocessing and this impoundment was incorporated in the MTI. The pressure oxidation process introduced in 1999 forced the oxidation of pyrite in the processed ore and increased the production of sulphuric acid. Lime was no longer added to the decant water ponds. There was consequently an almost immediate change in decant water quality in the MTI. These changes include:

- The pH decreased to generally between 6 and 8
- Sulphate concentrations increased to between 3,000 g/m^3 and 5,000 g/m^3
- Although lime was no longer added to the ponds, calcium concentrations in the decant water increased greatly, to generally come within the range of 400 g/m^3 to 700 g/m^3
- Total iron concentrations increased slightly although generally remaining below 20 g/m^3
- Total arsenic concentrations in the MTI decreased by approximately two orders of magnitude and were often below the analysis detection limit
- Cyanide$_{WAD}$ concentrations became less variable and increased to a range of generally between 0.2 and 5 g/m^3

The increase to full plant capacity in 2006 and the subsequent introduction of Reefton tailings to the process stream was followed by further optimisation of the pressure oxidation process. These changes have been reflected in changes to the decant water quality during this period (Golder 2011e), including:

- A decrease in the pH of tailings decant water to generally fall within the range from 3.5 to 6.5
- An increase in sulphate concentrations, with analysis results of greater than 7,000 g/m^3 being recorded during the past year
- An increase in both the variability and the maximum concentrations for total iron, with results of over 1,200 g/m^3 recorded since 2006

- An increase in total arsenic concentrations, with concentrations of over 10 g/m^3 recorded since 2006
- A decrease in cyanide$_{WAD}$ concentrations to a range of generally between 0.05 and 1 g/m^3

During the period since 2006 there appears to have been a clear difference in decant water quality between the SP11 and the MTI. This difference is evident through:

- Generally higher pH in the MTI decant pond
- Greater variability and lower sulphate and calcium concentrations in the MTI decant water compared to the SPI decant water

These differences are likely to result from differences in the management of the two tailings impoundments, including:

- Shifting of tailings deposition between the two impoundments
- Dilution and possibly geochemical changes in the MTI decant pond during the non-depositional periods
- Continuous cycling of decant water through the SPI decant pond (J. Yeats, Engineering Geology Ltd, pers comm.) appears to have reduced water quality variability in the SP11 decant water
- Some selective deposition of tailings from Reefton ore in SP11
- Addition of freshwater to the MTI as part of site water management controls and use for dust suppression during non-depositional periods.

OceanaGold have indicated that the process plant is now operating at maximum capacity and under optimal oxidation conditions. OceanaGold expects that the decant water quality should stabilise in the near future if it has not already done so.

4.2.4 Groundwater model decant water quality

Assessment of the geochemistry of tailings from MGP and RGP ore indicates the differences in decant water quality generated by processing the two tailings are not likely to be substantial (Golder 2010b). Tailings decant water quality is more strongly controlled by the quality of the water used in the process plant than by the nature of the ore being processed at any particular time. This conclusion applies to the operational period following the implementation of the pressure oxidation stage in the process plant, with the decant water quality prior to that time being substantially different.

Tailings decant water quality data from the past 10 years of operation, since the implementation of pressure oxidation at the MGP process plant, is considered to be the most appropriate dataset to use to derive the projected decant water quality for TTTSF groundwater modelling purposes. For modelling purposes the 90th percentile of the water quality analysis results from the combined MTI and SPI drainage systems for the period from 2000 to 2010 have been applied as tailings decant water quality (Table 2). The exception is cyanide$_{WAD}$. The use of cyanide at the plant has decreased as a result of plant optimisation. During the past two years cyanide$_{WAD}$ concentrations in the decant ponds have been very low compared to the concentrations detected in earlier samples. Consequently, the concentration applied in the groundwater model is the 90th percentile of the water quality analysis results for cyanide$_{WAD}$ since 2008 (Table 2).

The mine water management modelling for the site is expected to generate decant water quality projections that differ in detail from the groundwater model input values presented here. The values applied to the water quality parameters modelled for groundwater transport are considered to be conservative for use in generating contaminant transport projections.

4.3 Tailings Seepage Water

4.3.1 Introduction

Tailings seepage water samples have been obtained and analysed on a regular basis from each of the tailings storage facilities since operations at the MGP began. The quality of seepage water discharging from the base of stored tailings is influenced by numerous factors including:

- The decant water quality
- Time required for pore water to seep through the tailings mass
- The geochemistry of the tailings mass
- Adsorptive and desorptive reactions between the pore water and the tailings mass
- Dissolution and precipitation reactions within the pore water
- Changes in the redox environment within the tailings mass over time

Table 2: Tailings decant water summary statistics – 2000 to 2010.

Parameter (1)	FTI and MTI		SP10 and SP11		Combined TSF's
	Median	90th Percentile	Median	90th Percentile	Mean of 90th percentiles
Sodium	430	580	470	590	585
Potassium	92	130	90	120	125
Calcium	570	680	560	680	680
Magnesium	310	430	330	420	420
Chloride	29	62	29	46	54
Sulphate	3,800	5,500	4,200	5,800	5,650
Arsenic	0.7	2.4	0.8	4.5	3.4
Copper	0.52	0.99	0.26	0.29	0.64
Iron	39	520	100	660	590
Lead	0.02	0.014	0.004	0.006	0.01
Zinc	0.028	0.062	-	<0.001	0.035
$Cyanide_{WAD}$	0.7	0.85	0.10	0.23	0.47 (2)

Notes: 1) All units in g/m^3. Values rounded to two significant figures.
2) $Cyanide_{WAD}$ concentrations in the decant water based on the 90th percentile of decant water quality data from the past two years due to decreased concentrations resulting from optimisation of the process plant.

An assessment of the tailings buffering capacity indicated that the tailings have a considerable acid neutralising capacity. This acid neutralising capacity is likely to be predominately in the form of calcite. It is therefore expected that acidic drainage conditions will not develop in the future within the tailings mass to be stored at the TTTSF (Golder 2010b).

4.3.2 Flotation Tailings Impoundment drains

Prior to 1999 the quality of the water discharging from the FTI drainage systems (Golder 2011e) was characterised by:

- Stable pH, typically between 6.5 and 7.0
- Sulphate concentrations that increased over time. In 1992 concentrations were below 100 g/m^3. These concentrations had increased to between 1,000 g/m^3 and 1,300 g/m^3 by 1999
- Calcium concentrations that had stabilised at about 80 g/m^3 by 1999
- Total arsenic concentrations that were in the range of 2.0 g/m^3 to 3.0 g/m^3 from 1997 to 2002
- Cyanide_{WAD} concentrations that increased from below 0.75 g/m^3 in 1990 to between 0.70 g/m^3 and 1.5 g/m^3 in 1999

4.3.3 Mixed Tailings Impoundment drains

The pH of the underdrain water has remained relatively stable between 6.5 and 7.0. This stability is likely to be due to the acid neutralising capacity of the tailings mass.

Sulphate concentrations in the MTI drainage water have increased over time from between 1,000 g/m^3 and 1,300 g/m^3 in 1999. During 2010 sulphate concentrations in the drainage water have been consistently between 1,500 g/m^3 and 3,000 g/m^3.

Calcium concentrations in the MTI seepage water have increased since monitoring began. Prior to 2001 the observed concentrations appeared to reflect decant water quality of that period. Subsequently, there has been a greater rate of increase with calcium concentrations reaching 450 g/m^3 during 2010. These concentrations remain well below those detected in the decant water since 2001.

Total iron concentrations in the MTI drainage water have been relatively stable, ranging from 2.5 g/m^3 to 10 g/m^3 during the entire monitoring period.

Total arsenic concentrations in the MTI chimney drainage water have increased since 2002 to between 3.2 g/m^3 and 5.0 g/m^3. In comparison, the total arsenic concentrations detected in the MTI underdrain discharges were generally below the detection limit of 0.1 g/m^3 or 0.4 g/m^3 from 1992 to 2006. Since 2006 the typical concentrations detected increased slightly until 2008 and subsequently declined again. During 2010 the concentrations detected have been consistently below 1 g/m^3.

Since the start of 2008 cyanide_{WAD} concentrations detected in discharge water from the MTI chimney drains have consistently been less than 0.6 g/m^3.

4.3.4 Southern Pit Impoundment

Water discharging from drains installed beneath the tailings mass in the SP10 and SP11 has been sampled since February 2002. The analysis results are considered to be representative of seepage water that would be generated from new tailings deposited without specific leachate mitigation measures. OceanaGold have indicated that there has been some selective deposition of Reefton tailings in the SP11 (J. Bywater, pers comm.). The degree to which this selective deposition influences the seepage water quality is difficult to quantify.

In comparison to the MTI, the Southern Pit impoundments have been filled to the current tailings level over a relatively short period of time. This implies that the pore water initially incorporated in the tailings was representative of tailings decant water for the period since 2002. Much of the seepage water data from the MTI relates to tailings deposited prior to 2002, with delayed influence from newer tailings material.

The pH of the SPI seepage water has remained between 6.5 and 7.0. This stability contrasts with the SPI tailings decant water pH which has decreased from approximately 8 to approximately 4 over time (refer Section 4.2).

Sulphate concentrations in the SPI seepage water have generally been within the range of 3,000 g/m^3 to 4,000 g/m^3. These concentrations approximately reflect those from the SP10 decant water, although they are considerable less than those observed in the SP11 decant water.

Calcium concentrations in SPI seepage water have been typically in the range 400 to 600 g/m^3. This range is similar to that observed for the tailings decant water.

Total iron concentrations measured in the SPI underdrain discharge water have been higher and more variable than measured in the MTI drainage water. The SP10 outlet drain total iron concentrations have typically ranged from 14 g/m^3 to 35 g/m^3, and total iron concentrations in the SP11 underdrain were higher again, ranging from 40 g/m^3 to 70 g/m^3.

In 2006, total arsenic concentrations measured in the SPI underdrain discharge water were similar to those measured from the MTI drains. However, since that time total measured arsenic concentrations in SPI drains have increased to a range of between 12 g/m^3 and 15 g/m^3 in 2009.

The cyanide$_{WAD}$ concentrations in the SPI drain discharges have ranged from <0.0010 g/m^3 to 0.74 g/m^3, with the concentrations detected since early 2008 being typically less than 0.25 g/m^3.

4.3.5 Groundwater model tailings seepage water quality

Tailings pore water projections for the TSF's have been based on the analysis of MTI drainage discharges from the period 2000 through to 2010 (Table 3). The 90th percentile value from this data set has been applied as a conservative indicator for projected seepage water quality from the TSF's during the operational period of these facilities.

Table 3: Projected TSF pore water quality.

Parameter (1)	Tailings pore water quality		Waste rock leachate quality (2)
	Operational (3)	Post-closure (4)	
Sodium	498	416	62
Potassium	46	17	13
Calcium	411	410	470
Magnesium	245	200	390
Chloride	107	111	11
Sulphate	2,769	2,260	2,500
Arsenic	5.38	1	0.007
Copper	0.02	0.02	0.0027
Iron	31	21	1
Lead	0.01	0.013	0.00021
Zinc	0.02	0.009	0.035
Cyanide$_{WAD}$	0.47 (5)	0.35	0

Notes:
1) All units in g/m^3.
2) Based on water quality from Northern Gully WRS underdrains.
3) Based on the 90th percentile of the average of MTI drainage water quality since 2000.
4) Based on the 90th percentile of the water quality from the MTI underdrains since 2000.
5) Cyanide$_{WAD}$ concentrations based on the 90th percentile of decant water data from the past two years due to decreased concentrations resulting from optimisation of the process plant.

Following closure of the TSF it is expected that a gradual change in seepage water quality will occur over time. This change would reflect a shift in recharge water quality to the top of the tailings mass from process water to rainwater. During the operational period of the TSF the seepage water quality is expected to be controlled to a large extent by the quality of the water discharging from the slurry pipeline to the impoundment. Following TSF closure the groundwater level in the tailings mass would decline due to reduced recharge, thereby reducing the volume of water left over from the operational period. At the same time infiltrating rainwater would start to replace the existing pore water.

As the tailings decent water is oversaturated in sulphate, gypsum and potentially other salts are being precipitated and settle with the tailings solids during the operational period of the TSF. Infiltrating rainwater would leach these salts for a considerable period following TSF closure. Eventually these soluble salts would be leached away and contaminant concentrations in the drain discharges would decline to the level indicated by leach tests of tailings samples. These long term leachate concentrations are expected to be considerably lower than those indicated from analysis of the drainage records from most of the MTI and SPI drain discharges.

The concentrations applied in modelling the tailings pore water quality have been based on the 90th percentiles of water quality data from MTI and SPI drain water quality datasets for the period from 2000 through to 2010 (Table 3). The exception is the water quality value for cyanide, which has been reduced due to the lower concentrations observed in TSF decant water over the past two years. The long term seepage water quality is based on the 90th percentile of water quality data for the main MTI underdrain discharges (Table 3).

4.4 Waste Rock Seepage

The leaching of stored waste rock occurs under more oxidising conditions than those present in stored tailings. As such the leachate water quality from WRS's differs from that expected to apply to long term tailings leaching. Discharge water from underdrains installed beneath the Northern Gully WRS has occasionally been collected and analysed. Available information (Golder 2011e) indicates the seepage water is likely to be characterised by:

- Sulphate concentrations at approximately 2,500 g/m^3
- Calcium concentrations up to 500 g/m^3
- Arsenic concentrations at or below 0.005 g/m^3
- Negligible cyanide$_{WAD}$ concentrations

Although the data set is small, the 90th percentile of these results is considered to be reasonably indicative of the water quality that may be expected in seepage water from WRS's at the MGP into the future (Table 3). These concentrations have been applied in the groundwater model to represent seepage water quality from the waste rock stacks and the exposed TSF embankment area.

4.5 Contaminant Attenuation in Groundwater

4.5.1 Introduction

Conceptually, dilution of contaminants within the groundwater system affects all contaminants equally, provided the source concentrations exceed the background water concentrations. Conservative transport of contaminants implies the relative decrease in concentration with distance from the source is the same for all parameters. If the decrease is greater, other factors may be acting to remove the contaminant from the groundwater. If the decrease is less, the background concentrations in the groundwater may be similar to those at the contaminant source. Changes in groundwater quality may also result in a specific contaminant being taken into solution at an increased rate from the surrounding rock mass.

The water quality monitoring data from the MTI seepage drains and the MTI detection wells in MTG indicates that there is a considerable decrease in the non-conservative parameters occurring. In order to assess the relative importance of dilution and active removal processes, the decrease in concentration between the seepage water and the detection wells has been calculated for the MTI (Table 4).

Table 4: Concentration reduction between seepage and groundwater wells.

	MTI seepage concentration (1) (g/m^3)	Detection well concentration (2) (g/m^3)	Concentration reduction factor (3)
Conservative Parameters			
Sodium	530	110	4.8
Potassium	37	5.1	7.3
Chloride	102	25	4.1
Non-conservative parameters			
Arsenic	3.9	0.019	205
Iron	8.7	7.8	1.1
$Cyanide_{WAD}$	0.85	0.040	21

Notes: 1) Mean concentration in Sump B_CDE and Sump B_CDW from 2007 to 2009.
2) Mean concentration in detection wells GW46 to GW51 from 2007 to 2009.
3) Concentration reduction factor = MTI seepage concentration / Groundwater well concentration.

Sodium, potassium and chloride, which are considered to be conservatively transported contaminants, are reduced in concentration by a factor of between 4.1 and 7.3 during transport through the area up-gradient from the detection wells. This indicates that dilution between the seepage from the impoundment and the groundwater wells is responsible for an approximate 4 to 7 fold decrease in concentration.

4.5.2 Arsenic

The mean arsenic concentration in the groundwater wells is approximately 200 times lower than in the seepage water. As dilution is estimated to be responsible for a 4 to 7 fold reduction in concentration, it is evident that a considerable mass of arsenic is being removed by active processes in the aquifer. The most likely active removal processes for arsenic are mineral precipitation and adsorption. It is currently unclear which of these processes is primarily responsible for maintaining low arsenic concentrations in the groundwater.

If mineral solubility is the dominant factor then the concentration of arsenic should remain stable in the aquifer provided the hydrochemical conditions (redox and pH) remain stable in the aquifer. If adsorption of arsenic onto the rock mass is the dominant process, there will be a finite number of adsorption sites in the aquifer and there is potential for the arsenic concentrations to increase in the groundwater over time.

Arsenic adsorption by loess and weathered schist has been investigated through laboratory testing of samples of these materials obtained from the MGP site. The adsorption test procedure, the results of the testing and the derivation of adsorption isotherms applied to the mass transport model were documented in a report by Golder (2010a).

As discussed in Section 4.1, the seepage water from the TSF's is generated under reducing conditions. The K_d value and the maximum adsorption capacity of the rock mass has therefore been applied with respect to the arsenic (III) reduced form.

Adsorption of arsenic onto three different modelled materials has been taken into account in the mass transport model through the application of Langmuir isotherms as indicated in Table 5. Specifically, the isotherm applied in the modelling requires values for K_d and the concentration of available adsorption sites

(SP2) within the rock mass. In assigning the SP2 values used for the different model layers, factors considered included the degree of rock weathering and the fracture density (Golder 2010a).

The adsorption characteristics of the embankment rock fill are considered to be similar to those of weathered schist. This estimate is based on the use of compacted weathered rock to construct a low permeability zone within the embankment to reduce seepage losses. In addition, the broken and crushed rock used to construct the main body of the embankment is considered to have a considerably higher concentration of adsorption sites than would apply to the in-situ schist that has been classed as moderately weathered for modelling purposes.

Table 5: Langmuir adsorption input parameters for arsenic (III).

Model Layer	Distribution coefficient (K_d) L/mg	Concentration of adsorption sites (SP2) Kg/Kg
Highly weathered schist	2×10^{-5}	0.82
Moderately weathered schist	2×10^{-5}	0.1645
Slightly weathered schist	0	0
Embankment body	2×10^{-5}	0.82
Unweathered schist	0	0
Tailings	0	0

4.5.3 Iron

Iron concentrations in the MTI detection wells increased rapidly during the first three years of monitoring (Golder 2010e). Subsequently, the concentrations have remained relatively stable. The reduction factor of 1 (refer Section 4.5.1) suggests iron is taken into solution from the surrounding rock mass to a greater extent than may be occurring with the conservatively transported parameters.

The concentration of iron in the aquifer is likely to be dependent on the solubility limits of iron oxyhydroxide minerals such as ferrihydrite. The solubility of these iron minerals is highly dependent on pH and redox conditions in the groundwater. The initial increase in iron concentrations detected down-gradient from the MTI may have been due to a change in these conditions following the construction of that TSF.

4.5.4 Cyanide$_{WAD}$

The mean cyanide$_{WAD}$ concentration in the MTI detection wells is approximately 21 times lower than that in the MTI seepage water. Given that dilution is estimated to be responsible for a 4 to 7 fold reduction in concentration, it is evident that a considerable mass of cyanide$_{WAD}$ is being removed by active processes in the groundwater. These processes could include complexation and precipitation, adsorption and biodegradation. The primary mechanism of cyanide$_{WAD}$ removal is unclear.

The reduction in cyanide$_{WAD}$ concentration within the groundwater system down-gradient from the MTI is approximately 3 to 5 times greater than that for conservatively transported contaminants. As such, modelling of cyanide$_{WAD}$ transport within the groundwater system assuming conservative transport is likely to generate mass load results 3 to 5 times greater than may be expected at site. This factor may increase with increasing flow path length. There is no indication that cyanide$_{WAD}$ from the MTI has reached the down-gradient MTI compliance wells (Golder 2010e). As such, there is insufficient information available to estimate cyanide$_{WAD}$ attenuation rates over greater transport distances.

5.0 CONCEPTUAL FLOW AND MASS TRANSPORT MODEL

The conceptual groundwater model for the MGP site, as presented in Figure 6 and Figure 7 is based on the following site characteristics listed below.

