In-Pit Disposal of Reactive Mine Wastes: Approaches, Update and Case Study Results

MEND Report 2.36.1b

This work was done on behalf of the Mine Environment Neutral Drainage (MEND) Program and sponsored by:
The Mining Association of Canada (MAC) and MEND

June 2015

IN-PIT DISPOSAL OF REACTIVE MINE WASTES: APPROACHES, UPDATE AND CASE STUDY RESULTS

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June 2015

EXECUTIVE SUMMARY

The options for disposal of mine waste that represent low long-term liabilities are limited at a large proportion of mine sites. Although not new, the disposal of mine wastes into mined-out pits has, in recent years, received increased acceptance particularly for acid generating, metal leaching, radioactive and perceptively hazardous tailings, waste rock and water treatment sludges.

The published results for a selection of in-pit disposal options have been, in general, very positive in demonstrating the long-term isolation demanded by companies, regulators and the public. This document is an update of the 1995 MEND Report 2.36.1 and provides a summary of 12 case studies of in-pit disposal of mine waste at locations around the world. Case studies were selected to provide examples of in-pit disposal of mine wastes exhibiting a variety of environmental risks and located in various hydrogeological and climatic settings.

Some common aspects of in-pit disposal include the following constraints:

- A pit needs to be locally available and the use of the pit should not “sterilize” remaining mineral resources. In addition, pit filling above active underground mining is considered unsafe in most cases;
- In many cases, the amount of waste rock and tailings produced will not fit into the available opening, resulting in use of expensive dual waste disposal scenarios;
- Terminal water levels in a back-filled pit as well as local hydrogeological conditions greatly influence the selection of in-pit disposal options. In the case of Stratmat, in-pit disposal of acid rock drainage waste rock was initially selected but, after a detailed evaluation, was rejected in favour of surface storage; and
- In-pit disposal can be very costly in instances where significant material and pit engineering may be required to isolate mobile contaminants.

The benefits of in-pit disposal are many – the following aspects may have site specific importance:

- The isolation of solid mine wastes in an anoxic environment, which has been shown to inhibit acid generation and the release of soluble metals and other contaminants;
- The reduction or elimination for the need to maintain engineered structures;
- Improved social and regulatory acceptance of mining activities by restoring land forms and function; and, in some cases,
- The permanent isolation of hazardous substances resulting from emergencies and normal processing.

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Les options de disposition de rejets miniers qui représentent de faibles responsabilités à long terme se limitent à une grande proportion de sites miniers. Bien qu'il ne s'agisse pas d'un nouveau processus, l'élimination des rejets miniers dans les fosses des mines à ciel ouvert a fait l'objet, au cours des dernières années, d'une acceptation accrue, particulièrement pour les résidus, les roches stériles et les boues d'usines de traitement, favorisant la lixiviation des métaux, radioactifs et visiblement dangereux.

Les résultats publiés pour diverses options d'élimination dans une fosse ont été, de manière générale, très positifs pour démontrer l'isolation à long terme demandée par les entreprises, les organismes de réglementation et le public. Ce document constitue une nouvelle version du Rapport NEDEM 2.36.1 de 1995 et fournit un résumé des 12 études de cas sur l'élimination de rejets miniers dans une fosse à divers endroits dans le monde. Les études de cas ont été choisies pour des exemples d'élimination de rejets miniers dans les fosses des mines à ciel ouvert montrant une variété de risques environnementaux et situés dans différents contextes hydrogéologiques et climatiques.

Certains aspects communs de l'élimination englobent les contraintes suivantes :

- Une fosse doit être disponible localement, et l'utilisation de la fosse ne doit pas « stériliser » les ressources minérales restantes. De plus, le remplissage d'une fosse au-dessus d'une mine souterraine active est considéré comme non sécuritaire dans la plupart des cas.
- Dans bon nombre de cas, la quantité de roches stériles et de résidus produits n'entrera pas toute dans l'espace disponible, ce qui entraînera l'utilisation plus coûteuse de deux scénarios d'élimination de rejets.
- Les niveaux d'eau dans la fosse vacante et les conditions hydrogéologiques locales influencent grandement le choix des options d'élimination dans une fosse. Dans le cas de Stratmat, l'élimination des stériles découlant du drainage minier acide dans la fosse a été initialement choisie, mais, après une évaluation détaillée, cette option a été rejetée et remplacée par une halde en surface.
- L'élimination dans une fosse de mine à ciel peut être très coûteuse dans les cas où des matériaux importants et des ressources en ingénierie sont nécessaires pour isoler les contaminants mobiles.

Les avantages de l'élimination dans une fosse de mine à ciel ouvert nombreux; les aspects ci-dessous pourraient avoir une importance en fonction du site :

- l’isolation de rejets miniers solides dans un environnement anoxique, qui a démontré une répression de la production d’acide et de la diffusion de métaux solubles et d’autres contaminants;
- la réduction ou l’élimination du besoin d’entretenir des structures artificielles;
- une meilleure acceptation sociale et réglementaire des activités minières en rétablissant les formes et les fonctions des terres;
- l’isolation permanente des substances dangereuses découlant de situations d’urgence et du traitement normal.

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Marymia	Australia
Rabbit Lake	Canada
McClean Lake	Canada
Lichtenburg	Germany
Omai	Guyana

1.0 INTRODUCTION AND BACKGROUND

Permanent mine waste disposal remains a significant environmental, financial and permitting challenge for proposed, operating and inactive mines and mine sites. These challenges are magnified if the mine waste is acid generating, toxic metal and non-metal leaching, or contains elevated levels of hazardous substances such as radioactivity. Physical containment of these mine wastes using engineered structures such as tailings embankments or steep-sided large waste rock piles may represent a long term stability concern. In addition, there is a substantial proportion of public opinion that suggests that mine wastes should not be left as a visible surface legacy and mine openings to surface should be backfilled. Unsurprisingly, the public, regulators, land users and Non-Government Organizations (NGO's) often select the disposal of the mine wastes back into the mine openings as the preferred mine site rehabilitation strategy.

Unfortunately, in most cases the total volume of mine waste at a particular hard rock mine will not fit back into the mine openings, whether they are underground excavations, open pits or both. In addition, safety and cost considerations determine why most mines, even those shipping out a large proportion of the mined rock as product (i.e. coal and industrial minerals operations), develop long term surface storage facilities for process tailings and waste rock. Most operations need somewhere to put their mining wastes during production and there is often no availability of an in ground storage until mining is essentially complete. Purpose built pits and the relocation of materials later in the mine life are expensive options that most mining operations cannot support.

The disposal of mine wastes into underground mine openings has been practiced for a long time. Depending on underground mining methods, mill tailings and waste rock are often used as backfill for mine wall support during operations. Upon closure, inactive shafts, ventilation raises and mined out cavities are frequently used for the disposal of a variety of mine wastes – e.g., tailings, treatment sludges, and industrial refuse. In some cases, hazardous materials have been stored or disposed in underground, mined-out cavities.[1]

Mined out open pits are also used for waste disposal, including tailings, industrial wastes, municipal garbage, contaminated soils and debris, etc. The practice has been dictated by the convenience of a local, unoccupied depression and was rarely motivated by environmental considerations.

1 Thompson, N, Spencer, P, 2002, Management of Arsenic Trioxide Dust at Giant Mine, Department of Indian Affairs and Northern Development, Yellowknife, NT.

Recently, Pearman 2009[2] outlined some innovative uses for open pits – ranging from recreation and cultural centres to bioreactors and disposal facilities for reactive mine wastes. In the cases outlined by Pearman, pits were transformed from liabilities to assets.

The focus of this report is to update the success of using open pits for the disposal of reactive mine waste. MEND Report 2.36.1 (1995)[3] summarised in-pit disposal practices for reactive mine waste. This report is an update on the practice and 12 Case Studies are presented illustrating examples of pit disposal around the world. A few of the studies provide an update to previous (1995) results.

Each case study represents a unique combination of in-pit techniques, mine waste and waste containment under a wide spectrum of climatic, geographic and regulatory conditions. The pits vary in geometry, elevation, hydrogeological and rock conditions, connection to underground openings, etc. Mine-generated solid wastes are also site-specific and their properties vary from inert to reactive and hazardous minerals and chemical products.

Mine wastes that may be suitable for disposal in a mined out pit can be generally categorised as follows:

- Mine waste rock. Waste rock is coarse material and is only broken down in the mining process to a size which is convenient to transport. This rock can be classified according to reactivity, with acid generating, and metal leaching rock requiring the highest degree of management. Open pit mining typically produces a significantly larger quantity of waste rock than ore.

- Mill tailings. Tailings are generally discharged from a mill as slurry. The environmental risk from the tailings depends on mineral content and reactivity, the effect of processing and chemical additions during the process and on discharge, and local geographic restraints.

- Water treatment sludges. The predominant sludge types are those resulting from lime treatment of acidic drainage. Other treatment sludges are produced from water clarification, or arsenic, radioactivity and heavy metal removal. These sludges are usually combined with tailings or are disposed separately. Treatment sludges frequently contain a low percentage of solids and consolidate very slowly.

- Miscellaneous wastes such as contaminated soils, sewage, scrap materials, and industrial solid wastes.

[2] Pearman, G., 2009, “101 Things To Do With a Hole in the Ground”, Post Mining Allowance.
[3] MEND Report 2.36.1, 1995, Review of In-Pit Disposal Practices for the Prevention of Acid Drainage.

The incentives for using pits for disposal of mine wastes include the following aspects:

A. <u>Environmental</u>

- Prevention and control of acid generation;
- Reduction of metal leaching and isolation of dissolved metals;
- Permanent physical isolation of wastes;
- Minimization of the need for engineered control systems and long term monitoring;
- Return of waste rock and tailings to the original geochemical conditions; and
- Restoration of pre-mining conditions.

B. <u>Physical</u>

- Stabilization of pit walls;
- Elimination of potential accidental release of solids;
- Reduction of long term waste management care and maintenance; and
- Elimination of potential for unauthorized removal of wastes.

C. <u>Financial, Legal and Social</u>

- Potential lower cost[4];
- Potential earlier return of the land to previous and traditional uses;
- Improved acceptability by various stakeholders including permitting agencies; and
- Improved site aesthetics on closure.

In theory, wastes disposed in pits in anoxic conditions, such as those below a water table, are not subject to further oxidation. Soluble contaminant release may be reduced by either making diffusion the controlling transport mechanism or by reducing the hydraulic gradient across the pit to as low as possible, thus restricting convective outflow. Another way of restricting transport is to render the wastes relatively impermeable by "engineering" physical properties such as by maximizing consolidation. In general, controlling oxidation greatly reduces the potential for the spread of contamination.

Pit disposal of mine wastes is not always the best strategy because of mining cycle, safety, hydrological aspects and other site specific aspects:

- Mining cycle restrictions. A pit cannot be used for waste disposal until mining is completed and the mineral resource exhausted. Some mine operations are composed of only one pit or a series of connected pits. In this case, surface disposal of wastes needs to be established.

[4] Net Present Value costs, including operating, bonding, decommissioning and closure.

- Safety restrictions. Mining frequently progresses from open pit mining to underground mining beneath or connected to the pit. For safety reasons, waste disposal is usually not permitted above active workings. In addition on closure, crown pillar instability can compromise in-pit disposal integrity.

- Hydrological concerns. Undesirable hydrological conditions where metal leaching and acid generation may be increased by cyclical variation in water levels in the wastes as well as convective transport by water flowing through the wastes.

- Uncertainties. Uncertainties in waste stability in the pit and the inability to take corrective measures after placement may represent an unacceptable level of risk.

- Cost. Pit disposal can be relatively expensive, particularly if the pit surround and/or the waste materials need to be modified or engineered to ensure satisfactory performance. For the single pit scenario, pit disposal at end of operations would need to be temporary, fully engineered waste management facilities. In this case, the waste management facility costs could approach double that of a surface facility when transfer and the temporary site remediation costs are included.

- Restrictions on sterilizing residual resources. In several jurisdictions the filling of pits with solid waste may not be permitted if such filling renders inaccessible residual mineral resources under or near the pit.

In-pit disposal has become a well-accepted practice in many jurisdictions and sectors of the mining industry and in some instances (e.g., uranium in Saskatchewan, Canada) disposal of tailings in mined out pits has become a regulatory policy requirement.

1.1 INFORMATION SOURCES

Information sources have included the following:

- MEND Documentation – several reports, conferences and workshops;
- Specific reports provided by the MEND Secretariat;
- Proceedings, International Conference on Marine and Lake Disposal of Mine Tailings and Waste Rock, Egersund, Norway, Sept. 7-10, 2009;
- Literature searches; and
- Personal contacts[5].

[5] Including visit to uranium and engineering companies, as well as Canadian Nuclear Safety Commission Library, Saskatoon, December 2009.

2.0 ASPECTS TO CONSIDER FOR IN-PIT DISPOSAL OF REACTIVE MINE WASTES

2.1 GENERAL ASPECTS TO BE CONSIDERED FOR IN-PIT SELECTION

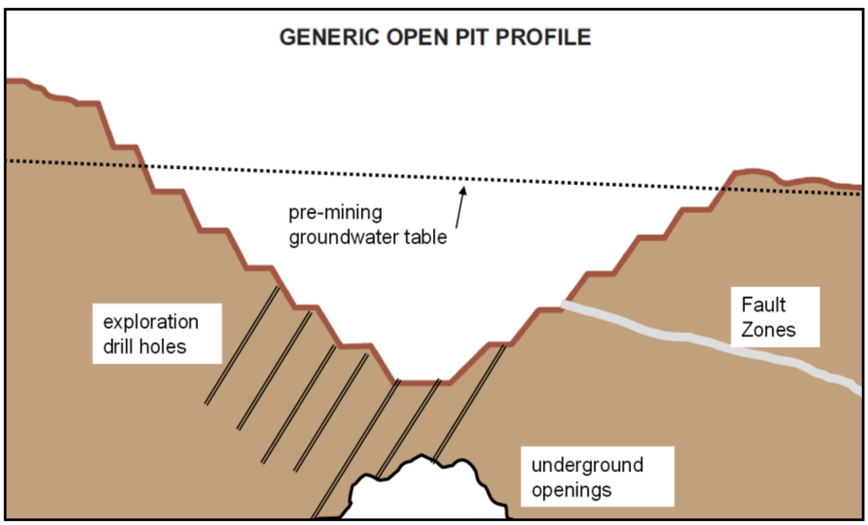
2.1.1 Site Characteristics and Limitations

The critical first consideration for in-pit disposal is to "know the pit". Information not needed for mining may be necessary for pit disposal design and engineering. Some typical aspects are shown in the generic open pit profile illustrated in Figure 1. The most important physical aspects include:

- Pit hydrology and hydrogeology. The location of the pre-mining groundwater table may or may not have been known, but it is critically important to determine where the final water table will be and the rate at which it will be re-established. Complicating factors include mining-induced fracture zones, open drill holes and zones of high hydraulic conductivity such as those created by fault zones.

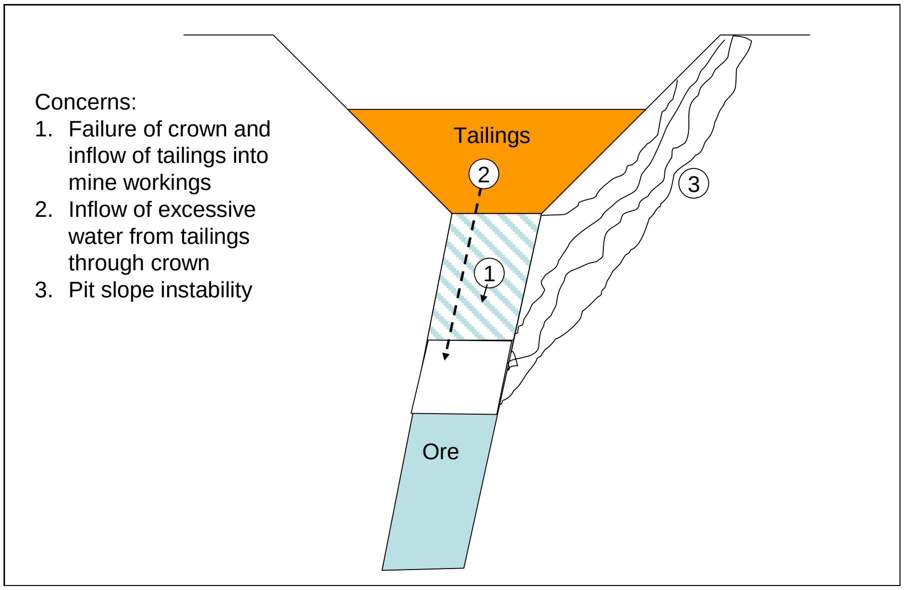
- Pit wall physical stability. Pit walls that are stable during mining can become unstable with time in wet climates, or experience an increased hydraulic pressure associated with elevation in the phreatic zone and as a pit lake develops.

Figure 1 Generic Open Pit Profile



As noted above, the use of a pit for waste disposal cannot be considered until mining is completed in both the pit and below the pit. As shown in Figure 2, the loading of tailings (or waste rock) could result in the failure of a crown pillar which would endanger underground workers as well as result in the sterilization of any residual mineral resources[6]. If mining were completed and in-pit waste disposal established, crown pillar instability could adversely affect the intended integrity of the waste disposal.

Figure 2 Concerns of Pit Disposal above Underground Mine Workings



- Wall Rock Characteristics. The potential for acid generation and metal leaching by broken and exposed wall rock will need to be considered. The determination of hydraulic conductivity of undisturbed and blast-fractured wall rock is also an important consideration for in-pit disposal design.

2.1.2 Waste Characteristics

An essential requirement is the determination of the chemical and physical characteristics of wastes when they are produced and at the time of potential placement in an open pit.

2.1.2.1 Waste Rock

Planning for waste rock disposal into an open pit would normally begin at the start of mining. With the high waste to ore ratios common at most modern mines, all of the waste rock may not fit into the pit. This provides an incentive to segregate mineralized and potentially reactive waste rock from clean waste rock. Reactive waste rock would produce oxidation or leachable products that would need to be managed during the temporary surface storage and after placement in the pit.

[6] Figure 2 courtesy Golder Associates 2009.

2.1.2.2 Tailings

The usual in-pit disposal strategy for tailings is to discharge the tailings directly into the pit as a slurry. In some cases, the tailings can be discharged into the pit without modifying the tailings chemical or physical properties. The tailings settle and consolidate and excess water becomes a water cover. This water may be drawn off to be recycled to the mill or treated and discharged. In other cases, the tailings may be chemically and physically engineered and deposited as a thick slurry that consolidates as a relatively impervious material (relative to the pit surround). State of the art in-pit tailings disposal technology includes paste-consistency disposal under a water cover to prevent particle segregation, maximize consolidation and prevent frost penetration in cold climates. In both the simple and highly engineered cases, the tailings geochemistry and potential metal leaching and acid generation dynamics would need to be considered in the pit disposal design.

2.1.2.3 Treatment Sludges

Precipitates from water treatment can be combined with tailings for pit disposal or discharged into pit lakes that have formed.

2.1.2.4 Contaminated Soils and Materials

Contaminated soils resulting from general mining activities or site mishaps can and have been disposed in mined out pits. Examples of this practice as illustrated below in Case Study #5 – Aznalcollar, Seville, Spain.

2.1.2.5 Regional Hydrology

Regardless of local climate conditions, consideration of in-pit disposal should include consideration of the potential effects on regional ground waters. Typically, this includes extensive site geotechnical investigations combined with modeling predictions.

3.0 GENERIC PIT DISPOSAL CONCEPTS FOR SUCCESSFULLY ISOLATING REACTIVE WASTES

3.1 OBJECTIVES FOR PIT DISPOSAL

The objectives for pit disposal are similar to those for surface facilities:

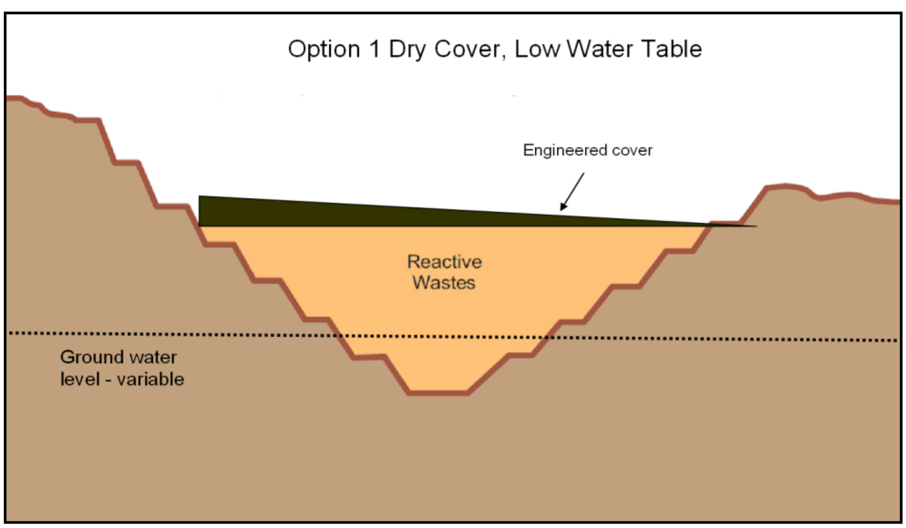
- Acceptably low environmental impact during operations and following closure;
- Minimum care and maintenance;
- Minimum potential for intrusion and disturbance; and
- Acceptable cost.

In-pit disposal offers the potential for minimizing environmental impact upon closure and the need for long term institutional control and monitoring. In some cases, in-pit disposal can reduce the potential for the formation of contaminated pit lakes and permits the land to be returned to previous or traditional uses.

3.2 DRY DISPOSAL WITH ENGINEERED COVERS (OPTION 1)

Some mined out pits are nominally dry or have low water tables, as shown in Figure 3. In addition, those with low water tables may have seasonably variable water tables which could result in the dispersion of oxidation products and leachable metals. An example of such in-pit disposal is shown in Case Study #1 for Whistle, Ontario, Canada.

Figure 3 Dry Cover In-Pit Disposal



3.3 WET DISPOSAL

In many locations in-pit disposal can be accomplished with water saturating the wastes as depicted in Figure 4. For waste rock disposal, since deposition would normally be by haul, dump and distribute methods, the water cover would be established on closure (Figure 5). Also tailings can be deposited under water or covered with water on closure (Figure 5).

Figure 4 Water-Saturated Wastes

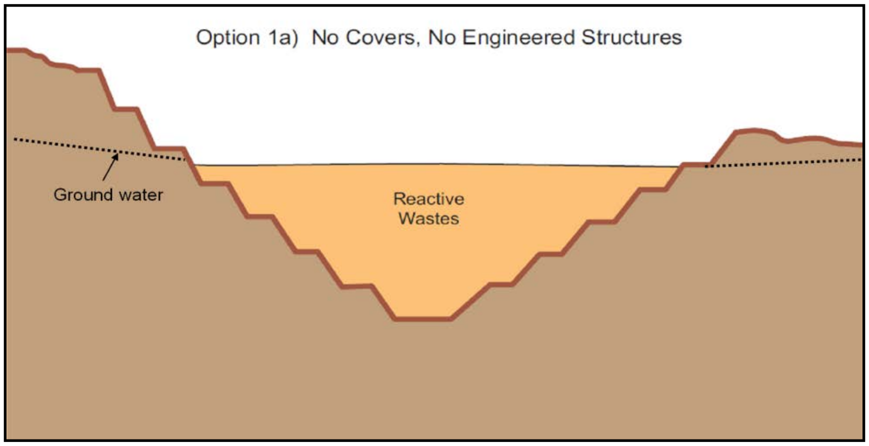
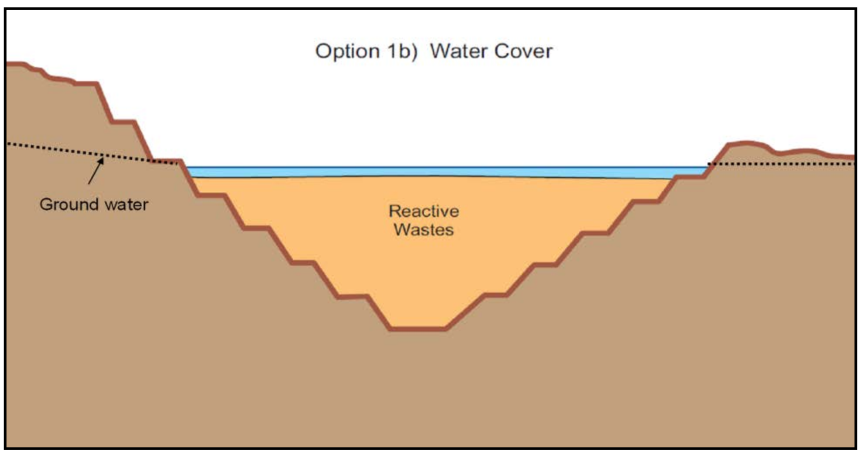


Figure 5 Water-covered In-Pit Disposal



3.4 CHEMICALLY AND PHYSICALLY ENGINEERED WASTES (OPTION 2)

In order to isolate oxidation products and mobile metals and toxic elements, advanced methods have been developed to prepare and dispose tailings in mined out pits as depicted in Figure 6. In this scenario, the engineered waste acts as a relatively impervious mass (to surrounding pit walls and local rock) holding contaminants in place and directing regional groundwater flow around the tailings as illustrated in Figure 7[7]. A detailed discussion of the practice of engineering waste for in-pit disposal is presented in Case Study #10 (McClean Lake - Saskatchewan, Canada).