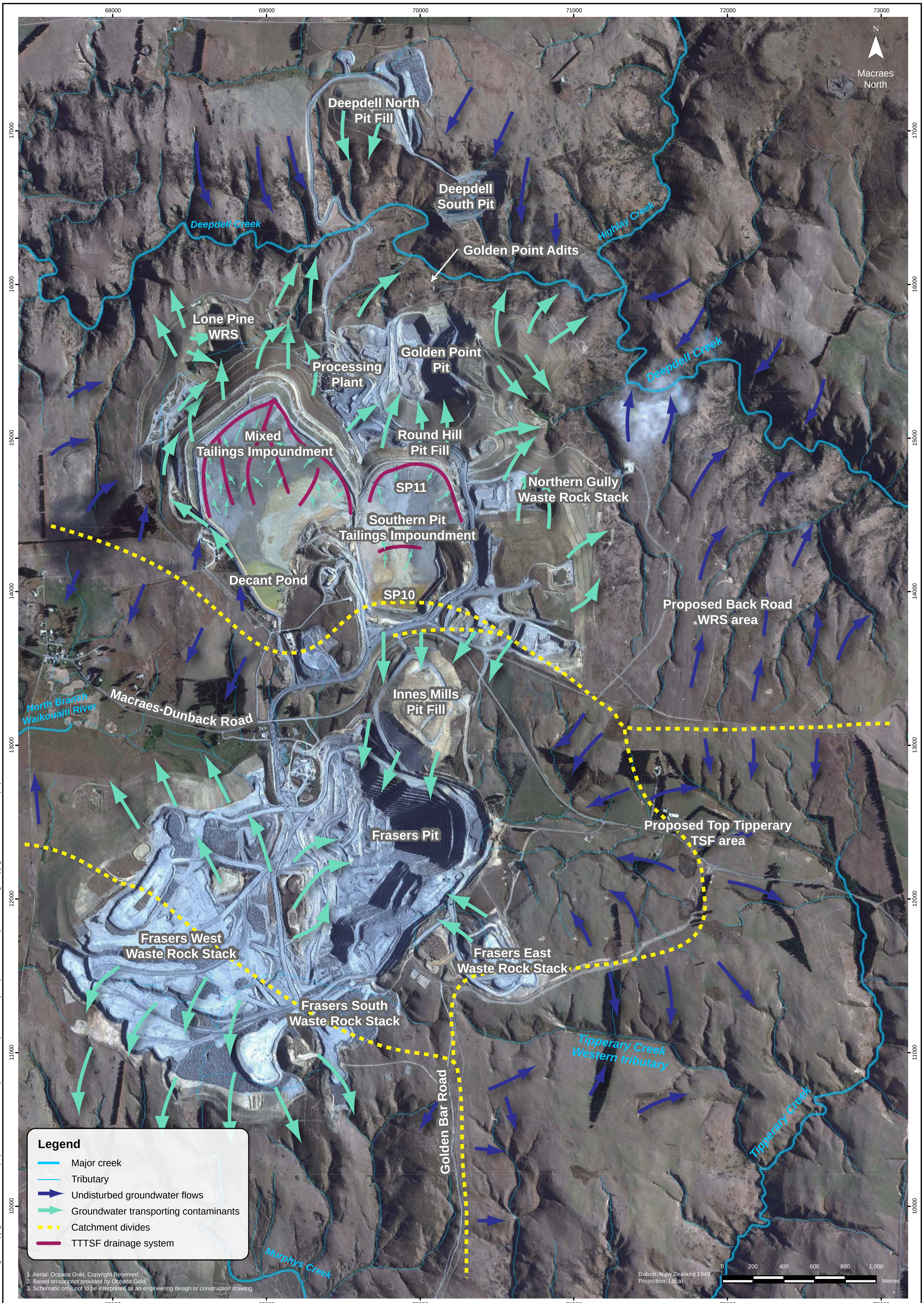
Natural groundwater system

- The schist basement rock has been differentiated into weathering zones on the same basis as applied in previous groundwater modelling work for the MGP site (Kingett Mitchell 2005a). In this modelling approach, each weathering zone has been defined with an approximate thickness as measured from the original topographic surface.
- Hydraulic conductivity and storativity applied to the schist decrease with depth until values consistent with unweathered, unrelaxed schist are reached at a depth of approximately 100 m below the ground surface.
- Based on previous work (Kingett Mitchell 2005a) the rate of recharge to the regional groundwater system has been defined as 32 mm/year.
- Natural drainage channels consist of:
  - Deepdell Creek and its tributaries
  - NBWR and its tributaries
  - Murphys Creek and its tributaries
  - The upper tributaries of Tipperary Creek

TSFs

- Each of the TSFs constructed at the site to date is simulated in the model. The TTTSF is not incorporated in the model. The effects on groundwater and the contaminant losses resulting from the TTTSF are documented in a separate report (Golder 2011a).
- Recharge applied to the TSF's during their operational phases greatly exceeds the regional recharge. The exact value is to be sufficient to ensure the groundwater table within the simulated tailings mass is equal to the top of the tailings mass at the close of the operational period of each TSF.
- Following closure of the MTI and SP10 and the rehabilitation of each, recharge to (and through) the tailings is expected to decrease to the regional background recharge rate. This reduction in recharge reflects:
  - The pumping down of remaining decant water from each of these TSF's at closure of the facility
  - The storage and landscaping of dry tailings on top of the MTI
  - The planned installation of drainage systems at the rear of the MTI to prevent ponding on top of the tailings
  - The reshaping of the tailings mass stored in SP10
  - The planned capping of the tailings with a soil and weathered rock layer suitable for rehabilitation with vegetation similar to the surrounding area

Tailings stored in the SP11 TSF as well as the embankment itself are removed prior to closure of the MGP.



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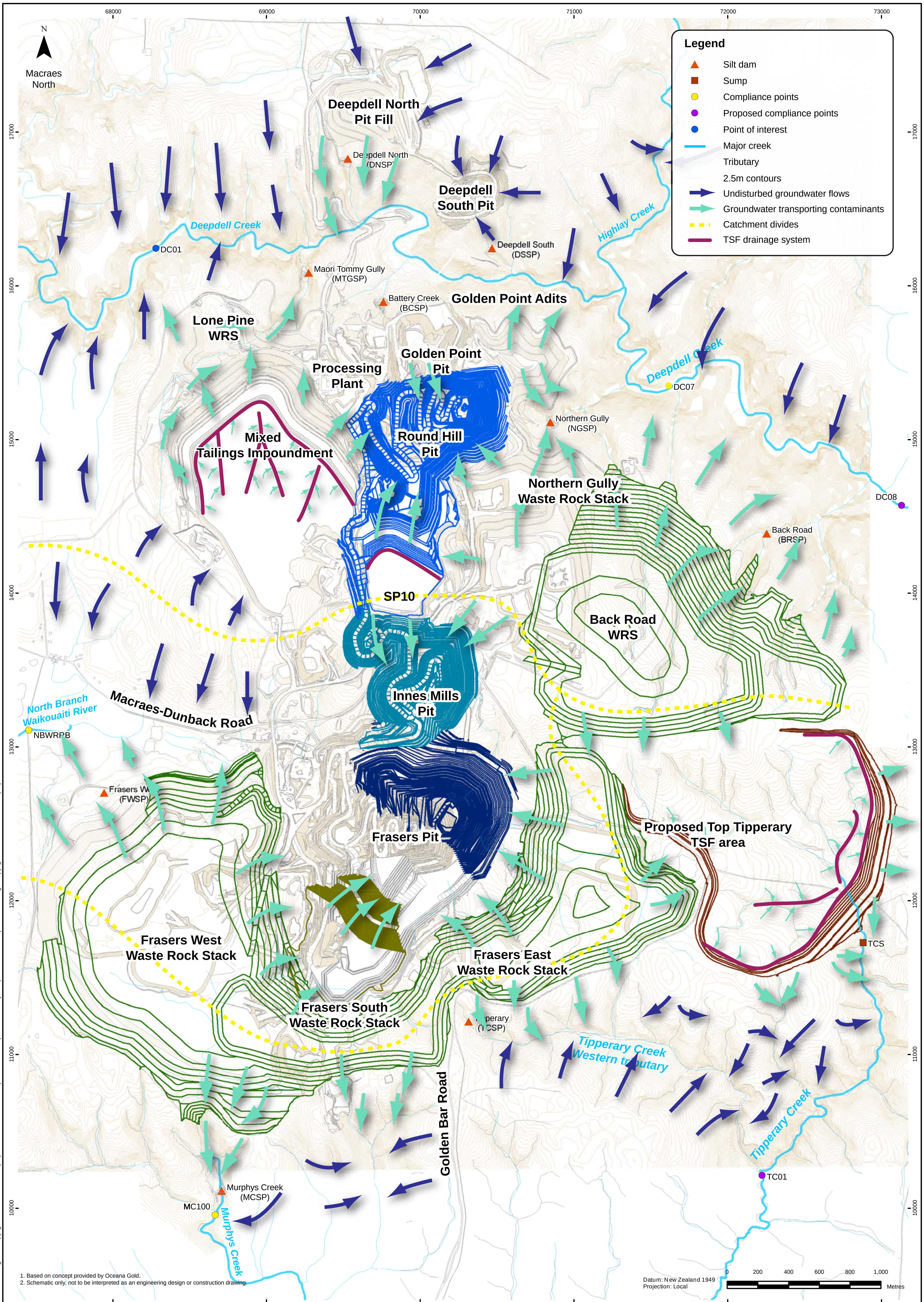

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TITLE CONCEPTUAL GROUNDWATER AND MASS TRANSPORT MODEL – MGP CATCHMENT LAYOUT AT CLOSURE

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Opencast pits

- Each of the opencast pits in the MGP is simulated at two stages. One stage reflects the current layout of the site. The second stage reflects the site layout at closure, including the simulation of backfilled areas.
- In each case the pits that are planned to be operational during Macraes Phase III are simulated as being completely dewatered as they would be under operational conditions.
- The simulation also assumes the pits are essentially dewatered into the long term future, as the water balances and filling rates were not available at the time of modelling. In effect, this means the contaminant mass loads discharging to the pits may be a little overstated over the long term.
- In order to assess pit filling rates for water quality assessment purposes, copies of the final model were used with the drainage elevations in the pits set at specified elevations below and at the overflow elevation for each pit. The groundwater inflows to each pit based on a specific water level in the pit were then calculated and a relationship between inflow and pit lake elevation developed.

TSFs

The simulated TSFs consist of:

- A simplified embankment structure with a hydraulic permeability consistent with waste rock.
- Elevations of stored tailings increasing over the operational life of the TSF's, culminating in the tailings surface reaching the design elevation of 551 mRL. It may be that the storage of dry tailings will occur to a greater elevation in the MTI, however the stacking of dry tailings would not have a major effect on the overall outcomes of the model.
- The permeability of the tailings mass and the waste rock used for construction of the TSF embankments is the same as that previously applied to the calibrated groundwater models of the MTI and SPI.
- The total porosity, effective porosity and specific yield values applied to the tailings mass are within the range of 0.35 to 0.4, reflecting published values for the tailings material.
- Drains built into the embankments to simulate the chimney drains as per previous modelling of the MGP area (Kingett Mitchell 2005a).
- Underdrains installed in gullies and low areas beneath the tailings storage area as per previous modelling of the MGP area (Kingett Mitchell 2005a).
- Following closure, the drainage systems built into the TSF's remain active as the tailings mass becomes progressively unsaturated. The exception is the SP11 drains, which are no longer simulated due to removal of this TSF during the operational period of the mine.

Contaminant transport

The simulated water quality in the model includes the following factors:

- Contaminant mass is introduced to the model through the use of groundwater recharge in the areas of the TSF's and the WRS's. As the volume of water increases within the stored tailings, so too does the accumulated mass of the various contaminants.

- During the operational period of the model, the contaminant characteristics of the recharge applied to the expected area of the decant ponds reflects the expected decant water quality. The contaminant characteristics of the recharge water applied to the remainder of the tailings mass reflect the underdrain seepage water from the existing impoundments.
- On closure of the TSF's, contaminant mass continues to be added to the tailings by way of the recharge water. This addition of mass continues for the entire simulation period of the model. The contaminant characteristics of the recharge water applied to the tailings mass following closure are to reflect a slightly better water quality than that exhibited by most of the existing underdrain discharges (Golder 2011e).
- The Back Road WRS and the Frasers East and Frasers South WRS's are simulated as existing from 2010. Although this does not reflect the actual situation, the difference in terms of contaminant mass introduced to the groundwater system over the term of the model is minor.
- Contaminant concentrations reflecting the quality of the following types of mine water at the site is introduced to the model in the form of concentrations applied to recharge water to the appropriate areas:
  - Tailings pore water, applied to areas of the tailings not considered to be covered by the decant pond
  - Decant water, applied to the tailings mass at the back end of the impoundment
  - Waste rock leachate water, applied to the WRS's and to exposed surfaces of the TSF embankment.
- The groundwater model incorporates an approximate 10 year mine life through to 2020 followed by a post-closure period of approximately 150 years to enable seepage flows to reach the local receiving waters and peak mass loads to these receiving waters to be assessed.

6.0 NUMERICAL FLOW AND MASS TRANSPORT MODEL

6.1 Software

Industry standard groundwater flow and mass transport modelling packages were used for the numerical modelling. The Visual MODFLOW Pro software package was used to construct the groundwater model.

The groundwater flow field in the model and physical flow calibration procedures were calculated using MODFLOW 2000 public domain code from the United States Geological Survey. The mass transport simulation was calculated utilising the MT3D99 code attached to Visual MODFLOW package.

The digital model developed for the simulation of the groundwater flow system and mass transport simulation is documented in Appendix C attached to this report.

6.2 Historical and Projection Simulations

The models documented in this report have been based on and are continuous with a series of models developed by Kingett Mitchell (2005a) for a previous study of the site. The existing model series simulated the groundwater system at the MGP as it changed between 1991 and 2005. From 2005 the models were generating projections for future contaminant discharges based on the final site layout planned at that time.

The groundwater flow patterns in the model series were not continuous between models and were not linked other than being based on similar model layouts and input parameters. In contrast, the mass transport models associated with the flow models were linked, with the final groundwater concentrations developed from one model being used as the starting concentrations for the next. Full documentation of input factors and results for the existing MODFLOW models has not been incorporated in this report, however it is available in the Kingett Mitchell (2005a) report.

The projections generated by Kingett Mitchell model series have remained reasonable and slightly conservative through to the end of 2009 (Appendix D). For this reason, the current model developed to simulate contaminant transport for the coming 10 years of operations and a further 150 year post-closure period has also been linked to the existing models. The contaminant concentrations calculated by the previous model for the end of 2010 have been used as the starting concentrations for the modelled projections documented in this report.

6.3 Flow Model Structure and Parameters Applied

Details of the model grid and layout are presented in Appendix C. The numerical groundwater model developed for this study is based on and aligned to the MGP site grid.

- The main natural drainage channels in the model domain, Deepdell Creek, NBWR, Murphy's Creek and Tipperary Creek, as well as the main tributaries to these creeks were simulated using drainage cells.
- Each of the opencast pits with the exception of the infilled Southern Pit and Deepdell North Pit are simulated using drainage cells.
- Regional recharge is applied to the uppermost active cells at a rate of 32 mm/year. This recharge rate also applies to the TSF's and WRS's following closure and rehabilitation of the mine.
- Hydraulic conductivity parameters have been applied as indicated in Table 6. These values are the same those applied to the calibrated Kingett Mitchell (2005a) model.
- The storage parameters applied to the schist rock mass on a regional basis (Table 7) are consistent with those applied in existing calibrated models of the MGP. The storage and porosity values applied to the tailings mass have been based on survey and mass balance data for the MTI as well as from documented values for similar tailings impoundments worldwide.

The TSFs have been simulated in the numerical model through:

- Increasing the thickness of the uppermost model layer to match the final proposed form of the TSF.
- Matching the hydraulic conductivity and storage parameters within the simulated TSF to correspond to those defined above for the embankment and tailings materials.
- Defining drainage cells to simulate the drainage systems installed in the TSF's planned construction of underdrains in the three gullies that intersect the TTTSF footprint.
- Defining drainage cells to simulate chimney drains within the upstream face of the planned embankment. The conductance values applied to the drainage cells have been defined to ensure the overlying embankment remains in an unsaturated state. This is the case for the MTI and it has been assumed that similar drainage efficiencies would be achieved for the TTTSF embankment.

The piezometric head within the tailings body during the operational period of the TTTSF has been maintained at an elevation above the general groundwater level in the aquifer system by applying increased groundwater recharge rates. The recharge applied to the tailings mass is 7,500 mm/yr, i.e., 7.5 x 106 m3/yr over an area of 1 ha. This recharge rate is considered to be realistic, as much of the recharge is taken up in storage within the tailings mass. This recharge rate was selected to ensure that the simulated groundwater table within the tailings mass is at the top of the tailings surface at the close of the TSF operational period.

Table 6: Hydraulic conductivity values applied to groundwater model.

Geological feature	K_X (m/s)	K_Y (m/s)	K_Z (m/s)
Weathered schist	3.5×10^{-7}	1.0×10^{-6}	2.5×10^{-7}
Moderately weathered schist	1.0×10^{-7}	2.5×10^{-7}	6.0×10^{-8}
Slightly weathered schist	9.0×10^{-9}	9.0×10^{-9}	1.0×10^{-9}
Unweathered schist	1.0×10^{-9}	5.0×10^{-9}	5.0×10^{-10}
Embankment body	1.0×10^{-6}	1.0×10^{-6}	1.0×10^{-8}
Waste rock	1.0×10^{-6}	1.0×10^{-6}	1.0×10^{-6}
Fine tailings	1.0×10^{-7}	5.0×10^{-7}	1.0×10^{-6}
Coarse tailings	5.0×10^{-6}	5.0×10^{-6}	1.0×10^{-7}

Table 7: Storage property values applied in the groundwater model.

Geological feature	Specific yield	Specific storage	Effective porosity	Total porosity
	(m^3/m^3)	(m^{-1})	(m^3/m^3)	(m^3/m^3)
Heavily and moderately weathered schist	0.02	1.0×10^{-5}	0.01	0.02
Slightly weathered and unweathered schist	0.005	1.0×10^{-5}	0.004	0.005
Tailings	0.38	1.0×10^{-5}	0.35	0.40
Waste rock	0.2	1.0×10^{-5}	0.15	0.2

Drainage cells are defined covering part of the impoundment to simulate the managed water level within the tailings decant pond. These cells ensure the groundwater level within the tailings does not exceed the projected tailings level during the operational period of the TTTSF. At closure of the TTTSF the recharge rate applied to the tailings mass reverts to the regional rate of 32 mm/year, with this rate applied for the remainder of the simulation period.

6.4 Mass Transport Model

6.4.1 Introduction

There are three potential sources of contaminant discharges to the Tipperary Creek catchment. These sources are decant water ponded on top of the tailings mass, tailings pore-water and leached contaminants from the TSF embankment and the WRS's located at the top of the catchment. The contaminants are introduced to the groundwater model through defining concentrations to the recharge water in the area of the tailings, the exposed embankment and across the WRS areas.

A dispersion value of 10 was applied to the numerical mass transport simulations (Kingett Mitchell 2005a), which is normally considered low for the length of the groundwater flow paths involved. Changes in observed water quality at the Observation Wells in Maori Tommy Gully suggest this dispersion value is too large (Golder 2011e). The measured rates of increase in contaminant concentrations is considerably greater than that generated from the models developed in 2005 (Kingett Mitchell 2005a). The change in water quality did not however begin until several years after the model indicated it would occur. On that basis it appears that the dispersion factor applied in the model is too large. Due to time constraints it was not possible to undertake a calibration process for this aspect of the modelling and the dispersion factor applied in the Kingett Mitchell model series has been retained.

The outcomes of the model are not sensitive to the dispersion factor applied due to the long term nature of the contaminant discharges. The peak mass loads calculated for the various receiving water bodies would not be expected to differ substantially from those presented below irrespective of the dispersion factor applied. The timing of the contaminant plume breakthroughs to receiving water bodies is however sensitive to the dispersion factors.