Figure 6 Engineered Mine Waste Disposal in Mined-out Pit

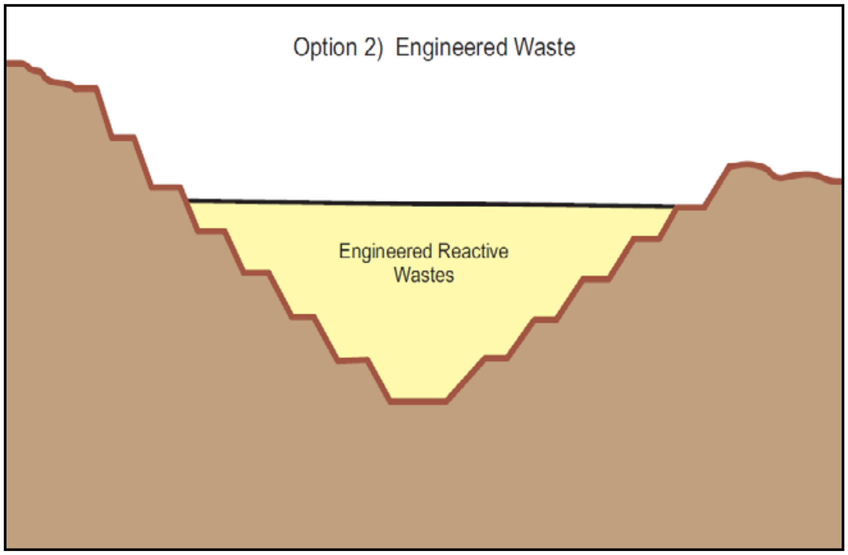
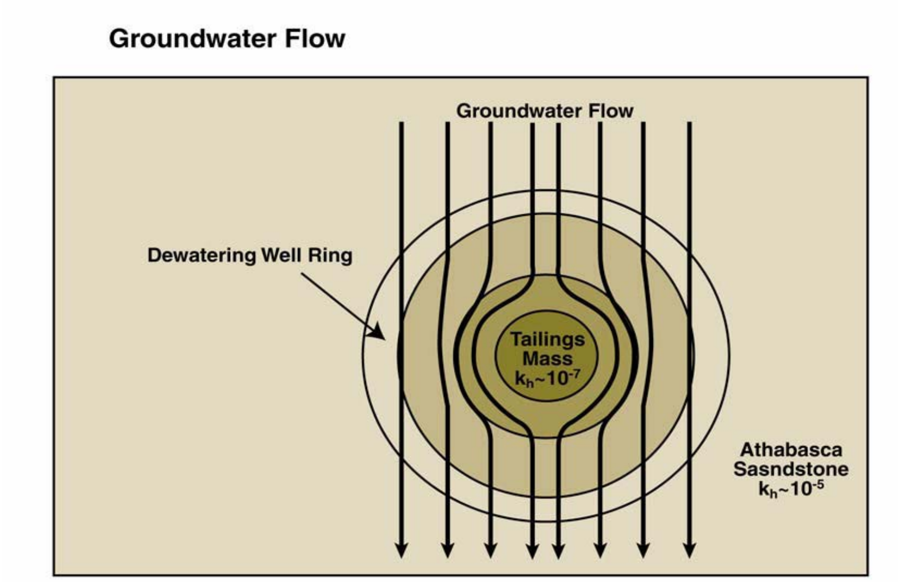


Figure 7 Conceptual Isolation of Engineered Tailings in Open Pit

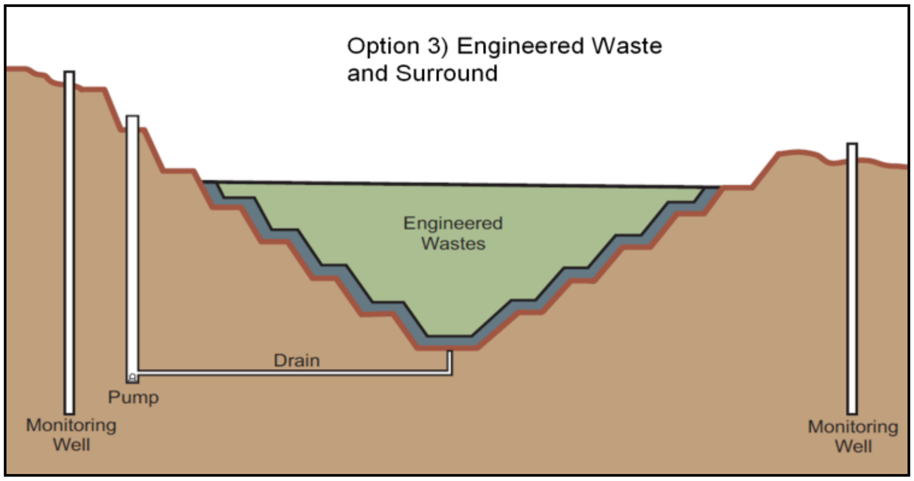


[7] Tailings Optimization and Validation Program, Areva Resources, 2005.

3.5 ENGINEERED PIT CONTAINMENT

A pioneer in engineered pit containment has been the Rabbit Lake operation in Saskatchewan, Canada. This option is shown in Figure 8 below and is discussed in detail in Case Study #9. The Rabbit Lake experience outlines 18 years of valuable pit disposal technology development.

Figure 8 Engineered Waste and Surround



4.0 CASE STUDIES

The following is a summary of the in-pit disposal case studies that are detailed in the attached Appendices.

#	Case	Country	MEND 1995[1]	Principle Metal(s) Mined	Disposed in Pit	Issues -ARD, ML, other	Pit Disposal Type	Pit Status	Information Available to Assess Performance	Comments
1	Whistle	Canada	No	Ni, Cu	WR	ML, ARD	Dry cover, small saturated zone	Closed	Good	
2	Owl Creek	Canada	Yes	Au	WR	ARD, ML	Water cover	Closed	Fair	Site Visit 2010
3	Solbec	Canada	Yes	Cu, Zn	WR	ARD, ML	Water cover	Closed	Good	
4	Equity Silver	Canada	No	Ag, Au, Cu	Sludge, WR	ML, ARD	Water cover	Pit receives sludge	Good	
5	Los Frailes	Spain	No	Zn, Cu	Tailings, WR contaminated soils	ML, ARD	Mostly water cover, some exposed, pit lake	Active care	Good	Used to clean up spilled tailings (1998)
6	Stratmat	Canada	No	Cu, Pb, Zn	WR (planned)	ML, ARD	Dry cover	Closed	Fair	No pit disposal
7	Jundee	Australia	No	Au	Tailings, WR	Cyanide N Asbestos		In closure	Fair	Complex site
8	Marymia	Australia	No	Au	Tailings	Cyanide Asbestos	Spigot from pit edge	Closed	Good	Good geotechnical
9	Rabbit Lake	Canada	Yes	U	Tailings, sludges	ML, Rad[2]	Engineered pervious surround	Operational	Good	Detailed experience
10	McClean Lake	Canada	No	U	Tailings, sludges, WR[3]	As, Rad ML, ARD,	Engineered tailings, porous surround	Operational	Very Good	
11	Lichtenburg	Germany	No	U	Highly Oxidised WR	ARD, ML, Rad	Engineered waste, dry cover	Closed	Good	Site converted to a park
12	Omai	Guyana	No	Au	Tailings	Cyanide	Simple discharge of tailings	Closed, now pit lake	Good	Excellent results

[1] MEND 1995, Review of In-pit Disposal Practices. MEND Report 2.36.1.
[2] Rad – radioactive elements – uranium, radium, thorium etc.
[3] Separate pits are being used for tailings and mineralized waste rock.

5.0 LESSONS LEARNED – SUCCESS OF IN-PIT DISPOSAL

At many metal mine sites when open pits are mined out, mining may proceed to other pits or to underground workings. Subsequently, a vacant open pit provides an opportunity for secure, reactive mine waste disposal.

From the examples reviewed in the following 12 Case Studies, global experience has indicated the following:

1. Metal mine wastes that contain significant sulphide oxidation products may be successfully disposed in a pit provided there is sufficient alkali addition to immobilize the oxidation products and the wastes remain submerged under water (e.g., Owl Creek);
2. The results of disposing of oxidised sulphide waste without sufficient alkali addition may result in a metal-contaminated pit lake, although contamination may successfully be removed by well-planned batch treatment techniques;
3. Pits with an elevated profile may be backfilled with oxidized waste rock with long term treatment as a closure strategy. The treatment requirements may be reduced by the addition of alkalinity to the wastes and the installation of engineered covers (e.g., Whistle);
4. In some cases, backfilling of oxidised waste rock into pits with elevated profiles unfavourable hydraulic profiles, may be rejected in favour of surface waste disposal and long term collection and treatment (e.g., Stratmat);
5. Open pits can provide both emergency operational disposal locations for mine wastes (e.g., Omai and Los Frailes);
6. In very dry climates, in-pit disposal is an acceptable technique for isolating mine wastes that could be dispersed or contaminate the water table if left on surface (e.g., Jundee and Marymia);
7. For high grade uranium tailings, in-pit disposal provides the optimum and highly acceptable management strategy provided that both the tailings and the pit containment are well engineered (e.g., Rabbit Lake and McClean Lake); and
8. At closed mine sites with a range of waste materials, open pits provide the opportunity to select and deposit the most potentially acid generating and metal leaching materials in the lower sections of a pit where oxidation and mobility potential may be the lowest (e.g., Lichtenburg).

Of course, pit disposal is not always the best option. Pits associated with active underground mining cannot safely be used. Frequently, potential mineral resources are present in pit walls. Pit disposal at the end of mining may also be unattractive because of relocation costs and surface environmental legacies.

CASE STUDIES

CASE STUDY #1 WHISTLE MINE

SYNOPSIS

In-pit disposal at the Whistle Mine is an example of relocating oxidized waste rock into a pit cavity that has a profile that extends above the surrounding terrain. Acid generating and metal leaching waste rock was placed into the pit with the acidity neutralized and soluble metals stabilized with lime during placement. Following extensive testing and design, an engineered cover was established to isolate the waste rock in the pit. Long term collection and treatment of small amounts of seepage is provided for.

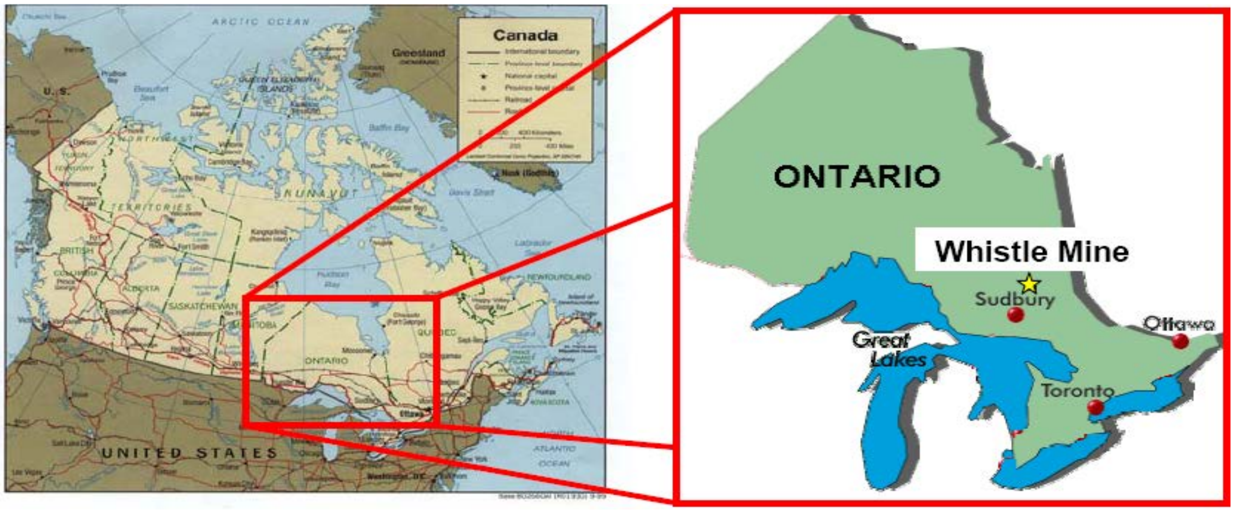
The Whistle Mine was a nickel-copper open pit mine that operated for less than 10 years. It is located on the north-east edge of the Sudbury Basin in Ontario, Canada. Decommissioning of the mine involved the relocation of 6.4 million tonnes of acid generating waste rock into the mined out pit. The untreated waste rock had a high acid generating potential (~44 kg H_2SO_4/t) with significant Ag, Cu, Ni, S and Sr content and leachable Al, Co, Cu, Fe, Ni, Sr and Zn. Lime was added to the waste rock at the rate of 1.8 kg lime/t of waste rock during emplacement. Pit disposal of oxidized waste rock took place in 2004 and 2005. The pit had a partial elevated profile and slightly less than half of the waste rock remains above the water saturated portion of the waste rock. A key feature of in-pit disposal at Whistle is the engineered cover which was designed and placed to minimize oxygen and water infiltration.

Data from 2007 and 2009 indicated that the cover has performed as designed with oxygen concentrations reduced to 1 to 3% of atmospheric in the waste and meteoric water infiltration has been limited to 2.2% of precipitation. For the catchment area of 9.7 ha, up to 2000 m^3 of seepage water is collected and treated annually.

LOCATION

The Whistle Mine is located about 30 km north-northwest of Sudbury, in Ontario (Figure 1-1) in the Post Creek watershed (an area of approximately 5400 ha that drains into Lake Wanapitei, 3 km east of the mine [6] on the Canadian Shield that is characterized by bedrock covered by a thin discontinuous blanket of glacial till [6][4][14]. The mine is located in a semi-humid climate with an annual precipitation of 900 mm (30% of which is snow) and average evaporation of 520 mm. Conditions are generally wet in the fall, winter and spring and dry in the summer. The mine is surrounded by undisturbed forest with wetlands and ponds in low areas [6] [14].

Figure 1-1 Location of Whistle Mine [6]

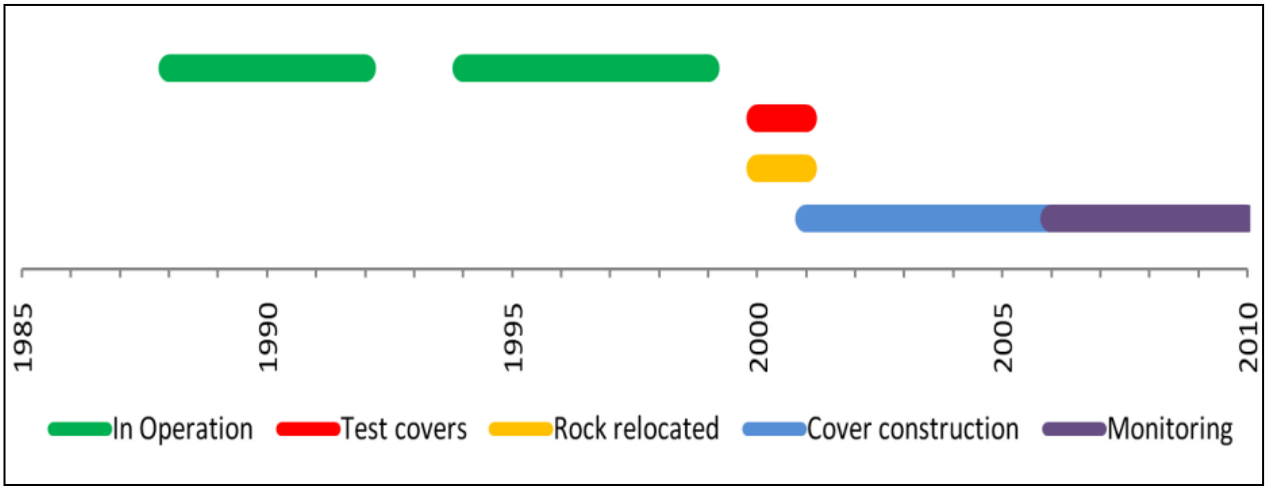


MINE TYPE and HISTORY

Vale's Whistle Mine was a Cu-Ni open pit mine which operated from 1988 to 1991 and from 1994 to 1998 [6]. The original orebody was discovered in 1897 and the original indicator gossan outcrop remained undisturbed at Whistle during mining [6]. The resource was exhausted in 1999 after producing 5 million tonnes of ore from the original 6 million tonne resource grading 0.9% Ni and 0.3% Cu [15]. The ore was shipped to the Clarabelle mill in Sudbury for processing [9]. The mine was decommissioned following the submission of a closure plan that was accepted as filed by the Ministry of Northern Development and Mines, Ontario in 2001.

The waste rock had been deposited close to the pit opening. The waste was relocated to the pit between July 2000 and December 2001. The surrounding bedrock that had been impacted from the storage of waste rock on the surface was cleaned between May and December 2002. A significant amount of earth work in preparation of a soil cover was completed in 2003 and early 2004 along the pit perimeter [4]. Construction of the pit cover system was completed during the snow-free periods of 2004 and 2005 [6]. The project was completed in 2006 for a total cost of $19 million [10], or approximately $3/tonne of waste rock. The timeline of operations at Whistle Mine is presented in Figure 1-2.

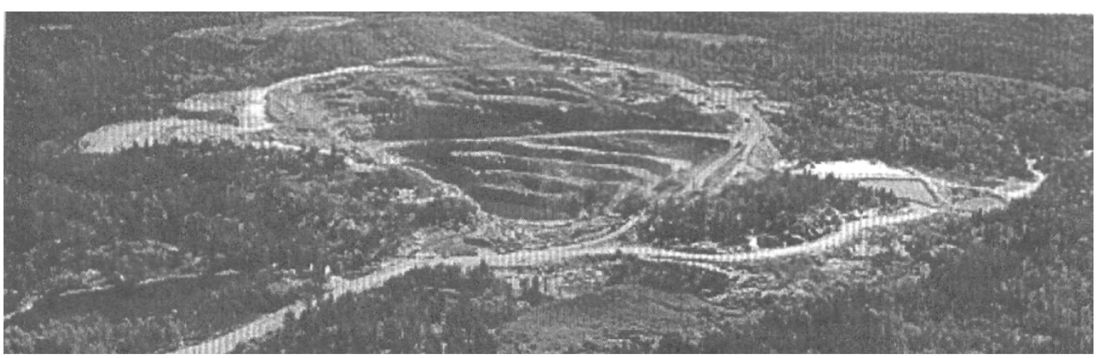
Figure 1-2 Timeline of Events: Production and Remediation Periods of Whistle Mine



PIT and WASTE ROCK DESCRIPTION

The pit cavity had an estimated volume of 3.2 Mm^3 [14]. The pit drainage area covered an area of 9.7 ha and sloped an average of 13% from the north to south perimeters due to the original natural relief in the area [6]. The local area, at the NW edge of the Sudbury basin, consists of granite host rock with mineralized volcanic intrusions and a thin mantle of surficial till [14]. The property is underlain by Archean-aged mafic and felsic gneisses, which are cut by several generations of diabase dykes and extensive belts of prospective Sudbury breccia. The waste rock had been stored in two waste dumps immediately north of the open pit, covering an area of approximately 14 ha [9].

Figure 1-3 NNE Perspective of the Whistle Open Pit [10]



The two piles of waste rock were both adjacent to the pit; the northwest waste rock pile was mainly granite rock, the northeast pile was acid generating mafic rock [14] (Figure 1-6). The porosity of the stockpiles was greater than 20%, and 92% of the material was greater than 100 mm in diameter (Figure 1-6 below). This resulted in an estimated net infiltration of >50% of annual precipitation and an unlimited supply of atmospheric oxygen. Information from MEND (1997) recorded internal temperature measurements, temperature of the collected seepage, and field observations of venting of cold air from the base in summer of the pile and some vapour jets from the top of the pile in winter [11]. Seepage discharging from the Waste Rock Dumps (WRD's) was characterized by high concentrations of sulphate (1760 to 5200 mg/L) and nickel (118 to 297 mg/L) as well as elevated concentrations of iron, copper, cobalt and zinc [6].

In studies by Tran *et al.* (2003), the originally deposited waste rock was determined to have undergone some weathering, but still contained a considerable amount of unoxidized sulfide mineral. The flushing of oxidation products had been limited, indicating significant storage of oxidation products. There was also indication that materials that had a D_{50} particle size in the range of 5 mm to 30 mm were more readily flushed then samples that had a D_{50} particle size of <5 mm or >30 mm. This suggested that preferential water pathways had developed within the dump site [14].

Table 1-1 Summary of the Results of the Testing done by Tran *et al.* (2003) on 38 Samples from the Whistle Mine Site

Parameter	Whistle Mine Results
Existing pH	8% (3 samples) had paste pH <4.5
Average total S	2%
Average acid neutralizing capacity	20 kgH_2SO_4/t
Average net acid producing potential	44 kgH_2SO_4/t
Potentially acid forming (% of total samples)	79%
Non-Acid Forming (% of total samples)	16%
Uncertain (% of total samples)	5%
Elements enriched in solids	Ag, Cu, Ni, S and Sc
Elements enriched in water extracts	Al, Co, Cu, Fe, Ni, Sr and Zn

Figure 1-4 Whistle Mine Pit in 1997 [11]

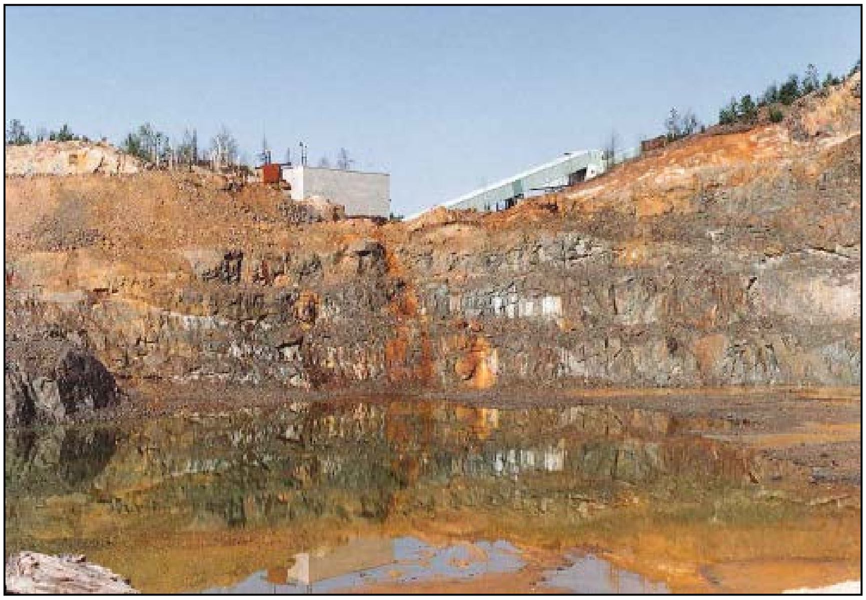


Figure 1-5 Whistle Mine Pit Waters, Green Indicating Nickel Content[12]

[11] G. Feasby Photo, 1997.
[12] G. Feasby Photo, 1997.