6.4.2 Operational phase contaminant loading

As discussed in Section 4, there are three main potential mining related sources of contaminant discharges to the groundwater system within the modelled area. These sources are decant water ponded on top of the tailings mass, tailings pore-water and leached contaminants from the embankments and the WRS's.

Contaminants are introduced to the groundwater model by the following means:

- In the areas of exposed tailings embankments and WRS's contaminants are introduced to the model as constant concentrations in the recharge water being applied to the uppermost active model cells. The concentrations introduced as recharge are summarised in Table 3.
- Contaminant concentrations in tailings beneath areas expected to be covered by decant water are applied as constant concentrations in the recharge applied to the uppermost active model cells. The concentrations applied as representative of decant water are summarised in Table 2.
- Contaminant concentrations in tailings not covered by ponded decant water are also applied as constant concentrations in the recharge applied to the uppermost active model cells. The concentrations applied as representative of decant water are summarised in Table 3.
- On closure of a TSF, the contaminant concentrations applied in the recharge water are reduced to simulate the removal of the influence slurry water has on the tailings seepage water quality. The concentrations applied are representative of the expected medium to long-term tailings pore water quality (Table 3).
- It is possible that seepage water quality would improve further during the post-closure period simulated as soluble salts become leached out of the tailings mass. This improvement has however not been incorporated in the model for the 150 year post-closure period.

6.4.3 Mass load calculation

The mass loads discharging to various receiving water bodies have been monitored using a zone budget facility available with Visual MODFLOW. This facility enables the groundwater flows in pre-defined zones within the model to be monitored. Contaminant concentrations within these zones are monitored using simulated monitoring points and average concentrations for each zone can be calculated over time. Contaminant mass loads being transported in groundwater through these zones can be calculated from the above information. Contaminants transiting each defined zone are calculated as mass loads with units of kg/day.

Zones have been specified in the groundwater model covering all of the potential natural receiving water bodies as well as cells simulating the drainage system within the TSF's and the opencast pits. The peak mass loads identified from the modelling are then applied to the site water management model to assess the effects on surface water quality at the site.

All of the contaminants introduced to the mass transport model, with the exception of arsenic, are simulated as being conservatively transported. As discussed in Section 4.5.2, arsenic is not transported conservatively in the natural groundwater system at the MGP site. The distribution coefficients and concentration of adsorption sites presented in Table 5 have been applied to simulate a reversible adsorption of arsenic onto the surrounding rock mass.

7.0 CONTAMINANT DISCHARGE PROJECTIONS

7.1 Introduction

Leachate from the TSF's and from the WRS's at the site is expected to discharge to:

- The drainage systems built into the TSF's
- The opencast pits
- The main creeks and tributary gullies close to the site, including those reaches managed through the use of sediment settling ponds and unmanaged sections

During the operational period of the mine, OceanaGold proposes to manage discharges from the TSF drainage systems built into the TSF's and water accumulating in most sediment settling ponds by returning the collected water to the ore processing system. Seepage water that cannot be collected by either the drains or the sediment settling ponds is expected to eventually discharge to the regional drainage system. Additional water management ponds may be established downstream from the sediment settling ponds for leachate mitigation purposes.

The calculated contaminant loads do not necessarily translate directly into contaminant loads in streams, especially during summer. Evaporative losses can result in ephemeral flows at the lower end of some gullies. If there is no discharge flow there is also no contaminant contribution to the main catchment during this time. Application of simulated mass load values for tributaries to the main creeks should be considered worst case values, which would normally only apply during periods of sustained medium to high surface water flow. Similar modelling previously undertaken for sulphate and other conservatively transported contaminants at the MGP has produced slightly conservative estimates of mass transport across the site (Kingett Mitchell 2005a).

As cyanide_{WAD} is not transported conservatively in the groundwater at the site (refer Section 4.5), the projections for cyanide_{WAD} mass loads over the short term are very conservative. Over the long term the simulation results for cyanide_{WAD} may continue to be conservative.

7.2 Tailings Drainage Systems

Following closure of the SPI, tailings from SP11 are to be recovered and dry stacked on top of the MTI. Those tailings that have not become dewatered in time for excavation will be recovered as a slurry and pumped to the TTTSF. This process is to be completed before the waste rock stored in Round Hill is removed. The recovery of the tailings is expected to require perhaps two years, during which time the upper levels of the SP11 embankment are also to be removed. At completion of the removal process drain discharges from SP11 would have ceased.

During the same period that SP11 is being removed, the tailings surface over SP10 is to be reshaped and rehabilitated. Discharges from the SP10 drains are expected to decline substantially during the operational period of the mine and continue to decline slowly over several decades following closure (Figure 8).

Due to the scale of the groundwater model, the simulated TSF drainage systems are limited in detail. The upper level drainage systems built into the MTI and SPI are poorly represented and this leads to understating of the drainage flows that may be expected at closure. As this water is essentially being directly recycled within the MGP mine water system, this issue is not considered to be crucial to the long term contaminant transport simulation.

Drainage discharges from the combined MTI and SPI at closure were simulated to be approximately 1,800 m^3/day. Monitoring of drainage flows at the site indicates the total flows at closure are more likely to be in the order of 2,500 m^3/day (Golder 2011c).

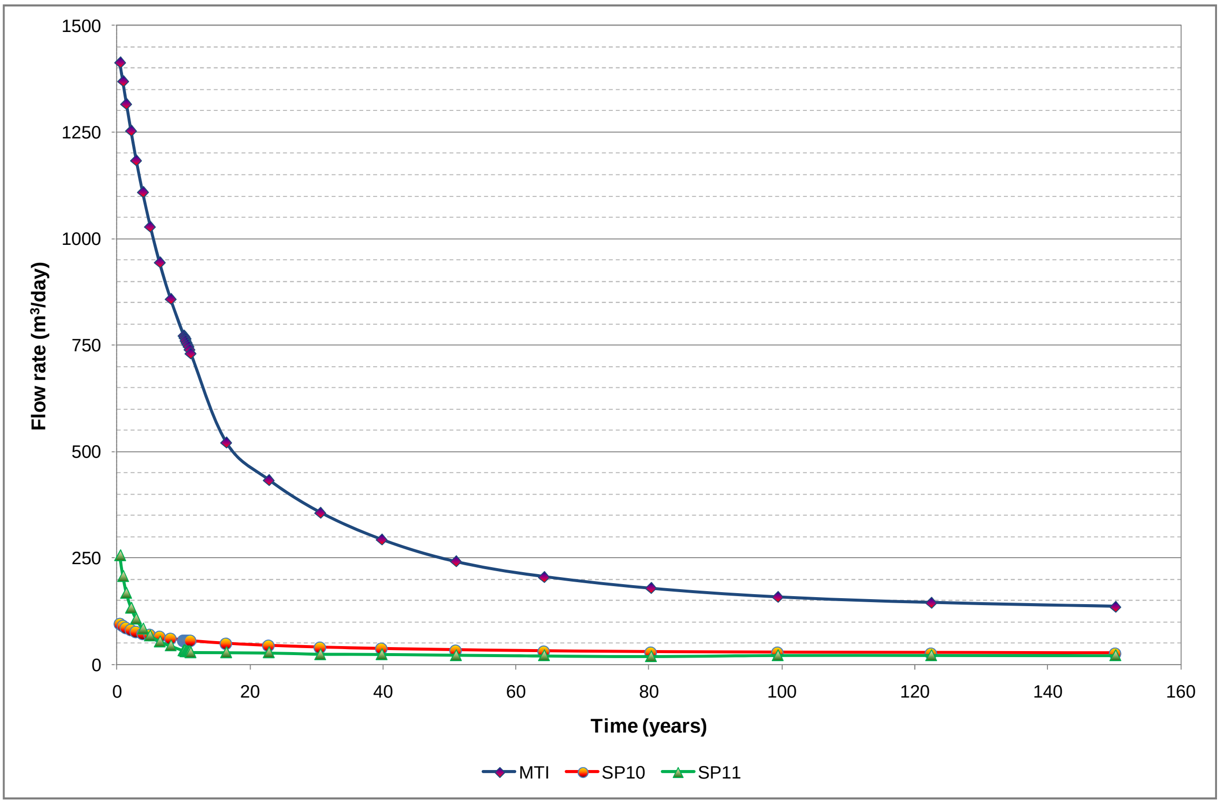

Figure 8: Simulated post-closure declines in TSF drain discharge.

Assessment of the rates at which MTI and SPI drain discharges have declined during inactive periods in the past indicates discharges are likely to decline by between 50% and 90% within two years following closure (Golder 2011c). These estimates support a previous assessment of post-closure drainage rate declines at the site (Kingett Mitchell 2005a).

For the purpose of post-closure mine water management, an estimate of drainage flows has been generated based on site monitoring of key embankment drains (Golder 2011c) rather than on the groundwater model (Figure 9 and Figure 10). The flows from Figure 10, extrapolated to 10 years following closure, have been matched to stages in the mine water management model (Appendix E). The projection for the rate at which TSF discharge flows decline is likely to be conservative, however this projection is still more rapid than that derived from the groundwater model (Figure 11).

It is expected that much of the stored tailings mass would become unsaturated during the 20 years following closure of a TSF. There is, however, considerable uncertainty with respect to the length of time required for the overall groundwater system within the tailings to reach a steady state flow pattern. This uncertainty is partly due to the inherent variability of the hydrogeologic characteristics of the tailings mass. In addition, dynamic factors such as compaction of both the tailings mass and the underlying soils have not been taken into account in this projection.

Once most of the tailings mass has become unsaturated the contaminant loads discharging from the tailings would be associated with the residual moisture content and ongoing recharge from rainfall. Further transport of contaminants from the tailings would mainly occur in response to significant rainfall events. These events would lead to pulses of seepage water travelling downward through the unsaturated tailings to the groundwater table. These pulses, averaged on a long term annual basis, are expected to be equivalent to the natural 32 mm/year groundwater recharge rate for the region.

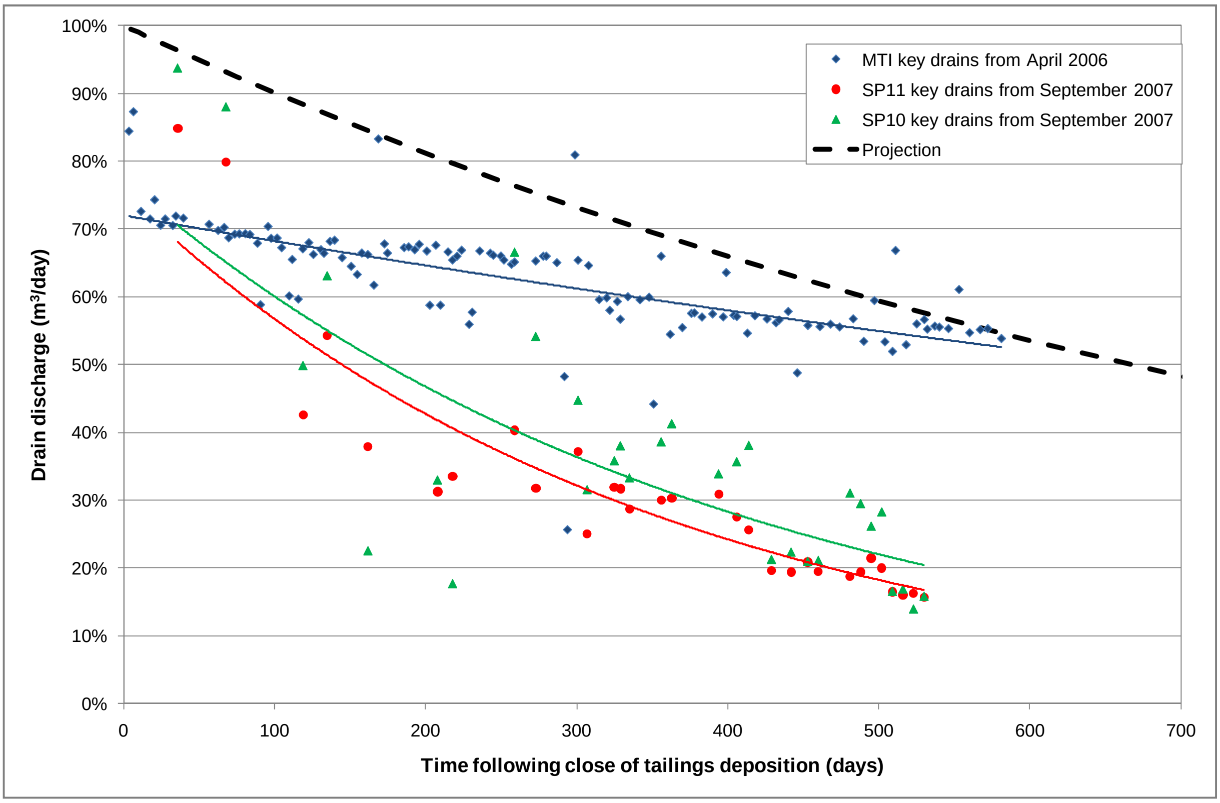

Figure 9: Projected decline in TSF drain discharge following closure as a percentage.

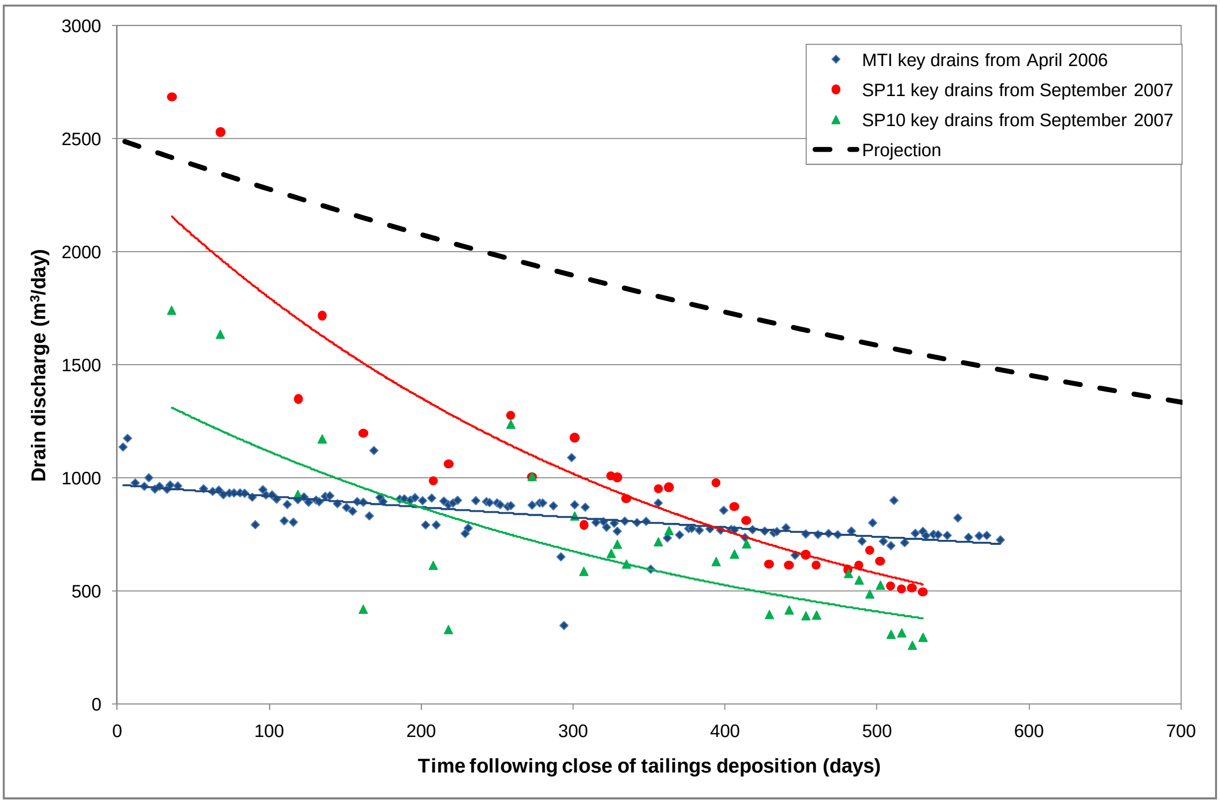

Figure 10: Projected decline in TSF drain discharge following closure as a flow rate.

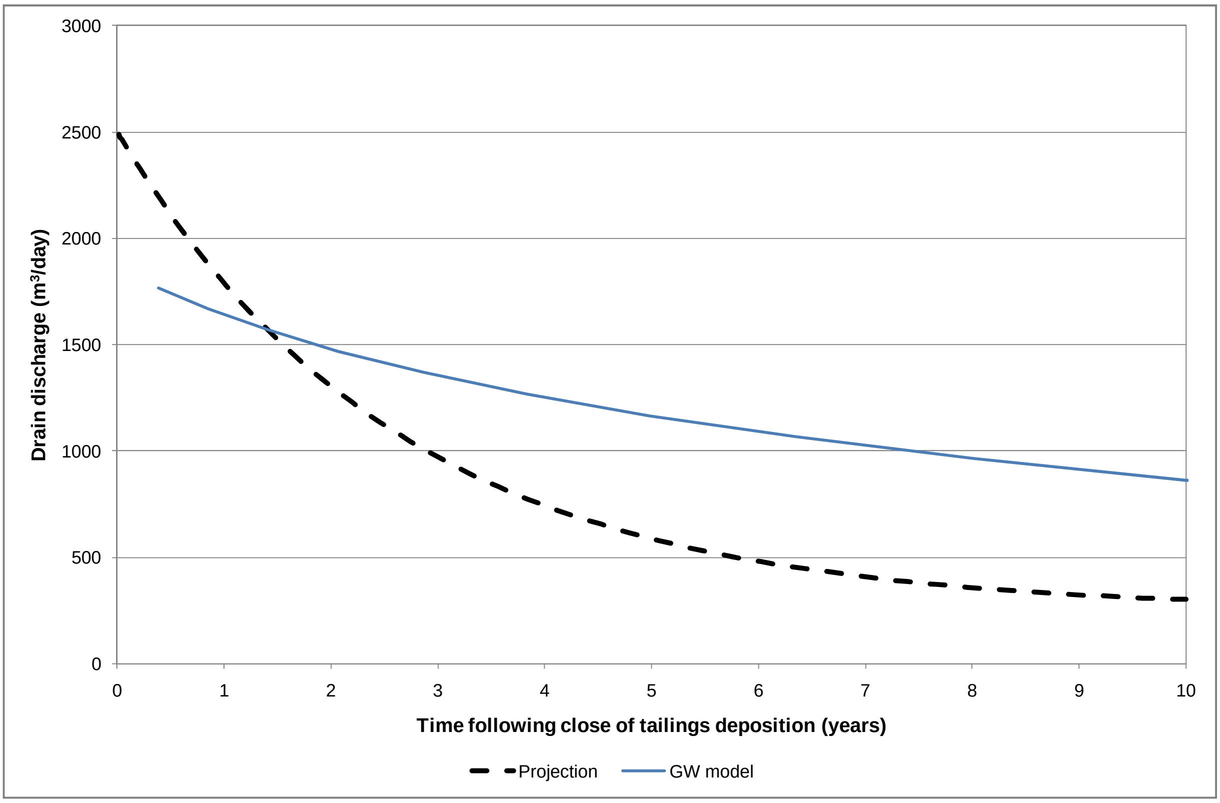

Figure 11: Projected decline in TSF drain discharge compared to groundwater model outcomes.

7.3 Deepdell Creek

7.3.1 Upstream from DC07

Groundwater discharge flows and average contaminant concentrations have been calculated for:

- Direct seepage to Deepdell Creek and its tributaries downstream from silt ponds that are used for water management purposes
- Seepage that discharges to tributaries upstream from managed silt ponds

The simulated groundwater seepage flows discharging to Deepdell Creek and the unmanaged tributaries in the stretch between DC01 and DC07 vary slightly over time. As the peak concentrations for the major ions relate to the long term future, the flow applied for the post closure contaminant load calculations is 730 m^3/day (Figure 12).

Average concentrations for the major ions derived from the MGP discharging to Deepdell Creek upstream from DC07 are summarised in Figure 13. Sulphate reaches an average concentration of approximately 590 g/m^3 over the long term. The outcomes of the contaminant transport assessment are summarised in Appendix E.