Figure 1-6 Waste Rock at Whistle Mine[13]

STRATEGIC USE OF PIT

Consideration was given to closing out the waste rock dumps (WRD's) in place, but there were several concerns. First, due to the proximity of the WRDs to the open pit and a nearby creek, re-grading some of the WRD slopes to facilitate construction of an earthen cover system was not feasible. Second, while a properly designed and constructed cover system would substantially reduce the infiltration of meteoric water at the cover surface, it would still allow for preferential ingress of oxygen through the underlying glacial till at the toe of the slope. Finally, seepage collection and treatment from the toe of the covered WRD's would be required for an indefinite period of time, large amounts of treatment sludges would be produced, and spills could significantly impact the surrounding wetlands [6].

Due to unrestricted acid generation and metal leaching from both the waste rock pile and the open pit walls, with the potential impact on local surface water receptors in addition to the need for indefinite collection and treatment of contaminated waters, the status quo was considered uneconomic and environmentally undesirable. It was considered that all the problematic waste rock could be consolidated back into the pit and covered with a water and oxygen restricting barrier. Fortunately, the waste to ore ratio was unusually low at Whistle and all of the waste could be accommodated into the pit volume [6] [15].

[13] G. Feasby Photo, 1997.

A "water" cover is recognized as the preferred technology for inhibiting the oxidation of sulphide minerals; the water acts as a barrier to the diffusion of atmospheric oxygen to the submerged sulphides (MEND, 2001). Although the climate at Whistle Mine has a moisture surplus on an annual basis ("the bathtub overflows"), a water cover was not reasonably possible at this site because of the absence of a large enough pit lake and extreme cost of building a containment and the potential instability of such a "high rise pit" [6][4].

Therefore, a "dry" cover system was selected to minimize ARD and metal leaching at the Whistle Mine pit which would be backfilled with neutralized waste rock [4]. The design objectives for the backfilled pit cover system included the reduction of atmospheric oxygen ingress to the underlying waste material to a minimal level as determined by geochemical modeling, reduce the entry of meteoric water to the underlying waste material (i.e., through percolation) to less than 5% of the annual precipitation at the site, and to provide a medium for establishing a sustainable vegetation cover that would be consistent with the current and final land use of the area [6][4].

All waste rock stockpiled on the surface as well as all acid-generating and metal leaching waste rock that had been used for the various site roads [6] was to be relocated to the open pit. Lime would be added to the waste rock during the backfilling operation to minimize future generation of acid rock drainage [14]. The cover design would include provision for long term physical stability and erosion resistance while maintaining low oxygen and water permeability. Gas, moisture content and water quality monitoring systems would be installed and the pre backfilling water collection and treatment system would be modified for use as required.

GEOCHEMICAL MODELLING

Geochemical modeling [16] was conducted to assess the effectiveness of various cover options on the long-term water quality of the backfilled pit that identified the following design objectives for the cover:

1. Reduce the ingress of atmospheric oxygen to the underlying waste material to the minimum acceptable level as determined by geochemical modeling;
2. Reduce the entry of meteoric water to the underlying waste material (i.e., net percolation) to less than 5% of the annual precipitation at the site; and,
3. Provide a medium for establishing a sustainable vegetation cover that is consistent with the current and final land use of the area.

The geochemical modeling also showed that the addition of lime to the backfilled waste rock material is essential for reducing the short-term acidity and metals loading from the pit overflow.

Without a cover, the neutralizing capacity of the lime would be quickly exhausted allowing the pile to become acidic within a short period of time, resulting in dramatic increases in sulphide oxidation, acidity and metals concentrations. In combination with a cover, the lime allows the pile to remain neutralized over the long-term, thus reducing future contaminant loads significantly.

Based on these objectives, a multi-layer cover system incorporating a fine-textured soil (compacted clay) layer ("barrier layer") was selected as the preferred type of cover system for the pit. The incorporation of a fine-textured soil layer achieves the first two design objectives by providing a capillary barrier, which a) maintains near saturation within the fine-textured soil layer under all anticipated climatic conditions thus limiting the ingress of oxygen due to low oxygen diffusion conditions, and b) in combination with the lower hydraulic conductivity of the fine-textured soil layer (usually compacted) provides a control on net percolation to underlying waste material.

MATERIALS PLACED IN PIT, METHODS OF PLACEMENT AND COVER INSTALLATION

Waste rock relocation was completed between July 2000 and December 2001. This required the relocation of 6.4 million tonnes of waste rock composed of approximately 80% mafic norite, which had an average sulphide content of 3% [5][6]. Lime (CaO) was added to the waste rock during backfilling at an average rate of 1.8 kg/t to minimize the future generation of acid rock drainage [14]. A total of 11,462 tonnes of lime was added to neutralize acidity.

Surrounding bedrock affected by the storage of waste rock on the surface was cleaned between May and December 2002. Rock outcrops and mineralized remnants of the pit wall along the western and northern perimeter of the pit were blasted to facilitate extension of the pit cover system to a minimum of 6.0 m beyond the pit perimeter. A 4.5 m wide band around the entire pit perimeter was cleaned with power washers to ensure intimate contact between the bedrock and low permeability clay section of the cover. The objective of this preparatory work was to minimize the entry of atmospheric oxygen and meteoric water to the pit rock backfill along the pit perimeter [4]. Provision was provided for the collection and treatment of small amounts of contaminated water seeping from the former waste rock storage areas and backfilled pit [5].

The waste rock relocation into the pit involved placement of the rock in 2.6 metre lifts; material was dumped, spread, and compacted in shallow lifts in order to avoid excessive settlement in the fill material once the cover had been placed. The final lift of waste rock was reduced to 1.3 meters and was composed of smaller sized particles to facilitate better compaction and a smoother surface for the placement of a soil cover [9]. It was considered that dumping the rock from the top of the pit wall would have allowed too much settlement in the pit, and this movement would have compromised the integrity of the engineered cover. The waste rock was trucked into the pit over a three-month period [10]. A compacted silt/trace clay material was chosen as the preferred barrier. Clay was excavated from the Copper Cliff property (another Vale project). This material had a nominal clay content (material <2 µm) of 24%; an XRD analysis determined the clay is comprised mainly of quartz, feldspar, chlorite, and kaolinite, with minor amounts of vermiculite and illite [4]. The clay had low to medium plasticity and a saturated hydraulic conductivity of 5×10^{-8} cm/s when compacted to 95% of standard Proctor maximum dry density [5].

Figure 1-7 Whistle Photos. (Clockwise from top left: cleaning of the pit perimeter prior to barrier layer construction, cross-slope ripping of topsoil into the granular cover material, and the finished pit cover in July-08 [5] with the collection Pond overshot gate in the foreground) [6]

The final cover system design for the backfilled pit consisted of a 0.1 m sand and gravel leveling course, a geosynthetic separation fabric (geotextile), a 0.45 m barrier layer comprised of compacted clay, and a minimum of 1.2 m of sand and gravel for a protective/growth medium layer with 0.08 m of topsoil admixed to the cover surface to increase nutrient levels for revegetation efforts [2]. The primary purpose of the leveling course was to provide a suitable foundation for the geotextile, but it also would act as a capillary break layer. A thin layer of topsoil was admixed to the pit cover surface to assist with growth of a seeded mixture of native grass and legume species [6]. The surface of the backfilled pit, which covers approximately 10 ha, has an average slope of 17% over a maximum length of 125 m. This conformed to the natural relief in the area [2]. It was anticipated that about 60% of the relocated waste rock would become fully saturated as the pit flooded [14].

In conjunction with the design of a sustainable final landform, provisions were required to minimize fluvial erosion on the pit cover and contain suspended sediment in runoff waters over the short and long term. Progressively higher levels of erosion protection were used in the hill slope channels as the contributing area and associated design flow velocities increased towards the south. This included the use of temporary erosion control blankets, a 150 mm thick layer of 60 mm diameter riprap, and finally, a 300 mm thick layer of 125 mm riprap. All riprap was underlain with geotextile to act as a filter medium [6].

The final landform consisted of several catchments oriented parallel to the slope with progressively higher levels of erosion protection in the channels down slope (Figure 1-8 and Figure 1-9). The resulting landform was more analogous to natural systems and provided a micro-topography to aid re-vegetation. The overall design included a series of managed ponds at the base of the slope. Over time a wetland area would be established to provide long-term attenuation of peak surface flows and diversified habitat for wildlife [15].

Figure 1-8 Final Design of the Landform for the Covered Backfilled Pit [15]

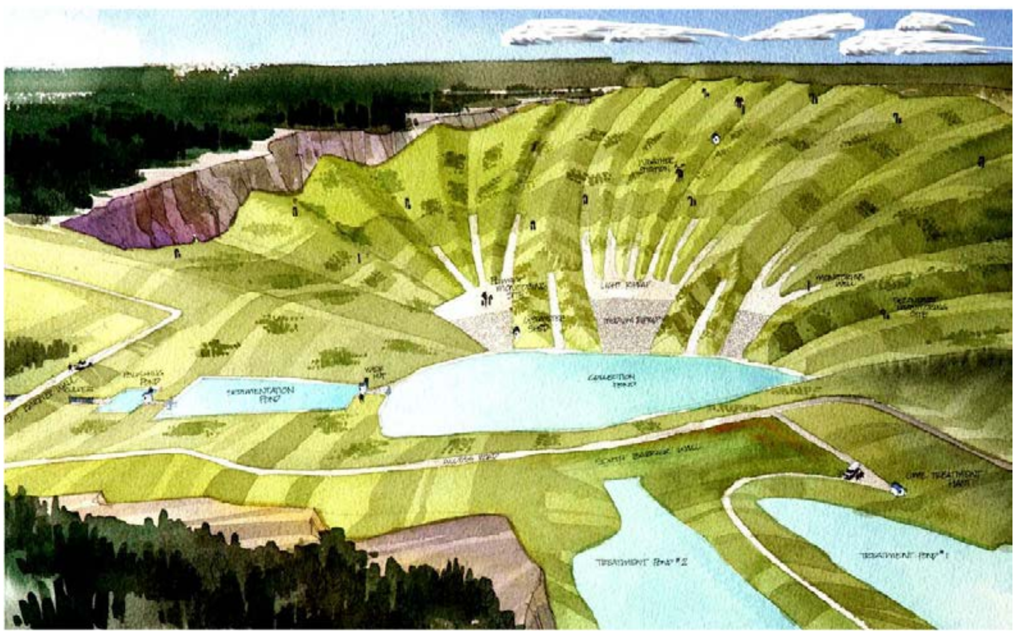


Figure 1-9 Arial Photo for Final Covered Backfilled Pit at Whistle Mine [8]

A key aspect of construction for this project was cross-slope mixing of 75 mm of topsoil into the surface of the pit-run material. This was designed to minimize erosion of the topsoil prior to development of a sustainable vegetation cover and maximize the root depth for plant growth. In addition, cross-slope ripping of the growth medium surface would ensure the catchment slopes function as planned, at least in the short term, such that runoff water flows along the rip lines to the erosion-protected channels [4]. The final, covered, filled in-pit has convex summits and is concave at the base with drainage systems that follow natural drop lines in the slopes [3].

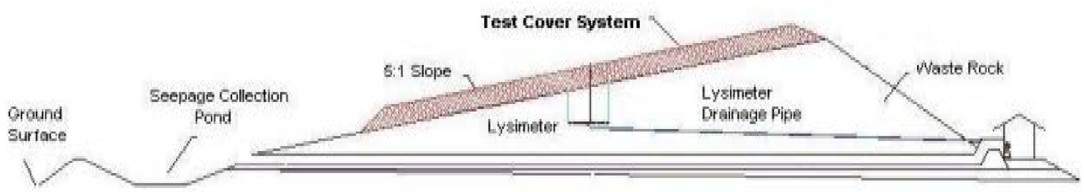
An important aspect of the backfilling and covering was the presence of an onsite testing laboratory routinely confirming that borrow materials conformed to specifications. Testing of completed work areas, particularly the barrier layer, consisted of in situ density, water content and permeability measurements, as well as surveying to check that the specified minimum layer thickness was achieved [6]. A perspective of the site immediately following the pit backfilling, cleaning of waste rock storage areas and pit covering is shown in Figure 1-10.

Figure 1-10 Aerial view of Whistle Mine after Remediation (Google Maps)

Three covers had been evaluated using on-site pilot tests by Ayres *et al*. from November 2000 to 2002; a 0.45 m compacted sand-bentonite mixture (sodium bentonite content 8% on a dry mass basis), 0.6 m compacted silt/trace clay and prefabricated geosynthetic clay liner barrier (GCL; Bentomat® ST a sodium bentonite (<0.1 m thick) between woven and non-woven geotextiles needle-punched together) layers. Each test cover was ~12 m wide x 24 m long and covered by a 0.9 m protective layer of non-compacted soil (from a nearby borrow pit) to protect the barrier layer which was hydroseeded. A waste rock platform with a 20% slope (the natural relief adjacent to the pit) was constructed to support the test cover systems, as well as a seepage collection system (lysimeters) to prevent contamination of the local groundwater system (Figure 1-11) [7]. The platform was ~85 m L x 40 m D x 6 m H containing ~12,000 m^3 of waste rock. These were compared to a test plot with no cover. The tests were set up in September and October of 2000 and conducted concurrently as the pit was being backfilled. The results from these tests lead to the final design of the cover used in 2004 and 2005.

The tests were monitored over a two-year period for net percolation through the cover, climatic parameters, gaseous oxygen and carbon dioxide concentrations, moisture and temperature within the cover and waste materials. Landform modelling was also used to decide on the actual contours of the land to ensure proper drainage, adequate grass cover and to reduce erosion. The objective was to develop a landform with catchments that are similar to natural systems. The cover and waste materials were monitored continuously for climatic parameters; gaseous oxygen / carbon dioxide concentrations (three sampling ports were installed; 0.05 m above the barrier layer, 0.05 m below the barrier layer, 0.6 m below the barrier layer), moisture / temperature and net percolation [10].

Figure 1-11 Cross-section of the Major Components of the Whistle Mine Test Cover Project (after OKC 2001) [6]



A series of three containment ponds were engineered and located at the south side of the backfilled pit for management of suspended sediments in the pit cover runoff water. The base of each pond consists of a minimum 0.6 m layer of compacted clay and slopes gradually east to west, towards individual hydraulic control structures (overshot gates) and the final discharge point (Post Creek wetlands). The Collection Pond is the primary catchment designed to accommodate a 1-in-50 year, 24-hour storm event. The Sediment Pond was designed to optimize the settlement of suspended particles. Finally, the Polishing Pond would serve as a final settling area, as well as a sampling location prior to water being released to the environment. It was anticipated that the runoff ponds would be decommissioned in 5 to 10 years, once mature grass cover is established and erosion from the cover decreased to acceptable levels. Over time, a wetland area would self-establish to provide long-term attenuation of peak surface flows and diversified habitat for wildlife [6].

PERFORMANCE OF IN-PIT DISPOSAL AT WHISTLE

A multi-disciplinary approach was used in the design and implementation of closure works at Whistle Mine. Collection and treatment of seepage from the backfilled pit overflow is expected to continue for tens of years. However, provided the current performance of the pit cover system is sustainable and based on geochemical modeling predictions, it is anticipated that Vale will be able to “walk away” from the site within several decades, ending active management once the cover system is proven to be self-sustaining [10][6].

A performance monitoring system was installed for the Whistle Mine pit cover to achieve the following objectives: 1) measure a water balance for the site; 2) develop confidence with all stakeholders with respect to cover system performance from a micro- and macro-scale perspective; 3) enhance understanding of the key characteristics and processes that control cover system performance at this site; and 4) track the evolution of the cover system in response to various site-specific physical, chemical and biological processes. The system includes a meteorological station, two weirs for measuring runoff flows, two automated stations for monitoring net percolation rates and in situ moisture and gas concentrations within and below the cover system, 13 secondary stations to monitor spatial performance, and four groundwater monitoring wells. Figure 1-12 shows the location of the monitoring stations on the pit cover. Collection of data from all the monitoring sites at Whistle Mine commenced in the fall of 2005 and presently continues [15].

Figure 1-12 Location of Monitoring Stations on the Pit Cover System [4]

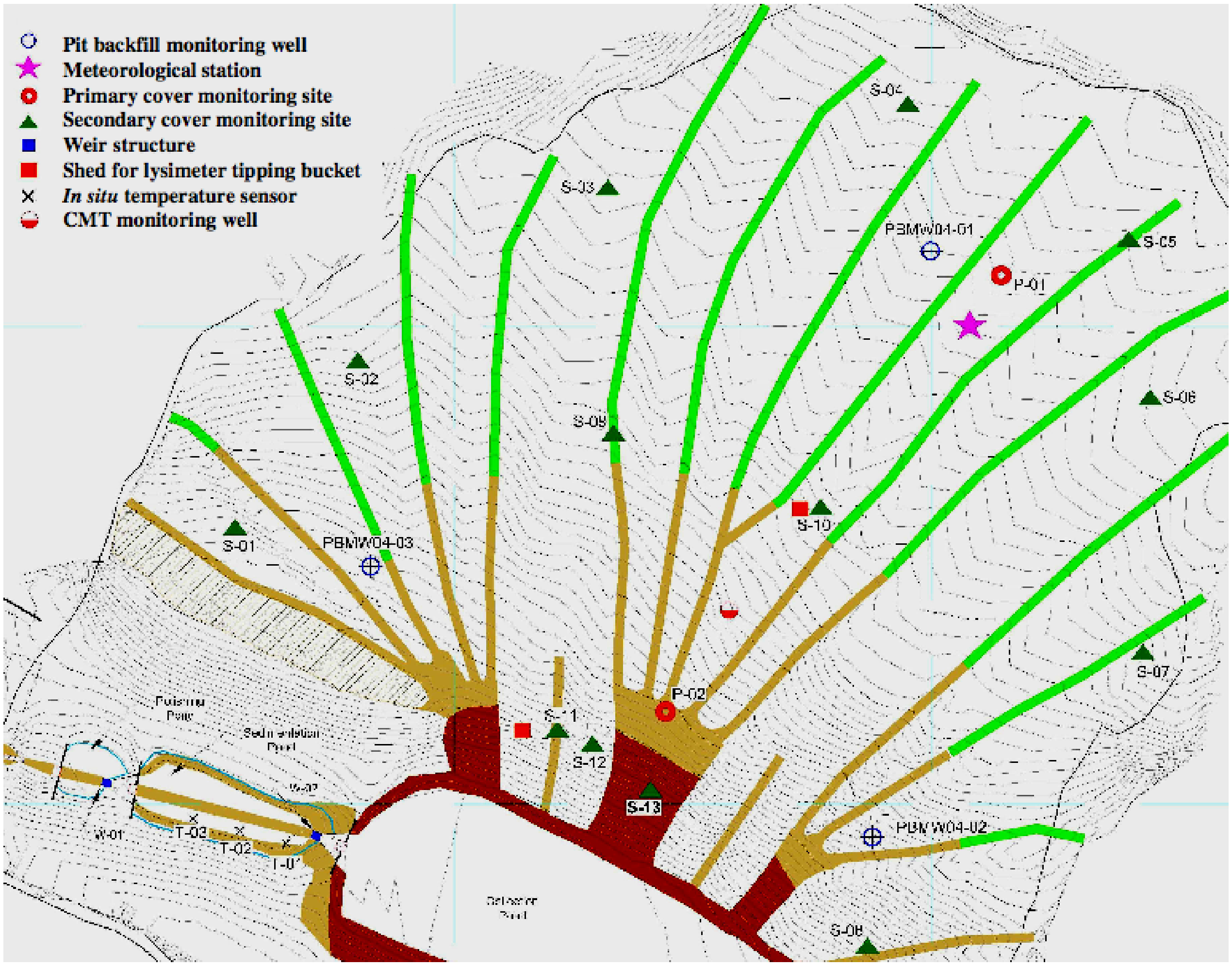
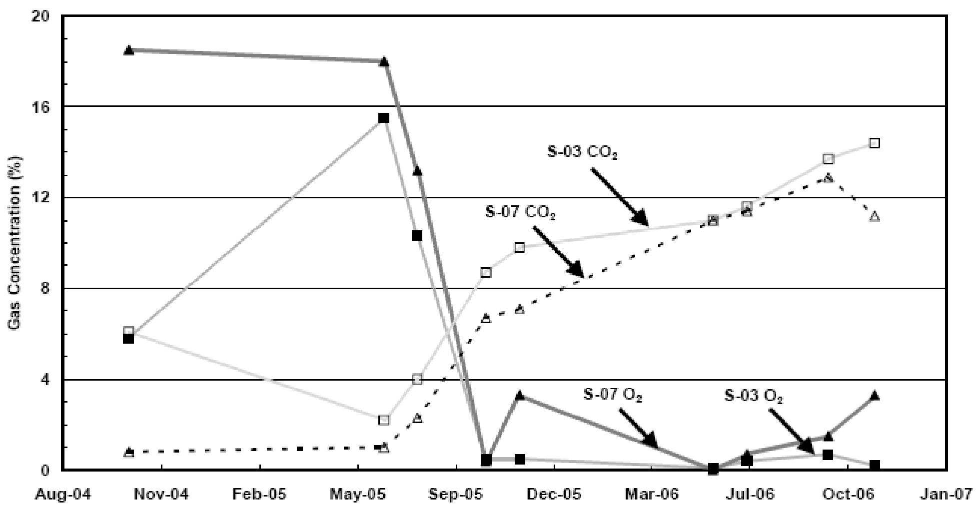


Table 1-2 Performance Monitoring System Program for the Backfilled Pit Cover [4]

Component	Parameters Measured	Details/Comments
Pit backfill monitoring wells	Water quality and hydraulic head	3 wells installed in a triangular pattern Pressure transducer installed in 2 wells for automated head measurements
Meteorological station	Rainfall, snowfall, air temperature, relative humidity, wind speed & direction net radiation	All sensors connected to an automated data acquisition system Snow surveys to be performed in the spring (now water equivalent)
Weirs	Surface runoff flow	2 weirs (one after the Collection Pond and one after the Sedimentation Pond) Automated stage recording, and weirs enclosed in heated huts
Primary *in situ* cover monitoring sites	Net percolation, volumetric water content, matric suction, *in situ* temperature and O_2/CO_2 as concentrations	1 site upslope and 1 site down slope Automated measurements for all parameters except O_2/CO_2 gas concentrations
Secondary *in situ* cover monitoring sites	Volumetric water content, in situ O_2/CO_2 gas concentrations	13 sites across the entire cover surface Portable soil moisture probe and portable O_2/CO_2 gas analyzer

The influx of atmospheric oxygen and meteoric water into the waste rock backfill was substantially reduced following construction of the cover system [15]. Figure 1-13 shows the concentrations of O_2 and CO_2 measured 90 cm below the barrier layer between August 2004 and January 2007. These results indicate that the gas concentrations were reduced and remained low, indicating that the system was stable [6].