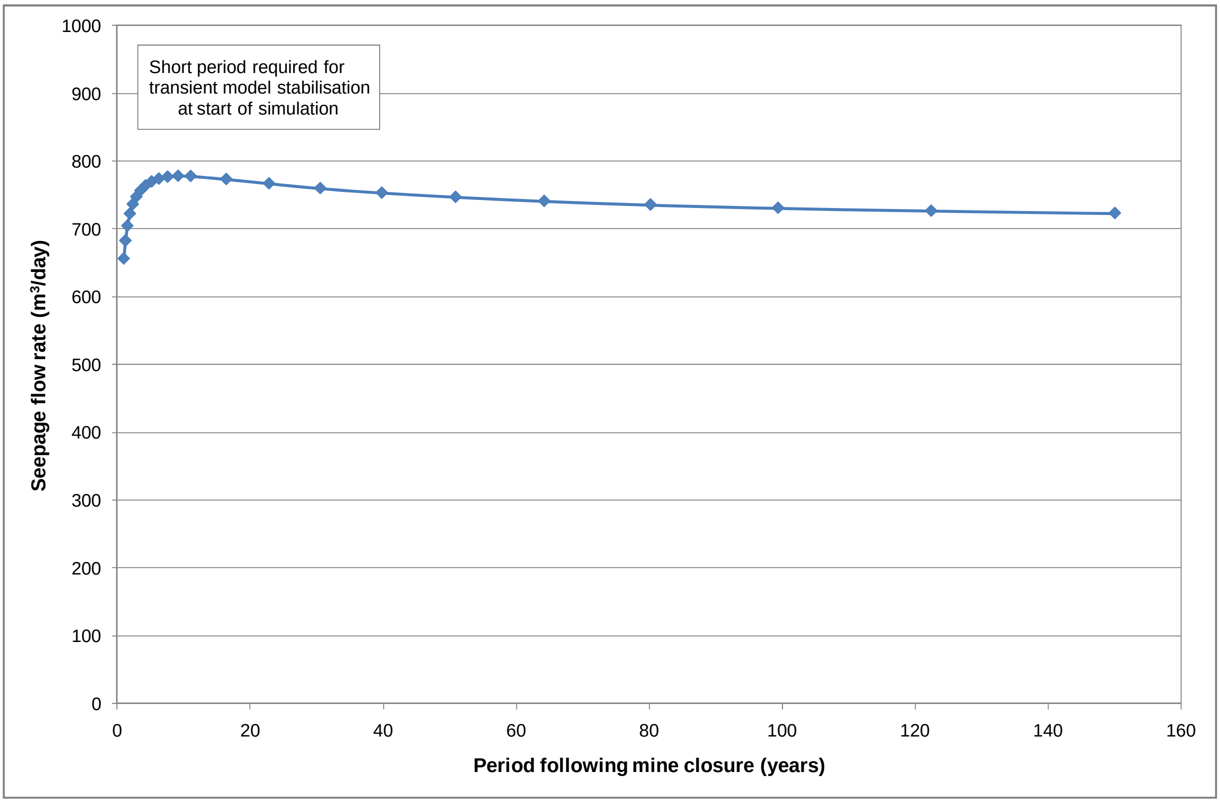

Figure 12: Total seepage flow to Deepdell Creek and tributaries between DC01 and DC07.

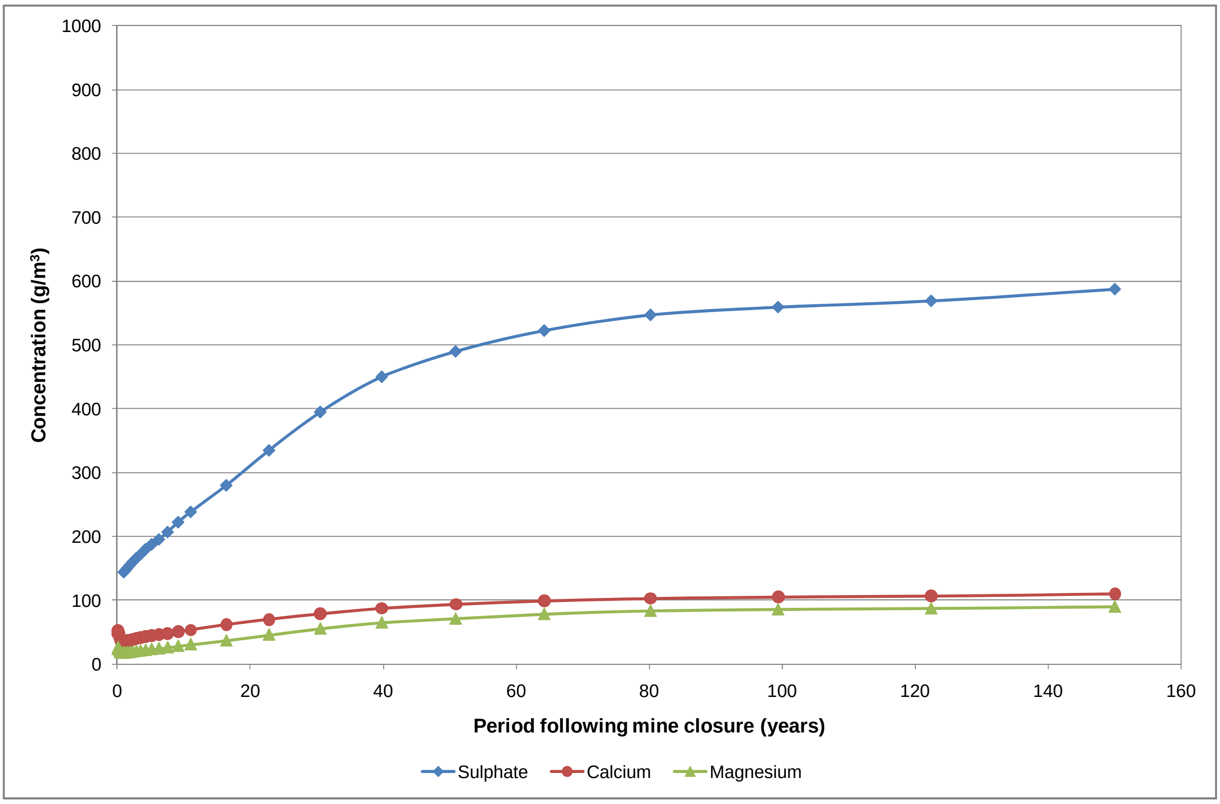

Figure 13: Average major ion concentrations in seepage to Deepdell Creek and tributaries between DC01 and DC07.

Arsenic concentrations in Deepdell Creek are primarily derived from contributions from the HMSZ area that were influencing the stream water quality prior to the start of MGP operations (Golder 2011e). It is assumed that this increase in arsenic concentration is due to ongoing contributions from natural groundwater and leaching of historical mining features at Golden Point. On that basis, adding a natural arsenic contribution of approximately 0.03 g/m^3 to the groundwater seepage flow discussed above enables an acceptable simulation of baseline water quality.

At present there is essentially no contribution of arsenic from the MGP to Deepdell Creek, primarily due to the natural adsorptive properties of the soils and weathered rock underlying the TSF's. It is expected the increase in average arsenic concentration in groundwater discharging to Deepdell Creek would peak at approximately 0.057 g/m^3.

The simulated average cyanide$_{WAD}$ concentration in groundwater discharging to Deepdell Creek reaches a maximum of 0.06 g/m^3. As the simulation assumes conservative transport of cyanide$_{WAD}$ whereas this is not the case in reality, the model outcomes are considered to be very conservative.

7.3.2 Between DC07 and DC08

The stretch of Deepdell Creek between DC07 and DC08 is calculated to receive groundwater flows of approximately 350 m^3/day. This is not expected to change greatly following construction of the Back Road WRS.

A contaminant plume would be generated by the WRS however the simulated time required for the contaminants to reach Deepdell Creek exceeds six years. This expectation is supported by the 12 to 14 year delay in contaminants transported by groundwater from the Detection Wells in Maori Tommy Gully to the Compliance Wells; a distance of approximately 300 m. At its closest approach, the Back Road WRS is approximately 500 m from Deepdell Creek.

The construction of the Back Road WRS is therefore unlikely to result in an additional contaminant load transported in groundwater to Deepdell Creek before the end of mining operations at the site (Appendix E). The average sulphate concentrations simulated in groundwater discharging to Deepdell Creek between DC07 and DC08 are expected to peak at approximately 1,050 g/m^3 some 100 years following site closure. The simulated concentrations do not decline again as the concentrations applied to the WRs are constant over the post-closure period of the model.

The simulation indicates arsenic is unlikely to produce a detectable increase in concentration within the groundwater at Deepdell Creek during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of the major ions and metals expected in the groundwater between DC07 and DC08 are summarised in Appendix E.

7.3.3 Maori Tommy Gully silt pond

Groundwater seepage to Maori Tommy Gully silt pond is calculated to be approximately 70 m^3/day. Contaminants in groundwater discharging to the NBWR are primarily sourced from the MTI. The model indicates that contaminant concentrations in the Maori Tommy Gully silt pond are unlikely to reach a peak until after site closure.

The long term average sulphate concentrations in groundwater discharging to the Maori Tommy Gully silt pond are expected to peak at about 628 g/m^3. Water quality monitoring at the Detection Wells installed in Maori Tommy Gully indicates the sulphate concentrations in groundwater upstream from the silt pond have been within a range of 600 g/m^3 to 1,400 g/m^3 during 2010 (Golder 2011e). These wells were, however, installed in a line less than 20 m long, positioned to intersect the maximum plume concentrations. In contrast, the cell size in the groundwater model in this area is 50 m. In effect, the groundwater model is averaging concentrations over a wider area. Although the calculated concentrations are less than those recorded from the Detection Wells, the mass load over the wider area is expected to be representative of the plume in Maori Tommy Gully.

The model indicates average arsenic concentration in groundwater discharging to the Maori Tommy Gully silt pond is likely to peak at about 0.6 g/m^3. Due to the adsorption of arsenic onto the soils and weather rock underlying the MTI this concentration is unlikely to be reached until decades after site closure.

$Cyanide_{WAD}$ in the groundwater discharging to the Maori Tommy Gully silt pond is calculated to peak at a concentration of approximately 1.1 g/m^3 about 20 years following site closure. This calculation is however based on conservative transport of $cyanide_{WAD}$ in the groundwater system, whereas this parameter is not conservatively transported (refer Section 4.5.4). The model results for $cyanide_{WAD}$ therefore substantially overstate the peak discharge concentrations.

The peak concentrations of other contaminants are summarised in Appendix E.

7.3.4 Northern Gully silt pond

Groundwater seepage to the Northern Gully silt pond is calculated to be approximately 40 m^3/day. Contaminants in groundwater discharging to the silt pond are primarily sourced from the Northern Gully WRS. It is expected that the contaminant concentrations in the silt pond have already reached their peak. The long term average sulphate concentrations in groundwater discharging to the Northern Gully silt pond are calculated to peak at about 2,300 g/m^3. The observed sulphate concentrations of about 2,500 g/m^3 are due to WRS underdrains that also discharge to this silt pond (Golder 2011e). The underdrains could not be incorporated in the groundwater model however the combined groundwater and drainage flows to the silt pond are small. It is not likely that this discrepancy would have a substantial effect on outcomes from the mine water management.

The model indicates arsenic is unlikely to produce a detectable increase in concentration within the groundwater at the silt pond during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of other contaminants are summarised in Appendix E.

7.3.5 Back Road silt pond

Groundwater seepage to the proposed Back Road silt pond is calculated to be approximately 65 m^3/day. Contaminants in groundwater discharging to the silt pond are sourced from the planned Back Road WRS. The model indicates the contaminant concentrations in the groundwater discharging to the Back Road silt pond will not peak until decades after site closure. The long term average sulphate concentrations in groundwater discharging to the Back Road silt pond are calculated to peak at about 2,200 g/m^3.

The model indicates arsenic is unlikely to produce a detectable increase in concentration within the groundwater at the silt pond during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of other contaminants are summarised in Appendix E.

7.4 North Branch Waikouaiti River

7.4.1 Direct discharges to North Branch Waikouaiti River

Groundwater seepage to the NBWR upstream from the Red Bank Road monitoring station is calculated to be approximately 100 m^3/day. Contaminants in groundwater discharging to the NBWR are primarily sourced from the Frasers West WRS. Elevated concentrations of the major ions have already been detected in monitoring wells along the NBWR (Golder 2011e). The model results indicate that average concentrations in the groundwater will not peak until decades after mine closure.

The long term average sulphate concentrations in groundwater discharging to the NBWR upstream from Red Bank Road are expected to peak at about 1,090 g/m^3. The model indicates arsenic is unlikely to produce a detectable increase in concentration within the groundwater at the NBWR during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of other contaminants are summarised in Appendix E.

The model indicates that seepage losses from the MTI are unlikely to be transported toward the NBWR. The groundwater catchment divide is located between the MTI and the NBWR.

7.4.2 Frasers West silt pond

Groundwater seepage to the Frasers West silt pond is calculated to be approximately 54 m^3/day. Contaminants in groundwater discharging to the NBWR are primarily sourced from the Frasers West WRS. Elevated concentrations of the major ions have been detected in water from the silt pond for several years although these contaminants are also partially sourced from run-off water (Golder 2011e). The model results indicate that average concentrations in the groundwater at the silt pond will not peak until after mine closure.

The long term average sulphate concentrations in groundwater discharging to the NBWR upstream from Red Bank Road are expected to peak at about 1,500 g/m^3. Although much of the Frasers West silt pond catchment is covered by waste rock, infiltration of rainwater immediately up-gradient stream from the silt pond is likely to provide a small contaminant dilution capacity.

The model indicates arsenic is unlikely to produce a detectable increase in concentration within the groundwater at the silt pond during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of other contaminants are summarised in Appendix E.

7.5 Murphys Creek

7.5.1 Direct discharges to Murphys Creek

Groundwater seepage to Murphys Creek, downstream from the Murphys Creek silt pond and upstream from the compliance monitoring site MC100 is calculated to be approximately 180 m^3/day. The model results indicate that average concentrations in the groundwater at the silt pond will not peak until decades after mine closure.

The long term average sulphate concentrations in groundwater discharging to the Murphys Creek upstream from MC100 are expected to peak at about 380 g/m^3. This is likely to be an overestimate as discussed in Section 7.5.2.

The model indicates arsenic is unlikely to produce a detectable increase in concentration within the groundwater at the silt pond during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of other contaminants are summarised in Appendix E.

7.5.2 Murphys Creek silt pond

Groundwater seepage to the Murphys Creek silt pond is calculated to be approximately 122 m^3/day. Elevated concentrations of the major ions have been detected in water from the silt pond for several years although these contaminants are also partially sourced from run-off water (Golder 2011e). The model results indicate that average concentrations in the groundwater at the silt pond will not peak until after mine closure.

The long term average sulphate concentrations in groundwater discharging to the Murphys Creek silt pond are expected to peak at about 1,320 g/m^3. This is likely to be an underestimate as the model indicates more leachate water passes in a plume under the silt pond than would be expected. The model indicates this plume discharges to Murphys Creek in the stretch between the silt pond and MC100, leading to a likely overestimate of the concentrations for this section of the creek.

The model indicates the concentrations of arsenic in the groundwater discharging to the silt pond are unlikely to increase above background levels during the 150 year simulation period of the model. This is due to the low concentration of arsenic in the WRS leachate water and the adsorption of arsenic to the underlying soils. The peak concentrations of other contaminants are summarised in Appendix E.

7.6 Pit Lakes

7.6.1 Introduction

Groundwater inflows to the opencast pits at the MGP can be expected to vary depending on the water level of the pit lake. As the water level in a pit rises following closure the groundwater hydraulic gradients toward the pit decrease and the groundwater inflows to the pit also decrease.

Only two of the MGP pits can potentially lose water to groundwater as the water level in the pit rises. These two are the Round Hill/Golden Point pit and the Deepdell South pit. In each case the pit is separated from the Deepdell Creek valley by a relatively narrow area of rock left intact following mining. The net flows to Deepdell South pit are not discussed in this report. The flows to Round Hill/Golden Point pit are discussed in Section 7.6.2.

7.6.2 Round Hill/Golden Point pit lake

Groundwater inflows to Round Hill pit greatly exceed those to the remnant Golden Point pit on closure. In effect, the much deeper Round Hill pit acts as a sump to Golden Point pit. The model indicates groundwater inflows to Golden Point pit would contribute less than 2% of the overall groundwater inflow to the combined pit.

At its maximum extent, groundwater inflows to the combined Round Hill/Golden Point pit are calculated to peak at approximately 180 m^3/day. Simulating a pit lake water level in the model at an elevation of 400 mRL, results in the projected groundwater inflows decreasing to 136 m^3/day. At a pit lake elevation of 430 mRL the groundwater inflows are calculated to be approximately 90 m^3/day.

Above an elevation of 340 mRL the inflows begin to be offset by seepage losses through the historical underground workings at the northern end of Golden Point pit (Golder 2011b). If no mitigation is undertaken the losses through the underground workings would exceed the inflows once the pit lake rises a few metres above 340 mRL.

The model indicates the concentration of sulphate in groundwater discharging to the combined pit would peak at approximately 900 g/m^3. During 2010 the sulphate concentrations detected in water pumped from the Golden Point pit sump have considerably exceeded this concentration. This is partly due to the large volumes of waste rock already stored in Round Hill pit, which are to be removed once mining operations shift to that pit. In addition, seepage losses from SP11 through either disturbed pit wall rock or leaking drainage systems have led to water quality being poorer than what would be expected from waste rock seepage alone. Each of the above issues would be removed once mining operations in Round Hill pit are initiated.

The peak concentrations of other contaminants are summarised in Appendix E.

Following completion of the groundwater modelling work for this project a report into the geotechnical stability of the MTI was provided to Golder (PSM 2010). This report indicates sections of the MTI are likely to settle after the SP11 embankment has been removed. It was not possible to take this factor into account in the current study, of seepage flows to Round Hill pit, however a short discussion of some of the implications is provided below.

Prior to construction of the SP11 embankment the eastern edge of the MTI had already been subject to subsidence. Movement was at least partially focused on the Footwall fault of the HMSZ, which intersects the MTI. This subsidence was managed through intensive drilling of low angle depressurisation holes into the Round Hill and Southern Pit walls beneath the MTI. Additional vertical dewatering wells were installed in the same area. At the time this report was produced, the combined flows from these dewatering systems had not been reviewed, however they are understood to have been significant in terms of the overall inflows calculated for Round Hill pit.

Removal of the SPI would presumably imply installing a new rock mass depressurisation system in order to manage the MTI stability. The flows that could potentially discharge from this system would be characterised by TSF pore water quality as they would be primarily sourced from the MTI. These flows are in effect a re-routing of flows that are currently modelled as discharging to the MTI drainage systems. On that basis, these flows could be expected to decline as the tailings in the MTI become dewatered.

During the operational period of the mine these rock mass depressurisation flows would be recovered through the mine water management system. Following closure these flows could be recovered and pumped to Frasers Pit, as is planned for the MTI and SP10 drain discharges, or they could be released to Golden Point pit provided the long term pit lake water quality was acceptable.

7.6.3 Frasers/Innes Mills pit lake

Groundwater inflows to Innes Mills and Frasers pit lakes are treated as being connected as they are expected to eventually become combined following closure. At its maximum extent, groundwater inflows to the combined pits are calculated to peak at approximately 1,280 m^3/day. These flows decline in response to declining flows from SP10 and rising lake water levels. At a pit lake elevation of 480 mRL, which is the approximate overflow level to the NBWR, the groundwater inflows are calculated to be approximately 52 m^3/day (Appendix E).

The model indicates the average concentration of sulphate in groundwater discharging to Frasers pit would peak at approximately 450 g/m^3, reflecting SP10 as the primary source of seepage water. This concentration is expected to decline slightly over time.

The projected average concentration of arsenic in groundwater discharging to Frasers pit is expected to peak at approximately 0.057 g/m^3, reflecting the discharges from SP10. Although the concentrations in water from SP10 are expected to decline slowly, the average concentrations of arsenic in the seepage water are expected to decline more rapidly. This decline in the average seepage water concentration reflects the decline in flows from SP10 and the increasing importance of waste rock leachate in the water balance.

Average cyanide$_{WAD}$ concentrations in the seepage water are projected to peak at about 0.4 g/m^3 shortly after mine closure. As discussed previously, this concentration is expected to be significantly overestimated due to the application of conservative transport for cyanide$_{WAD}$. The peak concentrations of other contaminants are summarised in Appendix E.

8.0 SUMMARY AND CONCLUSIONS

Modelling of contaminant mass transport within the groundwater system at the MGP has been undertaken to cover the period from 2010 through to the close of mining operations at the site at the start of 2020 followed by a 150 year post-closure period. Beyond that period potential changes in the hydrogeological behaviour of the tailings material and climactic conditions are considered to limit the usefulness of predictive modelling.

The mine operations and post-closure period have been divided into stages to support the development of a mine water management model. Groundwater discharge rates and peak contaminant concentrations have been calculated to coincide with these stages of the mine life and documented in Appendix E.

The groundwater model used for this assessment has been based on an existing calibrated model simulating the groundwater system at the site through to 2010 (Kingett Mitchell 2005a). Groundwater recharge rates applied to the TSF's are calibrated to ensure the water table within the tailings is at the tailings surface during the operational period of the mine. The regional recharge rate of 32 mm/year has also been applied to the rehabilitated WRS's and TSF's at the site.

The groundwater discharge flows to potential contaminant receiving waters within and surrounding the MGP have been calculated and documented in a format suitable for inclusion in the mine water management model. In most cases the results are based directly on the outcomes from the groundwater model documented in this report. The exceptions are the discharge flows from the TSF drainage systems. Projected discharges from the TSF drains have been calculated from site monitoring records and an evaluation of the rates at which discharge flows decline following closure of a TSF.

Groundwater quality input parameters have been based on water quality data from the site environmental monitoring program. These include leachate water quality representing TSF decant ponds, TSF drain discharges and WRS seepage. The water quality parameters simulated include the major ions, a range of metals, arsenic and cyanide_{WAD}.

Contaminant transport for each of the simulated contaminants with the exception of arsenic has been undertaken on the basis of conservative transport within the groundwater system. Arsenic transport has been modelled based on arsenic (III) being the main form of this element in the tailings seepage water. The adsorption parameters for arsenic (III) have been derived from testing of rock and soil samples from the site and applied in the groundwater modelling.

The model simulated flow rates and the concentrations of sulphate and arsenic in groundwater discharging to the receiving waters are summarised below.

The simulated groundwater seepage flows discharging to Deepdell Creek and the unmanaged tributaries in the stretch between the upstream water quality monitoring point DC01 and DC07 vary slightly over time. As the peak concentrations for the major ions relate to the long term future, the flow applied for the post closure contaminant load calculations is 730 m^3/day (Table 8).

Table 8: Summary of MGP post-closure groundwater discharges and discharge water quality.

Receiving Water	Groundwater discharge rate (m^3/day)	Arsenic (g/m^3)	Sulphate (g/m^3)
Deepdell Creek at DC07	730	0.03	590
Deepdell Creek at DC08	116	0.05	1,050
Murphys Creek at MC100	180	0.03	380
North Branch Waikouaiti River at NBWRRB	100	0.03	1,050

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APPENDIX A

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APPENDIX B

Summary of Hydraulic Test Results

Data relating hydraulic conductivity to depth below surface has been gathered during several phases of site investigation (GCNZ 1988, WWC 1992a, Kingett Mitchell 2004, EGL 2010, Golder 2010a). The available data is derived from a combination of pumping, slug and lugeon tests.

Almost all of the results represented as 1 x 10^{-9} m/s in Figure B1 through to Figure B3 are actually returns of <1 x 10^{-9} m/s, which was the detection limit of the lugeon tests under the injection pressures and time periods applied. A discussion of the hydraulic conductivity test results derived from the site geotechnical evaluation program by GCNZ (1988) indicates the results can be divided into two groups. The results of 1 x 10^{-9} m/s or less were taken to represent the hydraulic conductivity of the unfractured rock matrix as opposed to the more general fractured rock mass.

Where the results apparently derived from testing unfractured rock sections are included in the analysis of hydraulic conductivity with depth, no trend is discernable, as indicated by the red data and trend lines in Figure B1 and Figure B2. Where the values indicating the result is below the detection limit of the test are excluded a trend of decreasing hydraulic conductivity with depth is detectable, as indicated by the blue data and trend lines in the same figures. Results from the CTI area and the Frasers Underground area do not show a reduction in hydraulic conductivity with depth, although few tests were performed in the latter area.

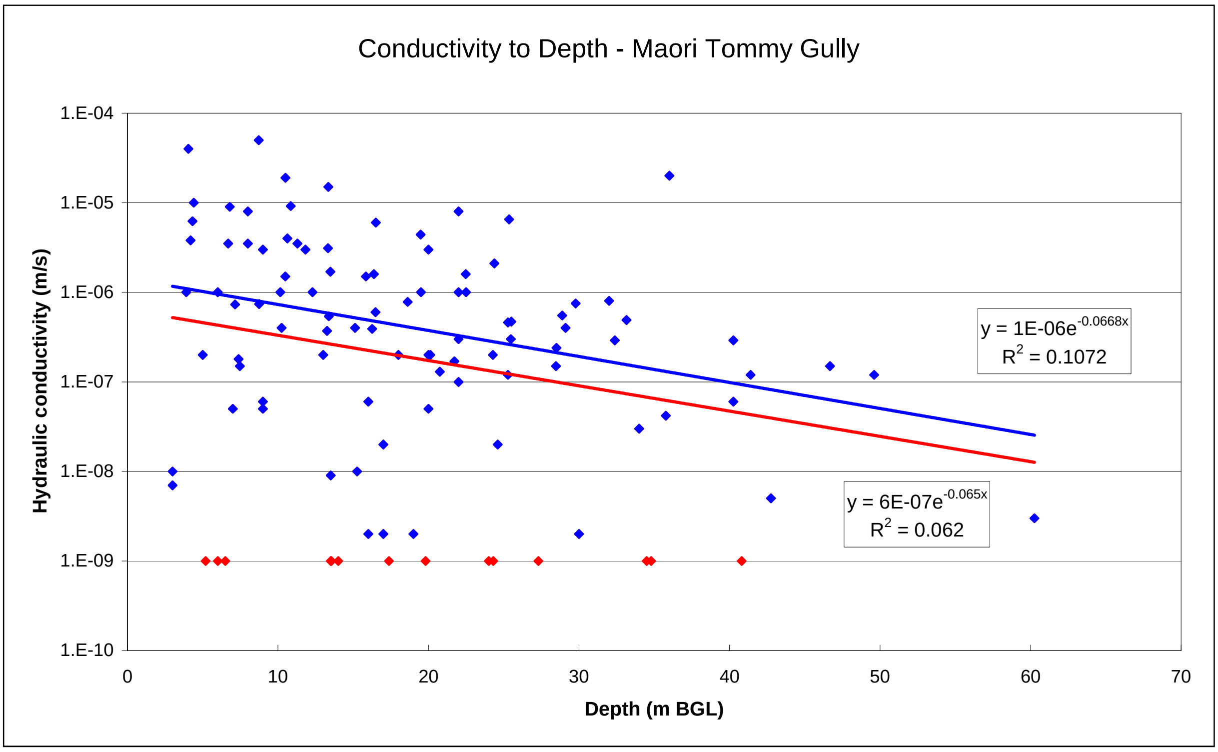


Figure B1: Hydraulic conductivity variation with depth – lower Maori Tommy Gully area (Kingett Mitchell 2005).

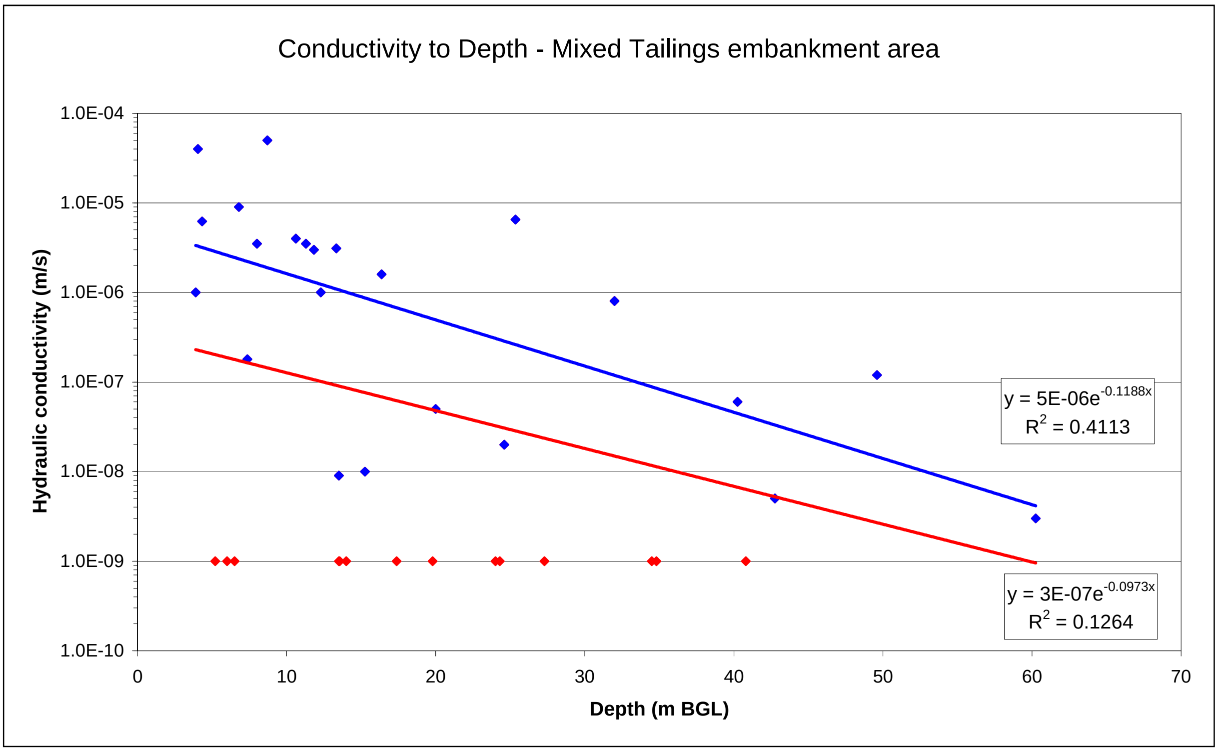


Figure B2: Hydraulic conductivity variation with depth – Mixed Tailings embankment area (Kingett Mitchell 2005).

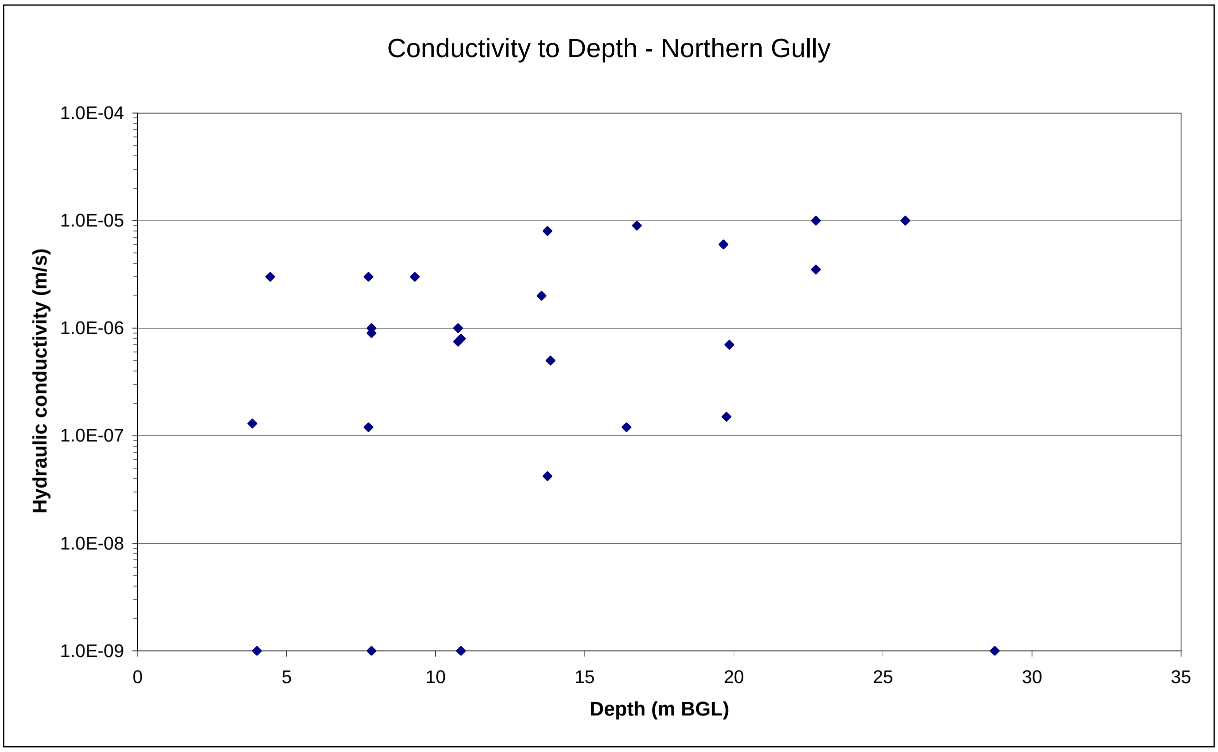


Figure B3: Hydraulic conductivity variation with depth – Concentrate Tailings embankment area (Kingett Mitchell 2005).

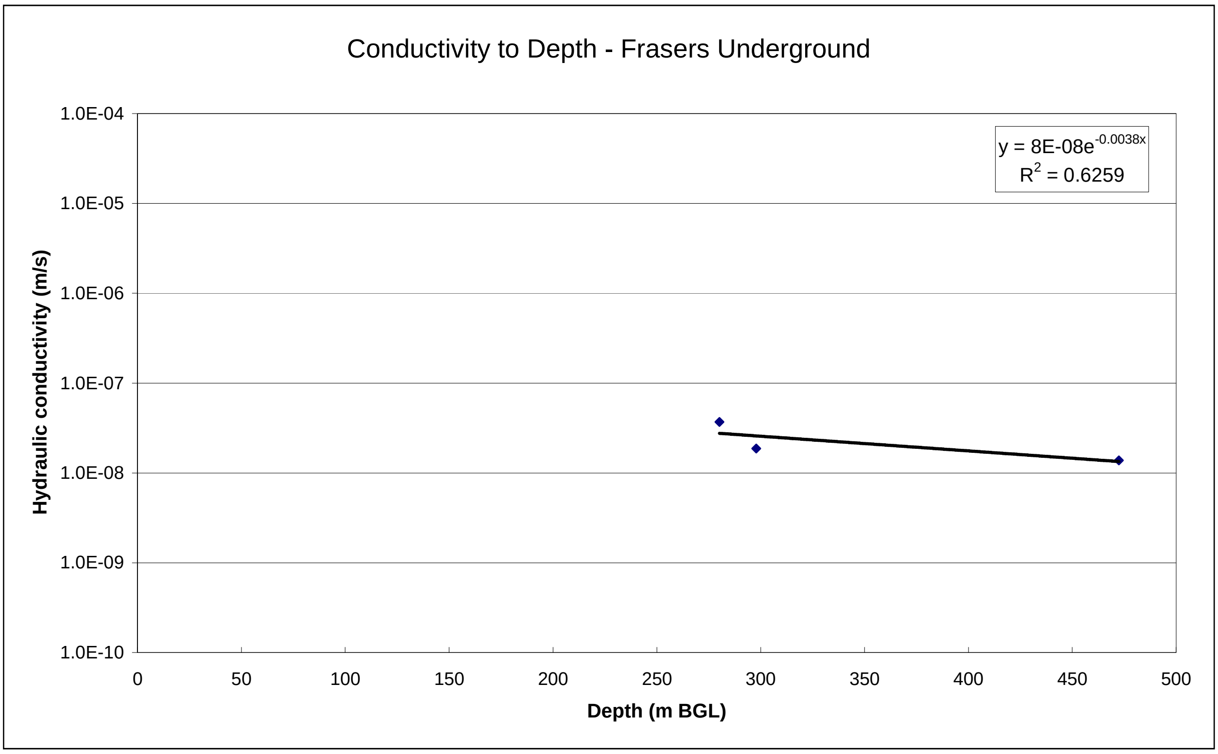


Figure B4: Hydraulic conductivity variation with depth – Frasers Underground mine area (Kingett Mitchell 2005).

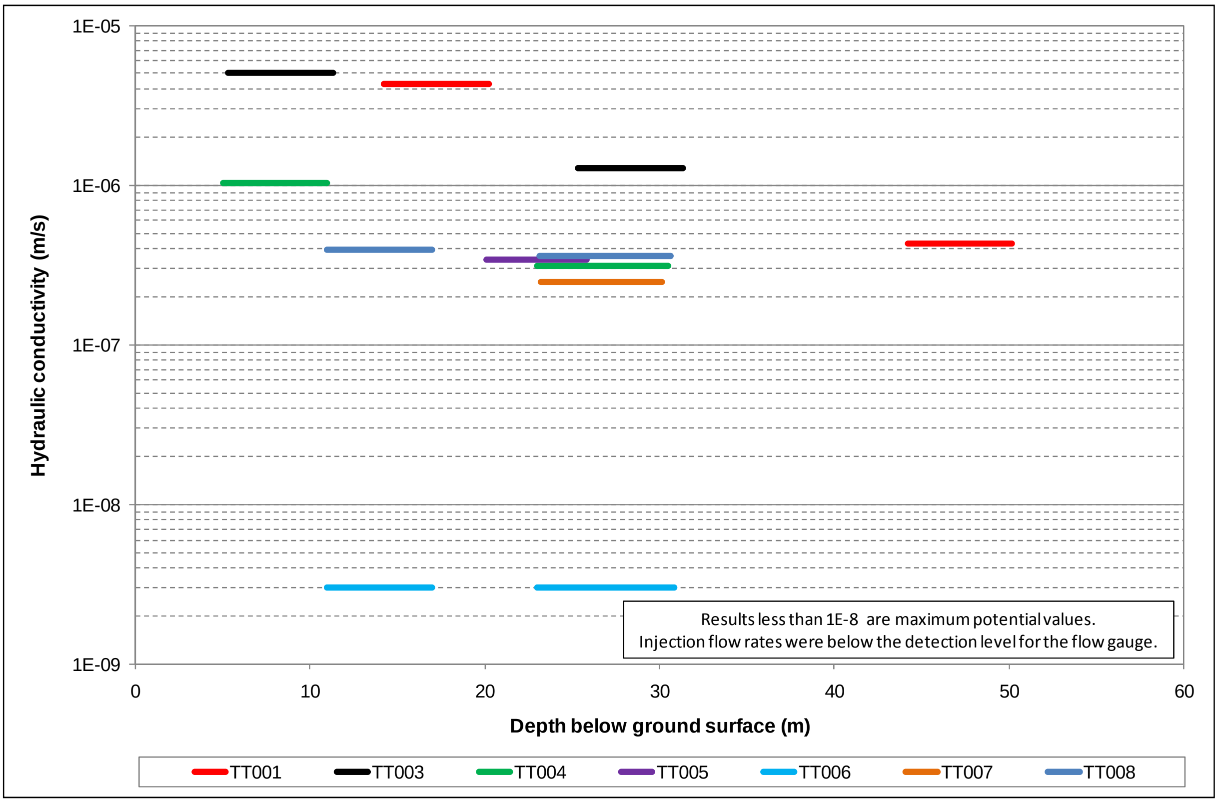


Figure B5: Hydraulic conductivity variation with depth, TTTSF area including detection limit results (Golder 2010).