Figure 1-13 Concentrations of O_2 and CO_2 Measured 90 cm Below the Barrier Layer [6]

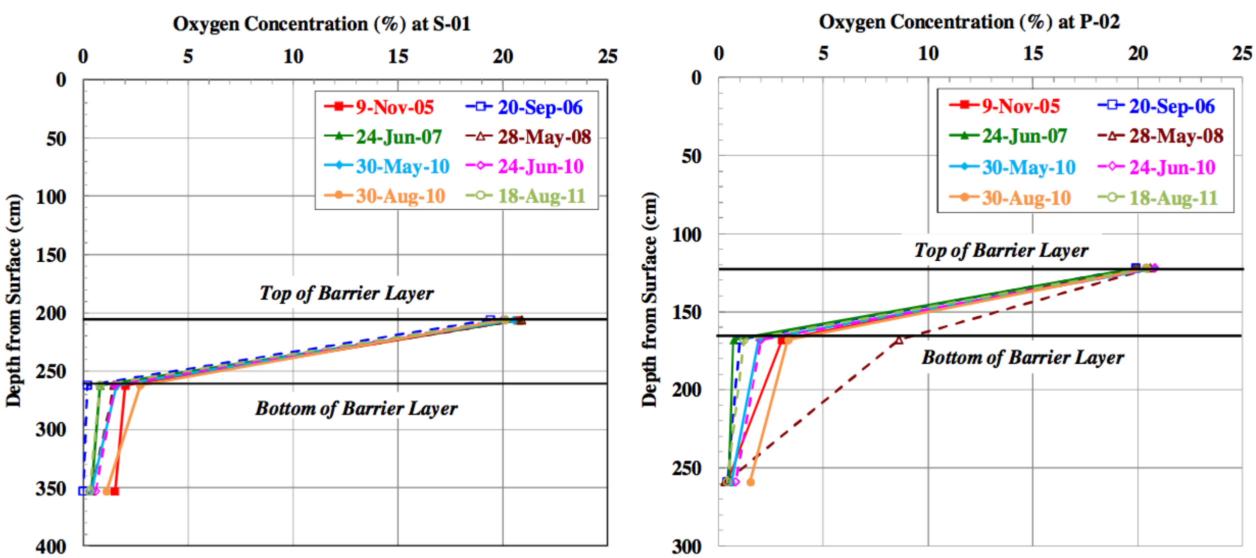


In 2006, the cumulative net percolation measured at station P-01 was 2.7% of the total precipitation measured at the site, which compares well with the annual net percolation of 2.2% of precipitation that was predicted by models for a normal climate year. Net percolation rates were expected to decrease as vegetation develops [6].

The degree of saturation of the clay layer of the pit cover needs to be above 85% in order to create an effective oxygen ingress barrier. Volumetric water content values measured at the P-01 upslope, crest location and the P-02 down slope, drainage channel location determined the degree of saturation of the clay layer which has remained above a 95% degree of saturation since 2006, successfully serving as an oxygen barrier [5]. Based on geochemical modeling the desired oxygen diffusion coefficient (De) for the barrier layer was 3.8 x 10^{-9} m^2/s or lower. The mean De values calculated from measurements taken at stations P-01 and P-02 have been an order of magnitude or two lower than the desired De between the years 2006 and 2010 [5].

Pore-gas concentrations of oxygen and carbon dioxide were measured at the primary and secondary monitoring sites on top and at the base of the barrier layer, and 90 cm below the barrier layer [5]. Oxygen pore-gas concentrations below the barrier layer are less than 2-3%, as seen in Figure 1-14 indicating the oxygen is not transported into the waste rock.

Figure 1-14 Oxygen Pore-gas Concentrations Measured across the Barrier Layer and Within the Underlying Backfill Waste Rock at Stations S-01 and P-02 [5]



In situ temperature and water content throughout the pit cover and upper waste rock profile have been recorded at stations P-01 and P-02. The freezing front in the growth medium extended to a maximum depth of 40 cm at both monitoring stations. Consequently, the freezing front did not penetrate to the barrier layer. Together the growth medium layer above the barrier and the annual snow cover at the site have been sufficient to prevent the pit cover barrier layer from freezing. Volumetric water content in the pit cover and upper waste rock profile at stations P-01 and P-02 is continuously monitored using calibrated Sentek EnviroSCA capacitance sensors. Little variation in water content of the covered profile was measured over the summer of 2006. The majority of wetting and drying in the cover profile was observed to occur in the upper 50 cm where the atmospheric and plant demands for soil water are greatest. The drying of the growth medium layer above the barrier layer during the summer is primarily attributed to lateral drainage or interflow as a result of the slope of the cover system and the low permeability of the barrier layer. The variation between stations P-01 and P-02 was attributed to moisture storage capacity and cover thickness, surface treatment of vegetation or rip rap, and the position along the slope profile [5]. There was no evidence of wet/dry cycling in the barrier based on data collected to-date at stations P-01 and P-02 [6]. Figure 1-15(a) shows the volumetric water content measured at stations P-01 and P-02 in 2006. Between June 1 and September 1, 2006 there was minimal change in the volumetric water content and a comparison to measures taken in 2010 (Figure 1-15(b)) show similar levels of water content indicated that the cover is stable [6].

Figure 1-15(a) Volumetric Water Content Measure at P-01 and P-02 in 2006 [6]

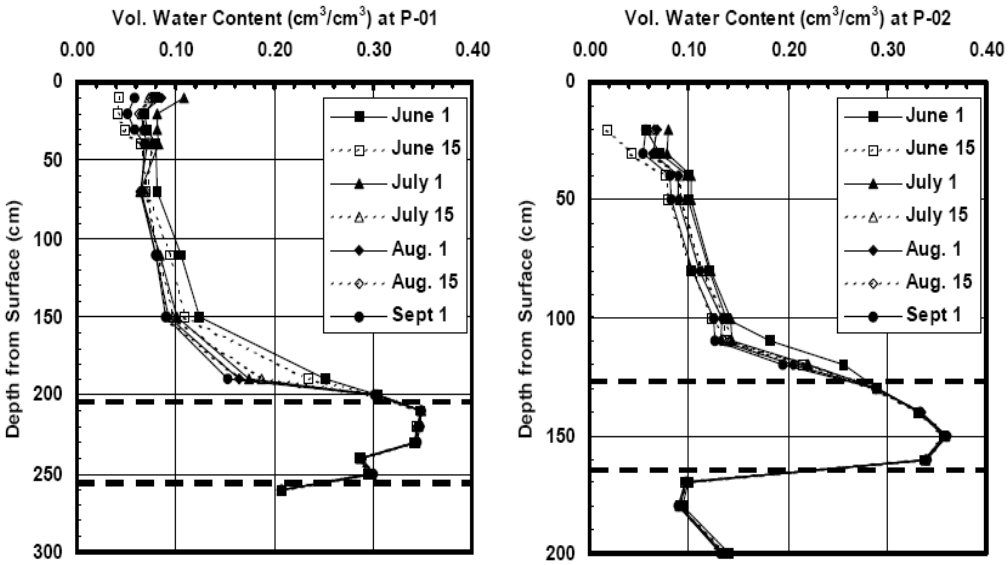
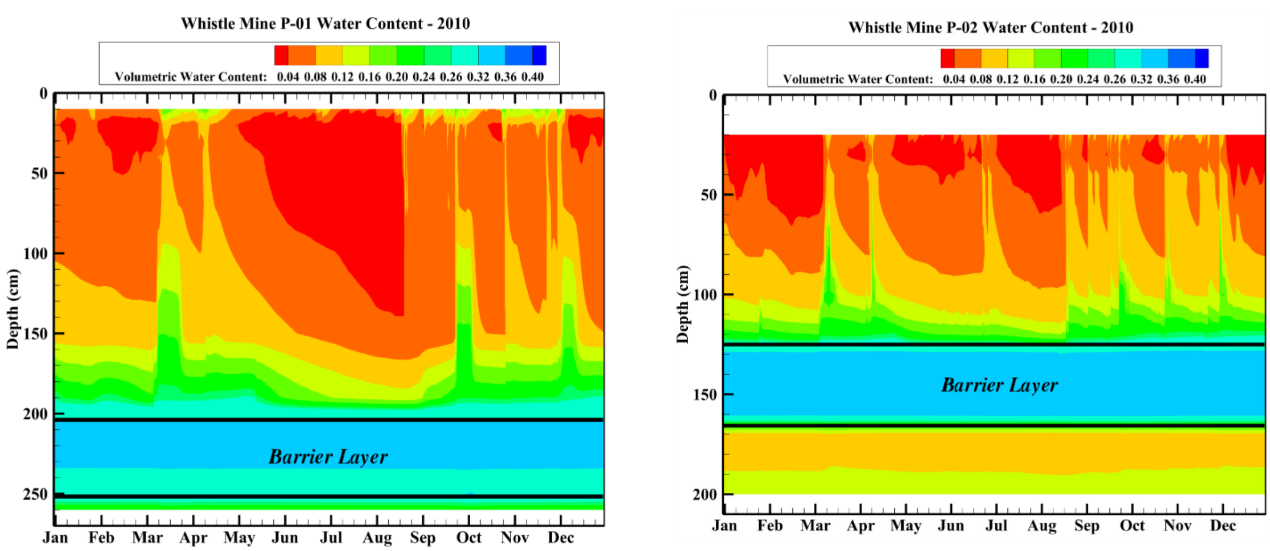
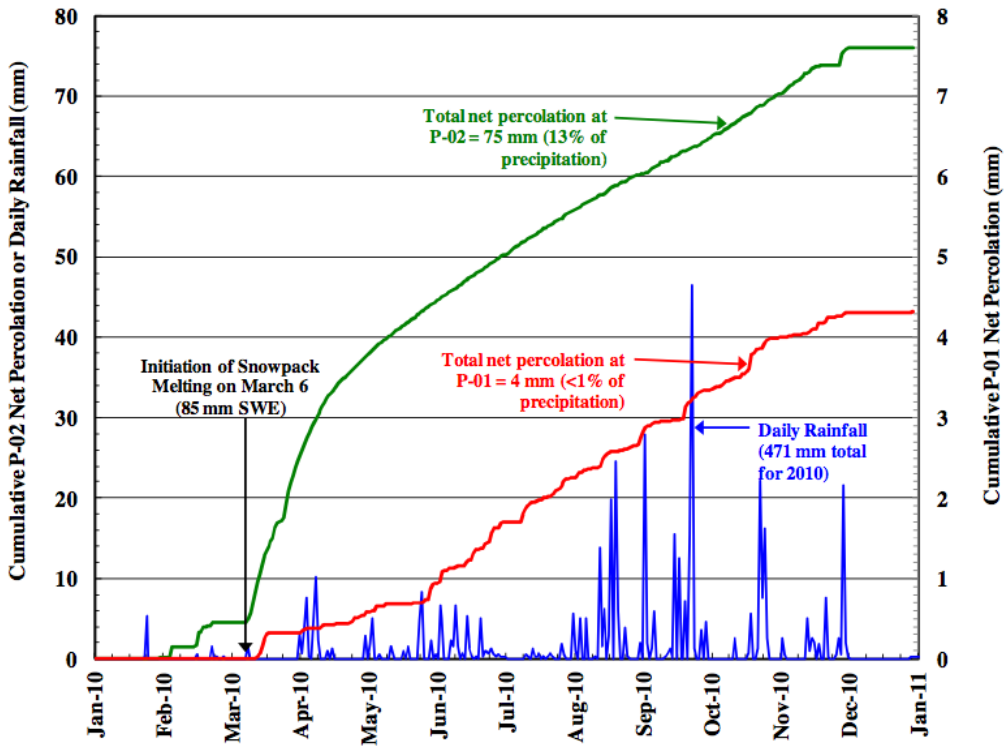


Figure 1-15(b) Volumetric Water Content Contours Measured in 2010 at P-01 and P-02 [5]



Net percolation is the net infiltration of meteoric water across the cover system and into the backfill waste rock [5]. Net percolation measured in terms of percent of precipitation has decreased from 3% during 2006 and 2007 to 1% from between 2008 and 2010 [5]. This decrease is believed to relate to the establishment of vegetation on the cover, which would take up the water in the upper layers of the cover [5]. Cumulative net percolation is measured from station P-01 and P-02 station tank lysimeters and 2010 data is shown in Figure 1-16.