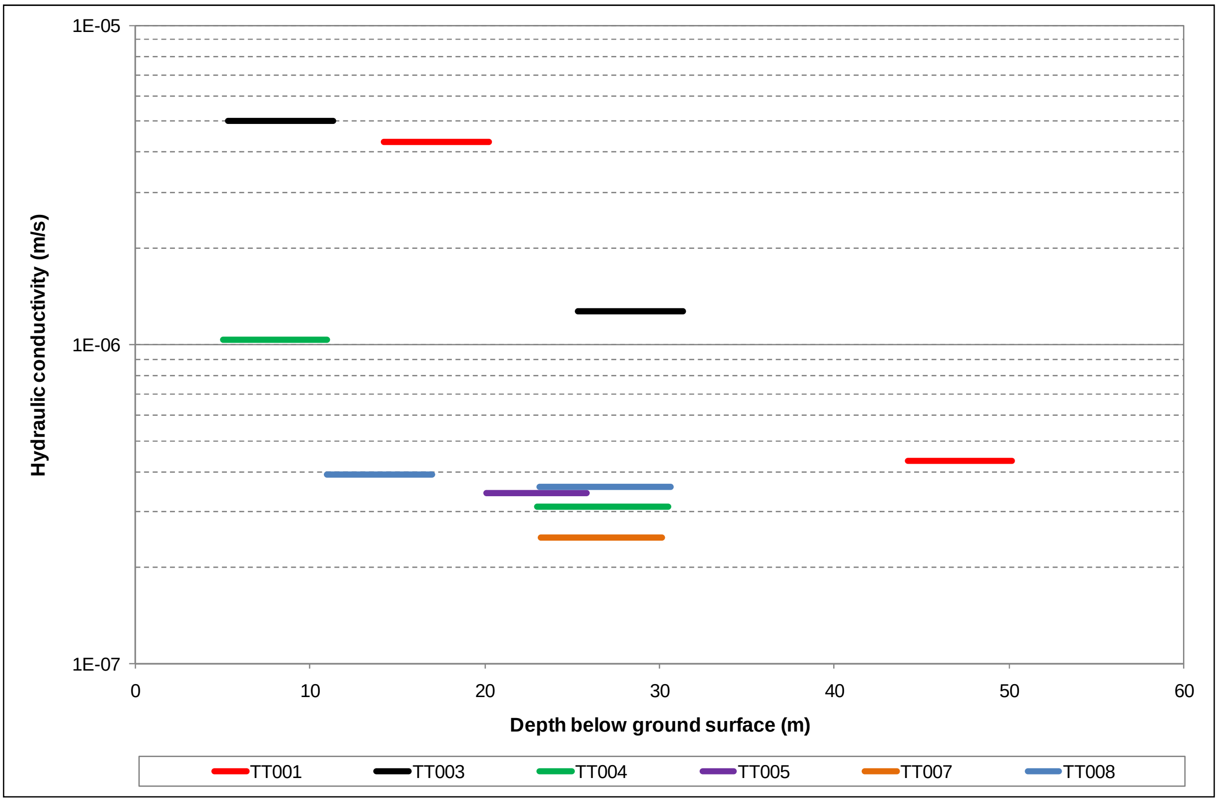


Figure B6: Hydraulic conductivity variation with depth, TTTSF area excluding detection limit results (Golder 2010).

Table 1: Back Road WRS area – slug test analysis results.

Drillhole ID	Test Number	Depth from (mbgl)	Depth to (mbgl)	Hydraulic conductivity [(1)] (m/s)
BH3	Test 1	11.76	20.31	1.7×10^{-7}
BH3	Test 2	25.26	35.31	7.0×10^{-7}
BH3	Test 3	40.21	50.31	4.2×10^{-8}
BH6	Test 1	12.76	20.31	2.7×10^{-7}
BH6	Test 3	27.95	35.31	1.1×10^{-6}
BH6	Test 4	40.30	50.31	1.3×10^{-6}
BH9	Test 1	10.26	20.31	1.7×10^{-6}
BH9	Test 3	29.76	35.31	2.2×10^{-6}

Note: 1) Hydraulic conductivity value presented is calculated after Sharp (1975).

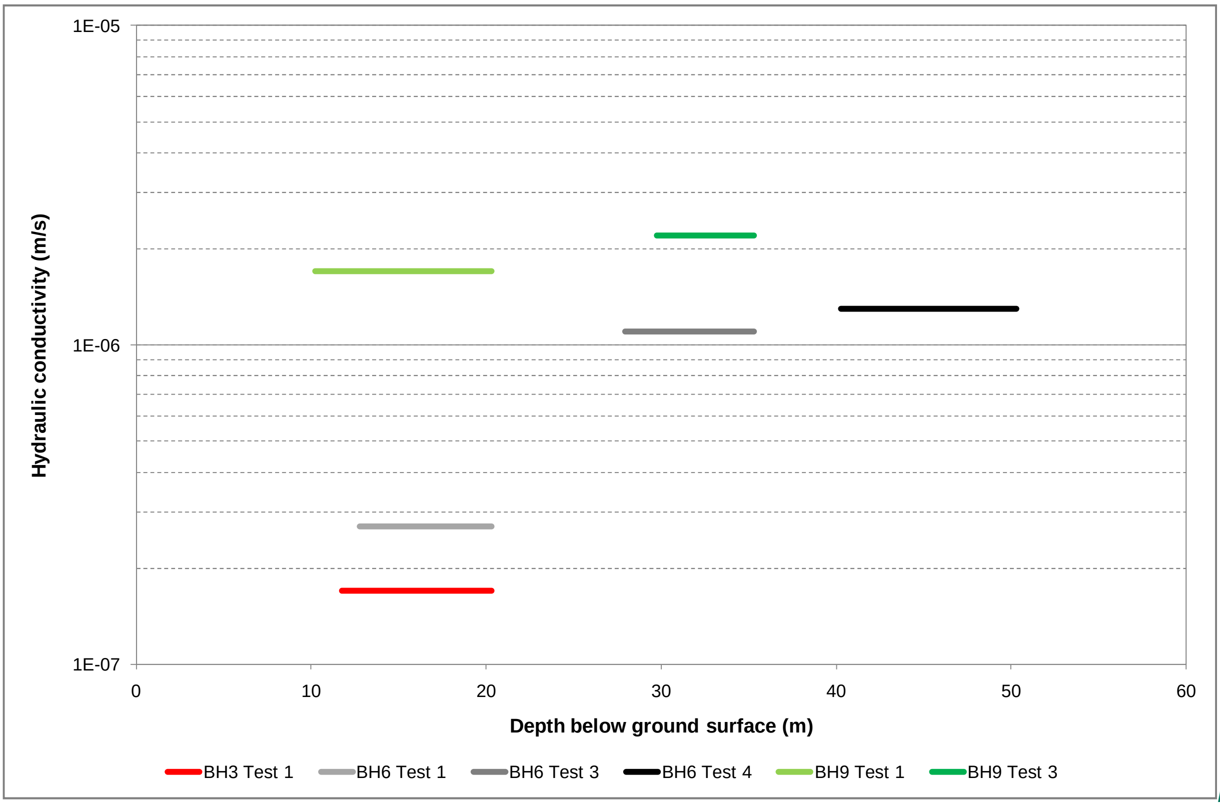


Figure B7: Hydraulic conductivity variation with depth – Back Road WRS area (Golder 2009).

APPENDIX C

Digital Model Structure

1.0 INTRODUCTION

The groundwater model used to simulate seepage flows and contaminant losses from the various waste storage areas at the MGP has been based on an existing series of models constructed to support past consent applications (Kingett Mitchell 2005). The model structure and the calibration and validation processes are documented in that report and parts of the model structure section from that report have been incorporated into this appendix.

2.0 SOFTWARE

The Visual MODFLOW Pro software package from Waterloo Hydrogeologic Inc. was utilised to generate the groundwater model.

Generation of the groundwater flow field in the model and physical flow calibration procedures were performed using MODFLOW 2000 public domain code from the United States Geological Survey. Using MODFLOW 2000 for the groundwater flow calculations, the conjugate gradient solver (PCG) with the block centred flow package was applied. The Cholensky pre-conditioning method was applied. The head change and residual criteria for a successful solution at each calculation stage were set at 0.1 and 864 respectively.

Mass transport was calculated utilising MT3D99, a software package proprietary to S. S. Papadopulos & Assoc. Inc. The advection term was solved using the Upstream Finite Difference Method and dispersion was solved with the implicit generalised conjugate gradient solver.

3.0 MODEL STRUCTURE

3.1 Model series and model continuity

3.1.1 Previous Models

In the models that were developed in support of past consent applications (Kingett Mitchell 2005) the minesite simulation was generated through a series of models, each of which simulated a distinct period before, during or after the operational phase of the MGP (Table C1). Each groundwater flow model in the series stood alone, with no linking of the models in the series beyond the basic geometry, boundary and hydraulic parameters.

The continuity of the mass transport simulation through the series of models was achieved through copying the contaminant mass load from each cell at the end of one model run and applying those loads to the corresponding model cells at the start of the following model run (Kingett Mitchell 2005).

Model 1988, which simulated the site prior to the start of mining operations, was a steady state groundwater flow model only. Contaminants were not introduced as there were no active contaminant sources related to the MGP operations during this period. Model 1988 was used for initial calibration of the natural hydrogeological features at the MGP against observations made of the piezometric surface at that time.

The steady sate Model 1998 was the primary calibration model for the MGP site simulation. Calibration was performed against a range of data collected by OceanaGold at the site. These observations include piezometric data, tailings embankment drain flows and contaminant breakthrough curves in monitoring wells (Kingett Mitchell 2005).

Table C1: Modelling sequence for Macraes Gold Project past models.

Model	Days		Concentrate Tailings		Flotation Tailings		Southern Pit	
	Simulation	Model	Start	End	Start	End	Start	End
1988	-	-	-	-	-	-	-	-
1998	2937	2937	29/11/1990	14/12/1998	27/02/1991	14/12/1998	-	-
2002	4066	1129	14/12/1998	16/01/2002	14/12/1998	16/01/2002	-	-
2005	5234	1168	16/01/2002	29/03/2005	16/01/2002	29/03/2005	16/01/2002	29/03/2005
2015	9163	4035	-	-	29/03/2005	31/12/2015	29/03/2005	31/12/2015
2165	63950	54787	-	-	31/12/2015	31/12/2165	31/12/2015	31/12/2165

Model 2002 was the final steady state calibration model and initial validation model. The final hydrogeological parameters applied to the calibrated Model 1998 were applied to Model 2002. Model 2002 was then used to determine if these parameters, when applied to a different minesite geometry, produced accurate contaminant breakthrough curve predictions for the period from December 1998 to December 2002. Model 2002 was also used to determine if sequential mass transport models based on steady state groundwater flow simulations could be used to generate seamless contaminant breakthrough curves. The end of the Model 2002 simulation period coincided with the start of tailing deposition in the Southern Pit tailings impoundment.

The steady state Model 2005 was the final validation model, with the accuracy of breakthrough curve predictions determined by comparison with the water quality observations from monitoring wells and from Deepdell Creek during this period. The end of the Model 2005 simulation period coincided with the incorporation of the Concentrate Tailings storage area in the Mixed Tailings impoundment, with a consequent change in recharge rates and chemistry (Kingett Mitchell 2005).

Model 2015 was based on the projected minesite layout at closure of the MGP as planned in 2005. This transient model was used to predict mass loading to Deepdell Creek, The NBWR and several tributary catchments during the operational period of the mine.

Model 2165 was also based on the projected minesite geometry at closure of the MGP, however, groundwater recharge to the tailings impoundments was reduced to 32mm/yr. This transient simulation enabled the groundwater table within the TSF's at the site to decline gradually following closure. The tailings pore water and the associated contaminants that moved out of the tailings mass during the post-closure recovery period discharged either to the drainage systems built into the TSF's or into the underlying rock mass.

3.1.2 Current Model

The aim of the current modelling is to simulate groundwater flow and mass transport projections through to the end of the planned mine life in 2020 followed by a 150 year post-closure period. The site layout simulated in this model is based on the currently projected minesite layout at closure of the MGP. The SP11 tailings mass has been removed and the Back Road TSF is in place. Although the timing of these two changes is not accurate from the point of view of the planned mine staging, the outcomes from the simulation of contaminant mass loads over the future life of the mine are not substantially affected.

The Model 2015 grid layout was used as the basis for developing the current model. Model 2015 is based on the projected minesite layout at closure of the MGP as planned in 2005. This layout is very similar to the minesite layout in 2010 with the following exceptions:

- The MTI and SPI tailings elevations in Model 2015 were set at 551 mRL, as was originally planned for these TSF's. However, the current TSF elevations are approximately 540 mRL.

- The size of the Frasers Pit in Model 2015 is somewhat larger than its current size.

These differences are not expected to have a major effect on the outcomes of the model series.

Initial groundwater conditions (hydraulic heads and concentrations of chemical species) for the current model were assumed to be approximately equal to heads and concentrations in Model 2015 halfway through the simulation (i.e. five year after the beginning of the simulation).

Contaminant concentrations applied to the recharge water in the current model, both during the operational period of the mine and following closure, are based on existing concentrations in the tailings decant ponds and in the impoundment and waste rock stack drainage systems. Although it is expected that the water quality in the tailings seepage water would improve over the 150 year post-closure period simulated, it is not clear how rapidly this improvement would occur. For that reason the post-closure concentrations applied to the tailings mass as well as the waste rock stacks in the simulation do not decrease over time.

3.2 Model Grid

The model grid has been constructed based on the MGP mine grid, with the grid limits defined to incorporate all topographic features likely to affect groundwater flow in the immediate vicinity of the mine workings (Figure C1). The grid limits in metres are:

X axis 66000 E to 73000 E **Y axis** 10000 N to 19000 N

The background grid spacing for both columns and rows was set at 100 m. At this spacing some important features of the mine site are considered to be inadequately represented in the mathematical model.

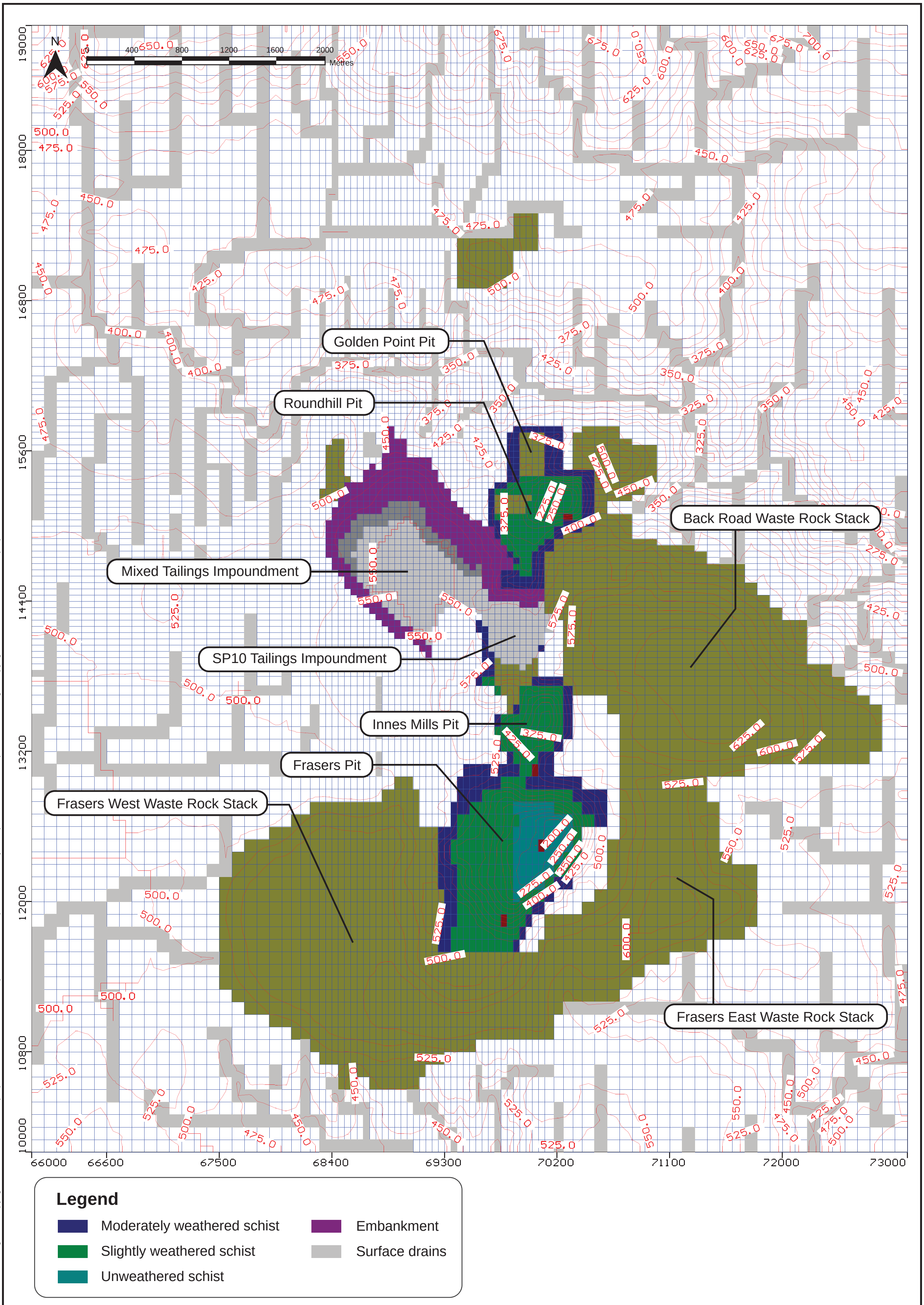
In order to improve feature definition for the tailings impoundments and opencast pits, the column and row spacing has been decreased to 50 m across these areas. An abrupt interface between the larger cells representing most of the mine area and the smaller cells in the core of the model may have resulted in model calculation problems. In order to reduce this risk an intermediate zone around the core, where grid spacing has been set at 75 m, was defined (Table C2).

Table C2: Model grid cell spacing.

Axis	From (m)	To (m)	Grid spacing (m)
East	66000	68000	100
	68000	68300	75
	68300	70100	50
	70100	70400	75
	70400	73000	100
North	10000	13500	100
	13500	13800	75
	13800	16200	50
	16200	16500	75
	16500	19000	100

3.3 Layer Structure

The MODFLOW model is constructed from stacked layers which are continuous across the whole extent of the model. The layer structure of the model of the MGP has been generated to simulate the geometry of two specific features at the site:



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TITLE MODEL GRID STRUCTURE

MARCH 2011

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C1

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- Four weathering zones, which are defined as having lower boundaries parallel to the ground surface.
- The Hyde Macraes Shear Zone, which is oriented at an oblique angle to the ground surface and intersects both the ground surface and the base of the model.

In order to generate a model adequately representing the HMSZ this feature had to be inserted into the model as three layers, defined for the purposes of this model as layers 5, 6 and 7. To achieve this four layer boundaries were defined and incorporated into the model separately (Figure C2).

Above the top of the HMSZ four layers, 1, 2, 3 and 4, have been defined to simulate the weathering profile of the hanging wall rock mass. A further three layers, 8, 9 and 10, have been created below the base of the HMSZ to simulate the weathering profile of the foot wall rock mass.

A minimum layer thickness of 10 m had to be stipulated in order to ensure the mathematical validity of the model was maintained. A smaller thickness tended to result in individual cells becoming isolated from other active cells during the calculation process, generally through dewatering of the majority of the surrounding cells. Where this occurred calculation of the model either ceased or no convergence to a unique solution for the flow equations was achieved.

It was not possible for each layer to be represented by a single combination of hydraulic parameters, mainly due to the compression of layers against the HMSZ. As a consequence, several sets of hydraulic parameters representing different geological features are represented in most layers.

3.4 Site Layout

Mining related features at the minesite have been simulated through updating the surface topography in models of later mine stages while maintaining the structure of the underlying units generated during construction of the pre-mining model. This resulted in an increased thickness of the topmost layer where impoundments and waste storage areas had been constructed. Where mine pits had been excavated the minimum layer thickness requirement resulted in the near surface layers being modified around the outside of the pit.

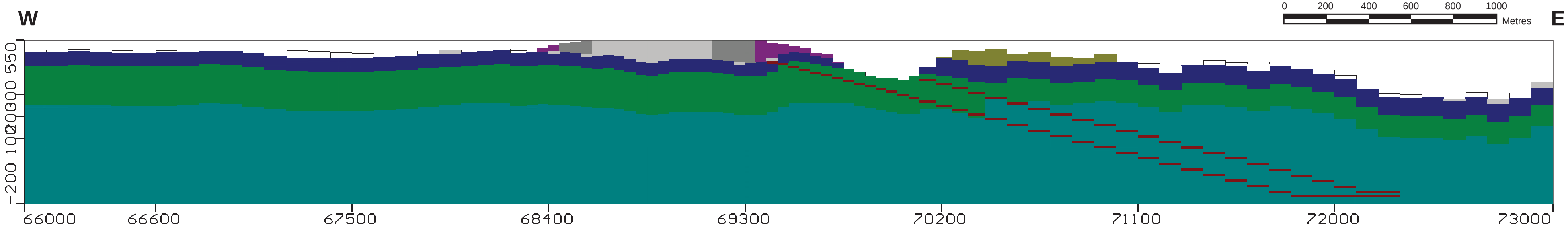
Tailings impoundments and their contents, water storage impoundments and waste rock stacks have been incorporated in the models through increasing the thickness of the uppermost layer and modifying the localised hydrogeological parameters to reflect the feature.

Given the scale of the model, small or thin features that may have a significant effect on groundwater flow at only a localised level have been combined with other units. Where this has been done the hydraulic characteristics of the combined units have been averaged and the result applied to the model.

Groundwater flow toward each pit is induced through the definition of drain cells within the pits. The controlling groundwater elevation defined for each drain cell is the same as the rock surface elevation within that cell.

Tailings impoundments have been defined through localised increases in the surface elevation within the limits of the impoundments.

Recharge to the surface of the tailings impoundments is substantially increased above the background rate to ensure the tailings remains fully saturated. The piezometric levels within the tailings impoundments are controlled through the definition of drainage cells at the beck of the impoundments to simulate the decant water ponds. These drainage cells have the piezometric level for active drainage set at the tailings decant pond elevation relevant to the closure date for each model.



Legend

- Weathered schist
- Moderately weathered schist
- Slightly weathered schist
- Unweathered schist
- Fine tailings
- Coarse tailings
- Embankment
- Waste rock
- Hangwall shear/footwall fault

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TITLE TYPICAL EAST WEST GEOLOGICAL SECTION

MARCH 2011

PROJECT 0978110562

C2

APPENDIX D

Groundwater Model Calibration and validation

1.0 INTRODUCTION

Groundwater monitoring wells at the MGP are distributed around the tailings impoundments and in the area between the Mixed Tailings Impoundment (MTI) and Macraes Flat township (Figure D1). Ongoing sampling and analysis of groundwater from many of these monitoring wells has resulted in a large groundwater quality database being available for model review purposes. The Kingett Mitchell groundwater flow and contaminant transport model was partially calibrated against water quality data from monitoring wells located in Maori Tommy Gully (MTG) to the north of the MTI (Figure D1)

Four monitoring wells designated as detection wells (GW46, GW47, GW48 and GW49) are located close to the MTG invert, between the MTI and the MTG silt pond. These monitoring wells were installed to act as early detection wells for tailings seepage water being transported in the groundwater system down MTG. All four of the detection wells are located within the extent of one grid cell of the MODFLOW model. For calibration purposes, the concentrations of several groundwater quality parameters averaged across the four detection wells were compared to the quality of the groundwater within the appropriate model cell.

Eight groundwater monitoring wells were installed between the MTG silt pond and Deepdell Creek. These eight wells (GW18 through to GW25) were designated as compliance wells. All eight compliance wells are located within the extent of one grid cell in the MODFLOW model. Changes in the simulated groundwater quality in this single model cell over time have been compared to the measured concentrations of several groundwater quality parameters averaged across the eight compliance wells.

In addition to the detection and compliance wells, other monitoring wells installed in and around the Maori Tommy catchment have been sampled regularly since deposition of tailings in the MTI was initiated. These monitoring wells include:

- GW43, GW52 and GW53, which are located on the slopes of MTG down-gradient from the MTI.
- GW02, GW38, GW03 and GW04, which are located in an arc from south to north around the western side of the MTI.
- Paired monitoring wells GW30 / GW31 and GW32 / GW33, which are each located immediately to the northwest of the MTI.
- GW01, which is generally considered to be an up-gradient background monitoring well for the site. The groundwater quality near this well may however have been be influenced by MGP operations (Golder 2010a).

2.0 MAORI TOMMY GULLY DETECTION WELLS

The outcomes from the updated contaminant transport model have been compared to observed groundwater quality data from the detection wells located in MTI. The comparison has only been undertaken for sulphate (Figure D2), magnesium (Figure D3) and calcium (Figure D4). To date there is no indication of increasing concentrations of either arsenic or cyanideWAD in groundwater at the detection wells. The results for chloride and iron are complicated by evidence that neither contaminant is transported on a conservative basis within the groundwater system.

On average, the updated model overstates the concentrations of conservatively transported contaminants by between 20% and 25% during the period 2005 to 2009. The measured contaminant concentrations in the detection wells do not appear to have increased substantially since 2006. Modelled concentrations are however continuing on an upward trend. This difference between observed and measured concentrations indicates that the simulated contaminant loads being transported in groundwater past the line of detection wells in MTG is a conservative representation of the actual contaminant loads.

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TITLE LOCATION MAP OF GROUNDWATER MONITORING WELLS.

MARCH 2011

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D1

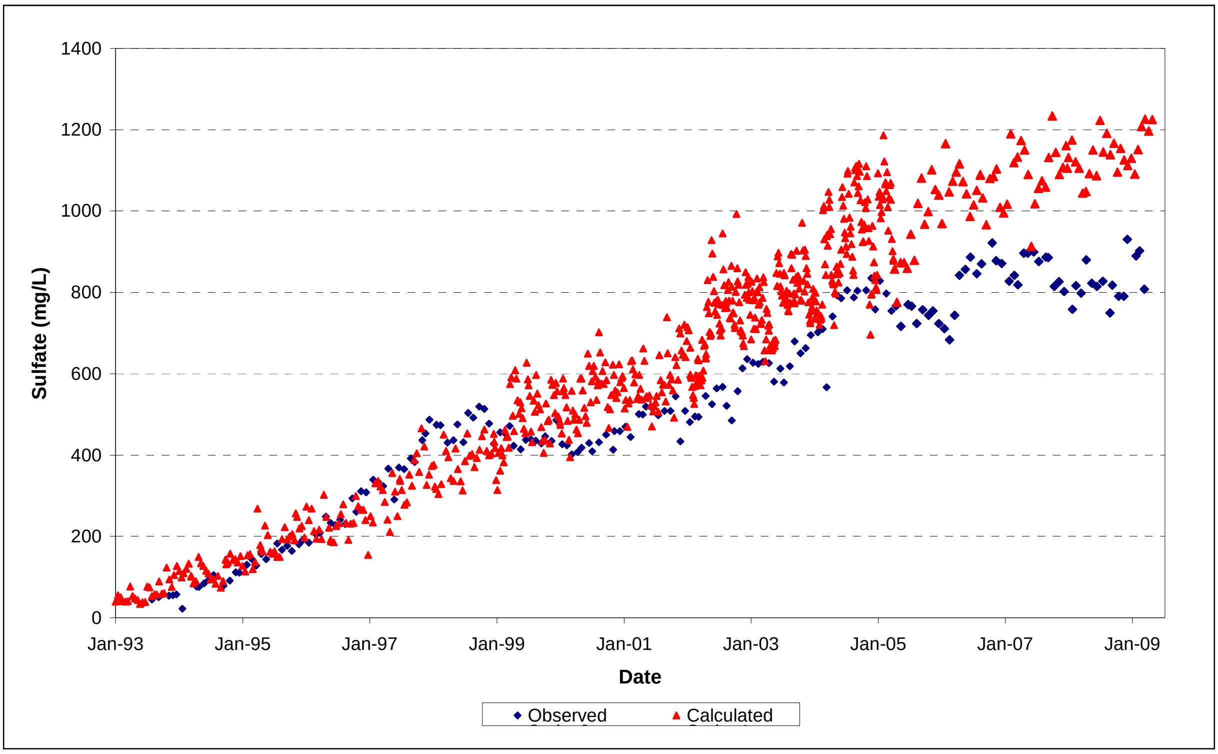

Figure D2: Detection wells observed and simulated sulfate breakthrough curves.

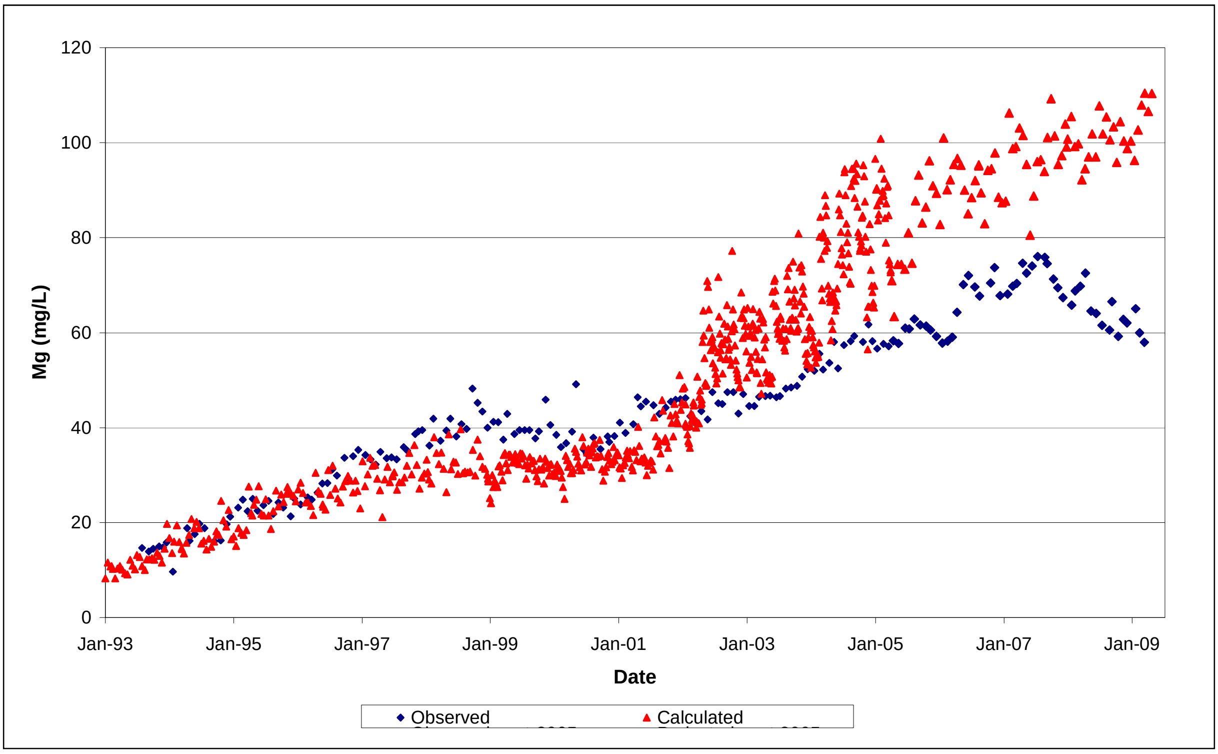

Figure D3: Detection wells observed and simulated magnesium breakthrough curves.

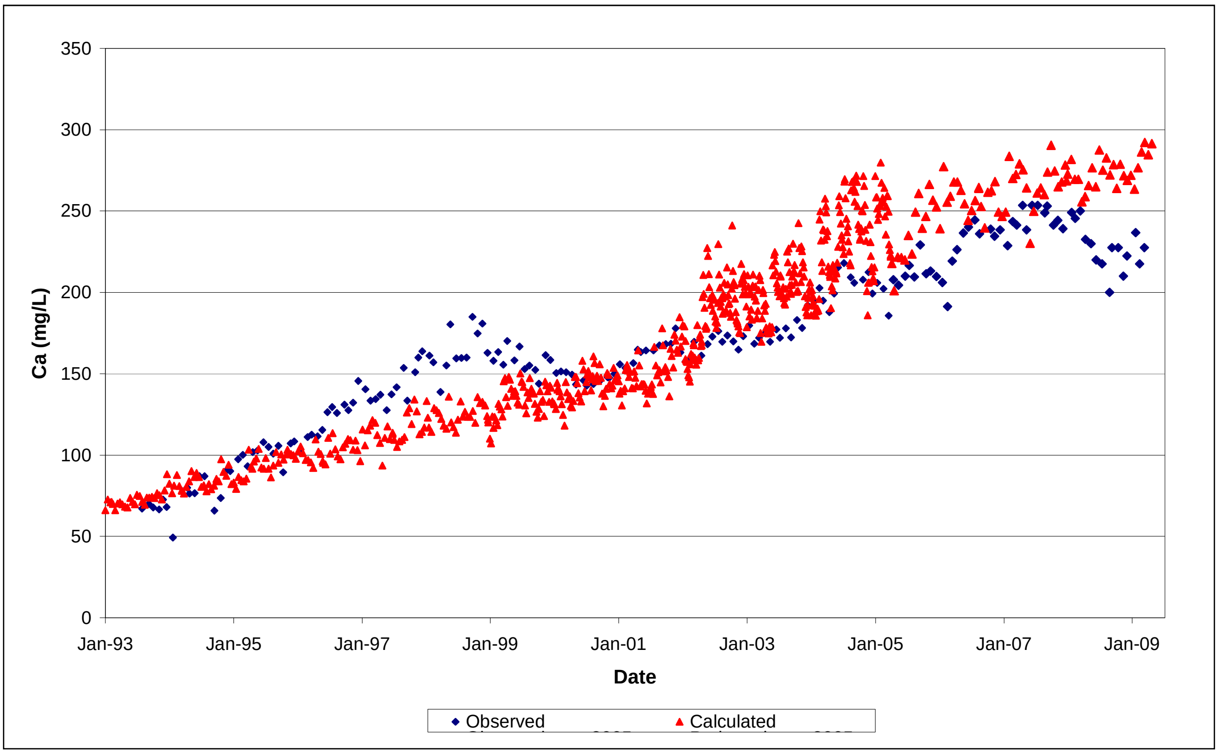

Figure D4: Detection wells observed and simulated calcium breakthrough curves.

3.0 MAORI TOMMY GULLY COMPLIANCE WELLS

The outcomes from the updated contaminant transport model have been compared to observed groundwater quality data from the compliance wells located in MTI. The comparison has again only been undertaken for sulphate (Figure D5), magnesium (Figure D6) and calcium (Figure D7). As with the detection wells, there is no indication to date of increasing concentrations of either arsenic or cyanideWAD being detected in groundwater at the compliance wells. Although the concentrations of chloride in the compliance wells have varied over time, there analysis results do not indicate a long term trend of increasing concentration.

The contaminant transport model results indicated increasing concentrations of sulfate, magnesium and calcium should have been detectable at the compliance wells as early as 2001. This did not occur and increases in contaminant concentrations only became detectable during 2006.

For both sulphate (Figure D5) and calcium (Figure D7) the rate of increase for the concentrations detected were greater than were indicated from modelling. By 2009 the simulated and observed concentrations were similar. It is expected that one reason for the differences in the contaminant breakthrough curves was the use of a longitudinal dispersivity factor of 10 in the modelling. In effect, the application of this factor results in the breakthrough curve becoming extended over a longer period, with the initial breakthrough occurring earlier than would be the case had a smaller dispersivity factor been applied. The long term contaminant loads discharging to the receiving waters are not greatly affected by the dispersivity factor.

The magnesium breakthrough curve indicates the simulated concentrations are substantially greater than the observed concentrations, indicating magnesium may not be conservatively transported within the groundwater system between the MTI and the compliance wells.

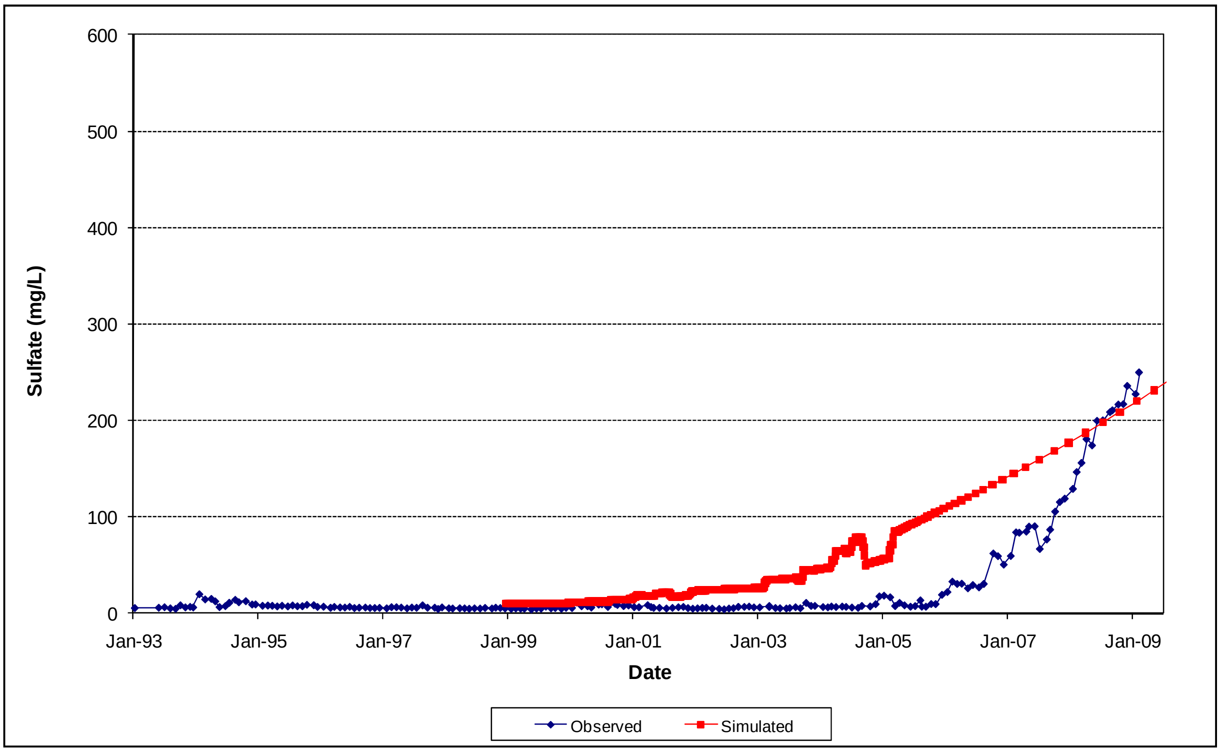

Figure D5: Compliance wells observed and simulated sulfate breakthrough curves.

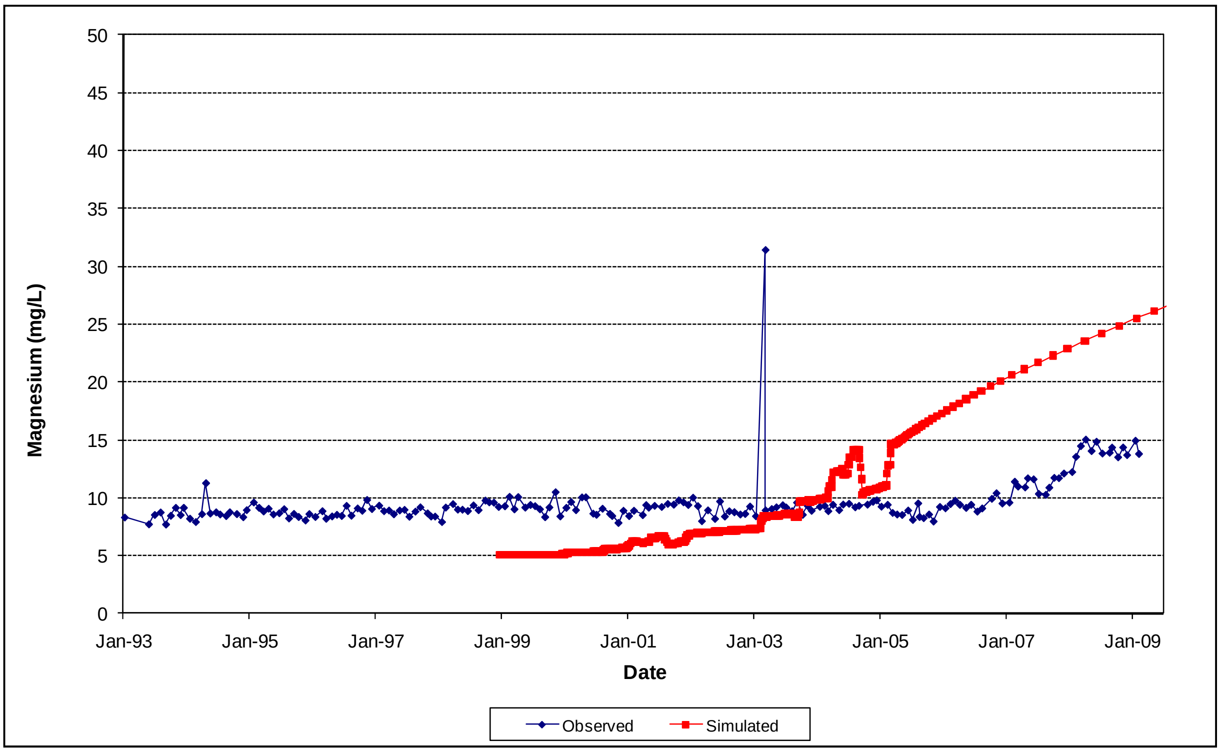

Figure D6: Compliance wells observed and simulated magnesium breakthrough curves.

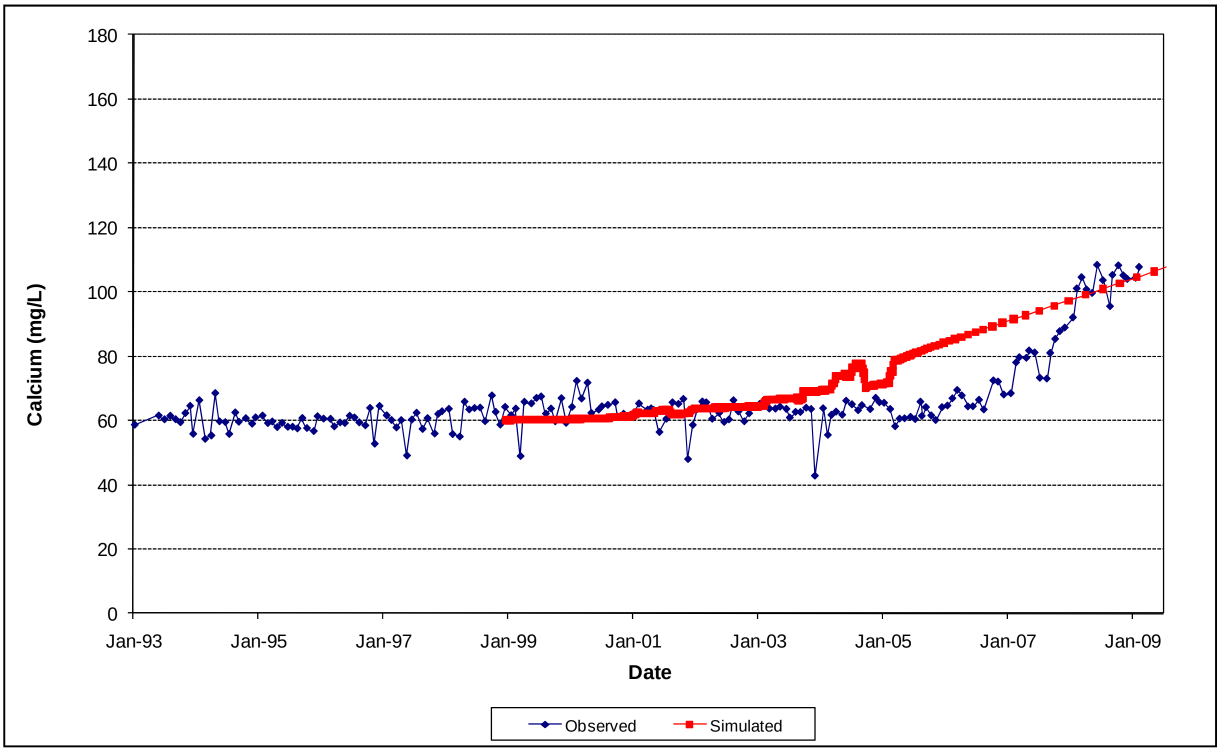

Figure D7: Compliance wells observed and simulated calcium breakthrough curves.

4.0 OTHER MONITORING WELLS

The four groundwater wells GW2, GW3, GW4 and GW38 located to the west of the MTI (Figure D1) have been regularly monitored for water quality since 1990. There is no indication from the OGL water quality monitoring records that tailings seepage water has resulted in substantial changes to the quality of the groundwater at any of these wells. This appears to support the indication that a partially infilled gully located immediately to the west of the MTI is acting as a groundwater discharge point and limiting the extent to which tailings seepage water is being transported in groundwater toward the west from the MTI. As the contaminant transport model does not specifically include this gully as a drainage system, the model indicates some transport of contaminants toward the west from the MTI. In this case the model is overstating potential losses of contaminants toward the west.

Measured concentrations of sulfate, magnesium and calcium in the paired monitoring wells GW30 and GW31 reflect background water quality in this locality. The contaminant transport model indicates concentrations should have gradually increased during the monitoring period. In the model series it had been predicted that the three chemical species would gradually increase over time due to contaminants moving from the mixed tailings dam but this trend did not exist in the field measured values.

Measured concentrations of sulfate, magnesium and calcium in the paired monitoring wells GW32 and GW33 differ significantly. The concentrations detected in GW32 reflect a strong influence of tailings seepage water whereas the quality of the water in GW33 is indicative of background water quality in this area. The two monitoring wells are screened at different depths below ground level however the reasons for the difference in water quality have not been specifically investigated to date. As such, these results have not been compared to the results from the contaminant transport modelling. The infilled gully mentioned above is located to the west of GW32 and GW33. It is therefore expected that seepage losses in this area would not be transported past this gully.

There is no indication, either simulated or observed, that contaminants from the MTI are being transported toward the Macraes Township, past monitoring well GW1. Minor increases in sulphate concentrations observed in this monitoring well are expected to be a result of other activities at the site (Golder 2010a).

5.0 REFERENCES

Golder, 2010a: Macraes Gold Project: Water quality and contaminant load model inputs. Report prepared for OceanaGold New Zealand Limited by Golder Associates (NZ) Limited, March 2010.

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APPENDIX E

Groundwater inputs for Mine Water Management Model

1.0 GROUNDWATER FLOW RATES

Simulated groundwater flow rates discharging to surface receiving waters, silt ponds and opencast pits at the MGP site are summarised on a stage by stage basis in Tables D1 through to D5. The presence of groundwater flowing into the catchments upstream from the silt ponds does not necessarily imply contaminants from WRS's or TSF's are also discharging to these receiving waters during the same stage. In some cases the groundwater can be expected to be at background concentrations for the areas upstream structures may not yet have been installed during the relevant stage.

Table D1: TSF drainage systems discharge rates.

TSF Drainage Systems	Operational Stage (1,2, 3)				
	0	1	2	3	4
Combined MTI and SPI	2,500	1,000	800	600	300
TTTSF (4)	0 (5)	1,800	1,800	1,800	260

Notes: 1) All values presented in units of m^3/day.
2) Refer to Section 1.2.4 for stage descriptions.
3) The rates at which tailings discharges to the TSF drain systems decline have been calculated based on observed rates of decline (Golder 2010c) rather than from the rates of decline simulated in the groundwater models.
4) The seepage water discharge rates to the TTTSF drain systems has been sourced from Golder 2010a and included here for completeness.
5) TSF not yet constructed.

Table D2: Roundhill/Golden Point/Southern Pit groundwater inflows.

Pit Water Elevation (mRL)	Operational Stage (1,2)				
	0	1	2	3	4
220	160	160	178	178	178
340	160	160	156	156	156
400	160	160	134	134	134
430	160	160	90	90	90
530	160	160	90	90	90
580	160	160	90	90	90

Notes: 1) All values presented in units of m^3/day.
2) Refer to Section 1.2.4 for stage descriptions.

Table D3: Frasers / Innes Mills Pit groundwater inflows.

Pit Water Elevation (mRL)	Operational Stage (1,2)				
	0	1	2	3	4
190	685	685	685	1275	1275
250	685	685	685	318	318
430	685	685	685	248	248
480	685	685	685	NA (3)	52
520	685	685	685	NA (3)	52

Notes: 1) All values presented in units of m^3/day.
2) Refer to Section 1.2.4 for stage descriptions.
3) Water level managed due to need to operate in Innes Mills pit.

Table D4: Silt pond groundwater inflows.

Silt Pond/Sump	Operational Stage (1,2)				
	0	1	2	3	4
Deepdell North Silt Pond (3)	0	0	0	0	0
Deepdell South Silt Pond (3)	0	0	0	0	0
Maori Tommy Gully Silt Pond	70	70	70	70	70
Battery Creek Silt Pond (3)	0	0	0	0	0
Northern Gully Silt Pond	70	70	70	70	40
Frasers West Silt Pond	150	150	150	150	54
Murphys Creek Silt Pond	24	24	24	24	122
Back Road Silt Pond	0 (4)	0 (4)	65	65	65
Tipperary Sump (5)	0 (4)	140	140	140	30
Tipperary Creek Silt Pond (5)	0 (4)	5	5	5	5

Notes: 1) All values presented in units of m^3/day.
2) Refer to Section 1.2.4 for stage descriptions.
3) Simulated silt ponds do not receive groundwater discharges. This is supported by observations at Deepdell South and Battery Creek however may not be the case for Deepdell North.
4) Silt pond not yet constructed.
5) The groundwater rates to silt ponds in the Tipperary Creek catchment has been sourced from the Golder 2010a report and included here for completeness sake.

Table D5: Receiving water groundwater inflows.

Receiving Water	Operational Stage (1,2,3)				
	0	1	2	3	4
Deepdell Creek at DC07	730	730	730	730	730
Deepdell Creek at DC08	BG (3)	BG (3)	116	116	116
Cranky Jims Creek at CJ01 (4)	BG (5)	54	54	54	54
Tipperary Creek at TC01 (4)	BG (5)	440	440	440	660
Murphys Creek at MC100	180	180	180	180	180
North Branch Waikouaiti River at NBWRRB	100	100	100	100	100

Notes: 1) All values presented in units of m^3/day.
2) Refer to Section 1.2.4 for stage descriptions.
3) Background groundwater status as construction on Back Road WRS not yet initiated.
4) The groundwater rates to receiving waters in the Tipperary Creek catchment has been sourced from the Golder 2011a report and included here for completeness sake.
5) Background groundwater status as construction on TTTSF not yet initiated.

2.0 GROUNDWATER QUALITY

Average contaminant concentrations above background concentrations in groundwater discharging to surface receiving waters, silt ponds and opencast pits at the MGP site are summarised on a stage by stage basis in Tables D6 through to D14. The presence of groundwater flowing into the catchments upstream from the silt ponds does not necessarily mean contaminants from WRS's or TSF's are also discharging to these receiving waters during the same stage. In some cases the groundwater can be expected to be at background concentrations for the area as upstream structures may not yet have been installed during the relevant stage.

Table D6: Quality of groundwater discharging to opencast pits and pit lakes.

Parameter [1]	Frasers Pit/ Innes Mills	Round Hill/Golden Point/ Southern Pit	Deepdell South Pit
Arsenic	0.057	0.43	<0.001
Sulphate	450	900	5
$Cyanide_{WAD}$	0.4	1.18	<0.001
Copper	0.001	0.003	<0.001
Iron	2.4	8.06	0.002
Lead	<0.001	0.001	<0.001
Sodium	18.7	51.7	0.13
Potassium	1.8	4.09	0.027
Calcium	83	220	0.99
Magnesium	40	118	0.81
Zinc	0.004	0.009	<0.001
Chloride	13.8	40.8	0.023

Notes: 1) All values presented in units of g/m^3.
2) Refer to Section 1.2.4 for stage descriptions.

Table D7: Quality of groundwater seepage discharging upstream from existing and proposed silt ponds.

Parameter (1)	Deepdell North	Deepdell South	Battery Creek	Northern Gully	Frasers West	Murphys Creek	Back Road	Tipperary Creek (2)
Arsenic	0.011	<0.001	<0.001	0.001	0.001	<0.001	<0.001	<0.001
Sulphate	38.6	15.4	1,200	2,300	1,500	1,320	2,200	100
$Cyanide_{WAD}$	<0.001	<0.001	<0.001	0.013	<0.001	<0.001	<0.001	<0.001
Copper	0.001	<0.001	<0.001	0.003	0.001	0.001	0.002	<0.001
Iron	0.170	<0.001	<0.001	0.93	0.42	0.52	0.91	0.037
Lead	0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Sodium	16	10.8	36	57.7	26.2	33	56.4	2.34
Potassium	5.5	1.89	10	12	5.47	6.69	11.7	0.49
Calcium	68.5	40.2	290	434	199	250	425	17.6
Magnesium	14.5	7.7	200	360	165	204	352	14.7
Zinc	<0.001	<0.001	<0.001	0.032	0.015	0.018	0.032	<0.001
Chloride	11.5	7.1	10	10.2	4.69	6.03	9.99	0.42

Notes: 1) All values presented in units of g/m^3.
2) The quality of groundwater discharging to silt ponds in the Tipperary Creek catchment has been sourced from the Golder 2011a report and included here for completeness sake.

Table D8: Quality of groundwater seepage discharging upstream from TSF primary silt ponds.

Parameter (1,2)	Tipperary Sump (3,4)		Maori Tommy silt pond (3,4)	
	Operational Stage 0,1,2,3	Operational Stage 4	Operational Stage 0,1,2,3	Operational Stage 4
Arsenic	<0.001	<0.001	0.98	0.98
Sulphate	100	1,500	1,270	600
$Cyanide_{WAD}$	<0.001	<0.001	1.14	1.14
Copper	<0.001	<0.001	0.001	0.001
Iron	0.035	0.035	6.92	6.92
Lead	<0.001	<0.001	0.001	0.001
Sodium	2.3	2.3	22.5	22.5
Potassium	0.45	0.45	1.04	1.04
Calcium	19.8	19.8	238	238
Magnesium	15.4	15.4	110	110
Zinc	0.001	0.001	0.001	0.001
Chloride	0.50	0.50	71	71

Notes:
1) All values presented in units of g/m^3.
2) Refer to Section 1.2.4 for stage descriptions.
3) The quality of groundwater discharging to silt ponds in the Tipperary Creek catchment has been sourced from the Golder 2011a report and included here for completeness sake.
4) All contaminant concentrations are expected to reach a maximum after closure of operations at the site. The outputs have been simplified through applying the highest modelled seepage concentrations for all parameters except sulphate to each of the operational stages. Sulphate concentrations reflect the projected maximum concentrations during the operational period and following closure.

Table 9: Quality of groundwater seepage discharging to Deepdell Creek upstream from DC07.

Parameter [1]	Stage 0	Stage 1	Stage 2	Stage 3	Stage 4
Arsenic	0.08	0.08	0.08	0.08	0.057
Sulphate	20	40	80	160	590
$Cyanide_{WAD}$	<0.001	<0.001	<0.001	<0.001	0.06
Copper	<0.0001	<0.0001	<0.0001	<0.0001	0.0007
Iron	0.47	0.47	0.47	0.47	0.47
Lead	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Sodium	<1	1	2	3	15
Potassium	0.2	0.4	0.8	1.6	3
Calcium	10	20	40	80	110
Magnesium	8	16	32	64	90
Zinc	<0.001	<0.001	<0.001	<0.001	0.008
Chloride	0.6	1.2	2.4	4.8	9

Notes: 1) All values presented in units of g/m^3.

Table 10: Quality of groundwater seepage discharging to Deepdell Creek between DC07 and DC08.

Parameter (1)	Stage 0 (2)	Stage 1 (2)	Stage 2 (2)	Stage 3 (2)	Stage 4
Arsenic	0	0	0	0	<0.001
Sulphate	0	0	0	0	1050
$Cyanide_{WAD}$	0	0	0	0	<0.001
Copper	0	0	0	0	0.0012
Iron	0	0	0	0	0.44
Lead	0	0	0	0	<0.0001
Sodium	0	0	0	0	28
Potassium	0	0	0	0	6
Calcium	0	0	0	0	203
Magnesium	0	0	0	0	167
Zinc	0	0	0	0	0.015
Chloride	0	0	0	0	5

Notes: 1) All values presented in units of g/m^3.
2) No additional contaminant load as Back Road WRS not yet constructed.

Table 11: Groundwater seepage quality discharging to Cranky Jims Creek upstream from CJ01.

Parameter (1)	Stage 0 (2)	Stage 1 (2)	Stage 2	Stage 3	Stage 4
Arsenic	0	0	0	0	0
Sulphate	0	0	290	580	580
$Cyanide_{WAD}$	0	0	0	0	0
Copper	0	0	0	0	0
Iron	0	0	2	3	3
Lead	0	0	<0.0001	<0.0001	<0.0001
Sodium	0	0	27	54	54
Potassium	0	0	2	3	3
Calcium	0	0	47	94	94
Magnesium	0	0	35	70	70
Zinc	0	0	0	0	0
Chloride	0	0	5	9	9

Notes: 1) All values presented in units of g/m^3.
2) No additional contaminant load as TTTSF not yet constructed.

Table 12: Groundwater seepage water quality discharging to the North Branch Waikouaiti River upstream from Red Bank Road.

Parameter [(1)]	Stage 0	Stage 1	Stage 2	Stage 3	Stage 4
Arsenic	0	0	0	0	0.00003
Sulphate	60	120	240	480	1,090
$Cyanide_{WAD}$	0	0	0	0	0.00007
Copper	0	0	0	0	0.0012
Iron	0.44	0.44	0.44	0.44	0.44
Lead	0	0	0	0	0.00009
Sodium	2	4	8	16	27
Potassium	0.4	0.8	1.6	3.2	6
Calcium	12	24	48	96	206
Magnesium	10	20	40	80	170
Zinc	0	0	0	0	0
Chloride	0.4	0.8	1.6	3.2	5

Notes: 1) All values presented in units of g/m^3.

Table 13: Groundwater seepage chemistry in Tipperary Creek at TC01

Parameter [1]	Stage 0 [2]	Stage 1 [2]	Stage 2	Stage 3	Stage 4
Arsenic	0	0	0	0	0
Sulphate	0	0	265	530	530
$Cyanide_{WAD}$	0	0	0.01	0.01	0.01
Copper	0	0	0	0	0
Iron	0	0	1.64	3.28	3.28
Lead	0	0	0	0	0
Sodium	0	0	27	54	54
Potassium	0	0	1.65	3.3	3.3
Calcium	0	0	47	94	94
Magnesium	0	0	35	70	70
Zinc	0	0	0.01	0.01	0.01
Chloride	0	0	4.5	9	9

Notes: 1) All values presented in units of g/m^3.
2) No additional contaminant load as TTTSF not yet constructed.

Table 14: Groundwater seepage chemistry in Murphys Creek at MC01

Parameter [1]	Stage 0	Stage 1	Stage 2	Stage 3	Stage 4
Arsenic	0	0	0	0	0
Sulphate	380	380	380	380	380
$Cyanide_{WAD}$	0	0	0	0	0
Copper	0.001	0.002	0.003	0.007	0
Iron	0.3	0.61	1.22	2.43	0.15
Lead	0	0	0	0	0
Sodium	0.5	1	2	4	10
Potassium	0.1	0.2	0.4	0.8	2
Calcium	4	8	16	32	72
Magnesium	3	6	12	24	59
Zinc	0	0	0	0	0.005
Chloride	0.1	0.2	0.4	0.8	2

Notes: 1) All values presented in units of g/m^3.

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