Figure 1-16 Cumulative Net Percolation Measured at Station P-01 and P-02 for 2010. Daily rainfall and initiation of the snowpack melt is also included [5]



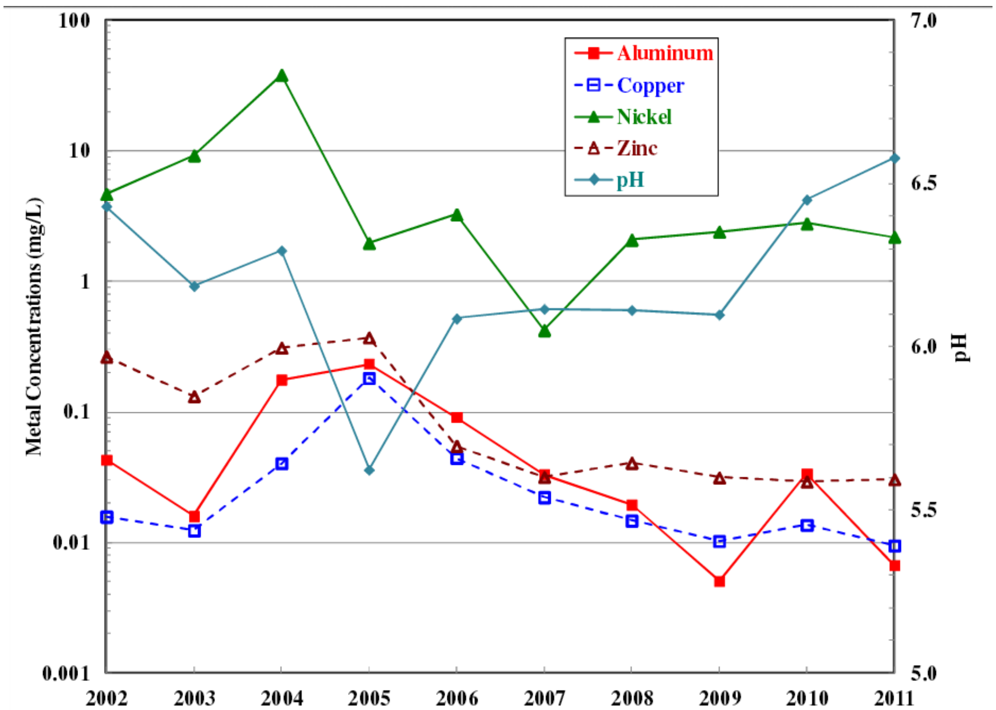
A simple water balance was completed for the backfilled pit and cover system in 2006 for five years between 2006 and 2010 (Table 1-3), using the performance monitoring data collected at the site. A total actual evapotranspiration (AET) of 269 mm (35% of total precipitation) was calculated for the pit cover in 2006. The amount of potential evaporation (PE) in 2006, which is a theoretical maximum assuming free water on the surface at all times, was 50 mm based on the Penman (1948) method and climate data collected at the site. This equates to an AET/PE ration of 0.41. The calculated AET total for 2010 was 377 mm, the amount of PE calculated was 655 mm, and the estimated AET/PE ratio was 0.58 [5]. The AET/PE ratios are reasonable given the upper soil profile on the pit cover and the current state of the vegetation community. The AET/PE ratio was anticipated to increase over time as the vegetation cover matured and transpiration rates increased, and this has been the case between 2006 and 2010 [2][5].

Table 1-3 Annual Water Balance for the Pit Cover System from 2006 to 2010. (2008 values were not obtained due to equipment malfunction [5])

	Water Balance – Precipitation			
	2006	**2007**	**2009**	**2010**
Precipitation (mm)	*765*	*584*	*809*	*556*
Runoff and interflow -%	62	39	40	34
Evapotranspiration -%	35	57	54	68
Change in storage -%	0	1	5	-2
Net percolation -%	3	3	1	1

A high water level drain was installed along the south side of the backfilled pit to prevent any uplift pressures from developing on the underside of the pit cover system, and the drain is the single discharge point from the pit [5]. Figure 1-17 shows metal concentrations and pH of the pit discharge waters starting in 2002. The backfilling of the pit had occurred in 2000 and 2001, and the construction of the pit cover system was completed between mid-2004 and late 2005 [5]. Upon completion of the pit cover, the metal concentrations have decreased and the pH is increasing and approaching neutral. Nickel concentrations remained slightly above discharge water quality objectives requiring continuity of water treatment.

Figure 1-17 Geometric Annual Mean Quality of Pit Overflow Waters [5]



REMEDIAL

The final landform consisted of a number of catchments oriented parallel to the slope in a pattern of crests and troughs. To achieve this topography, an additional 0.6 m of sand and gravel was placed in the crests. This has been beneficial for revegetation because snow accumulates in the troughs, thereby increasing soil moisture levels, and wind velocities are reduced across the ground surface, thus reducing potential erosion of topsoil and grass seeds. The objective was to develop a landform with catchments that are analogous to natural systems (i.e., avoid “fighting” nature) [6].

L. Lanteigne (Vale) reported that there is no erosion, the surface water is of good quality and there is good sediment control [10]. No further physical remedial work is currently anticipated.

BENEFITS and LESSONS OF IN-PIT DISPOSAL AT WHISTLE MINE

The Whistle Mine experience has provided a guide for mine and closure planners to work closely together from the outset of a mine’s development to optimize waste handling for the most effective reclamation [8].

The Whistle pit disposal experience demonstrates a successful approach to rehabilitating a mine site which includes oxidized waste work and an elevated pit profile. Fortunately, all of the potentially problematic waste rock volume was slightly less than the original pit volume and drainage from the pit could be efficiently collected. The two key aspects in the success of the pit disposal were the liming of the oxidized waste rock on disposal and the application of a well-researched and designed engineered cover.

While there are no standard procedures for backfilling a pit with ARD and metal leaching waste rock and designing such soil covers, several steps were successfully followed, including:

A. Waste Rock
   1) quantify and characterize;
   2) relocate to pit in a compact condition; and
   3) clean residual contamination from temporary waste rock storage locations,

B. Cover Design

1) define cover function;	7) monitor test cover performance;
2) define design constraints;	8) calibrate predictive model;
3) scoping level design;	9) design and construct full-scale covers;
4) collect design data;	10) monitor cover performance;
5) predictive modeling;	11) recalibrate predictive model; and
6) construct test covers;	12) design and implement long-term maintenance.

RESIDUAL ISSUES

Long term collection and treatment of seepage water is required at Whistle. The volume of water being treated is small and the concentration of metals and acidity is declining with time. In June 2009, Vale was presented with the Tom Peters Memorial Mine Reclamation Award for the Whistle mine reclamation at the second annual Ontario Land Reclamation Association and Ontario Mining Association Conference [10].

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CASE STUDY #2 OWL CREEK MINE

SYNOPSIS

Owl Creek Mine was an open pit gold mine northeast of Timmins, Ontario operating from 1981 until 1989, extracting more than 1.6 Mt of ore and more than 4 Mt of waste rock. Overburden and waste rock were stockpiled near the pit at three large surface dumps: the North, Southeast Overburden, and South Clay dumps. The mining of Owl Creek required a minor diversion of the adjacent Porcupine River. The pit was approximately 300 m by 500 m and 100 m in depth at the end of mining. Exploration openings were developed below the pit and these developments were hydraulically connected to the pit bottom. No significant amount of ore was removed from the underground workings.

In 1990 an evaluation of waste rock from the Owl Creek pit, determined that the material was both acid generating and potentially acid generating (PAG), with associated metal leaching. The evaluation also determined that some of the wastes were acid consuming.

Five options for managing the waste rock and closing out the pit were evaluated. The selected option consisted of backfilling the bottom of the open pit (about 80 m deep) with waste rock interlayered with limestone. Over 3 Mt of waste rock was relocated to the pit and the final layer was capped with a clay-rich soil. The pit was then allowed to flood; flooding was accelerated by pumping water from the Porcupine River. Waste rock placement in the pit was completed in 1992 and water levels 5-6 m above the waste rock were achieved that year. Three years later the water depth was 8 metres. In later years, the pit lake depth was maintained between 22 and 23 m depth.

About 1.7 million tonnes of waste rock remained on surface with the majority of this mixed with acid consuming overburden in one of the surface dumps. Some waste rock had been used for mine roads and the pit diversion dams. Portions of this waste rock had been identified to be PAG and this was excavated for inclusion in the pit disposal.

From 2000 to 2012, the neighbouring Bell Creek Mine used the Owl Creek pit lake as a final effluent polishing pond. This effluent contained slightly elevated levels of copper, minor levels of nickel and arsenic and varying levels of other parameters including suspended solids. Despite these additions, the Owl Creek pit surficial water quality has been below Ontario effluent regulation limits. Environmental Effects Monitoring (EEM) studies of the river have also shown no negative effect from Owl Creek pit water effluent. Discharges to the nearby Porcupine River were managed by pumping at times when the river had the capacity to receive addition flows of approximately 100,000 m^3/month.

The Owl Creek pit lake has been and continues to be a meromictic water body with a strong stratified layer present in the lake at a depth of 8 m, possibly due a higher concentration of dissolved salts and metals in the lower levels including Cd, Co, Fe, Pb, and Ag and principally sulphate, present at depth.

The Owl Creek pit surface water quality has been consistently below regulation limits for many years. In spite of the use of the pit lake as a polishing pond, water could be discharged without the need for chemical treatment. Groundwater monitoring shows a similar trend, as has mine site surface water runoff from the remaining waste rock piles.

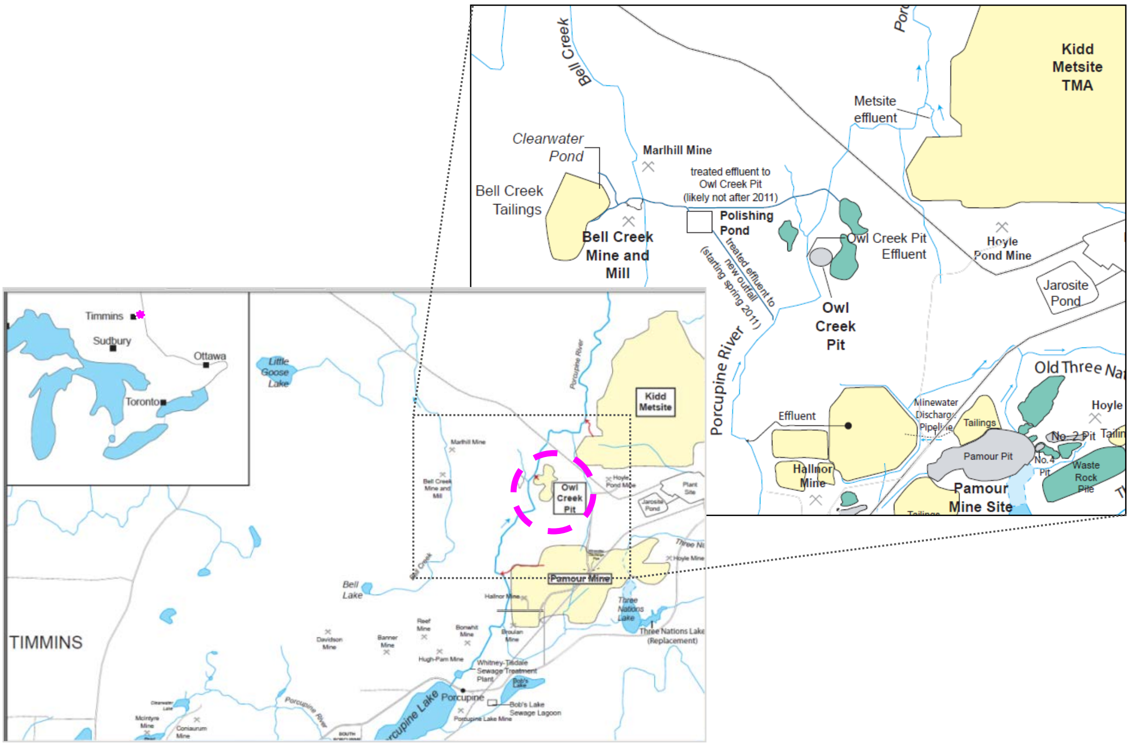
The strategies employed at Owl Creek have resulted in the isolation of potential contaminants with the placement of the most reactive rock into the bottom of the pit. Also, the lower zone of pit column appears to have acted as a sink for metals (e.g. copper) contained in the effluent from the nearby Bell Creek mine facility.

The backfilling of the Owl Creek pit with PAG waste rock has been a very successful strategy. No remediation measures or water treatment have been required in the 21 years since the in-pit disposal was completed.

LOCATION

The closed-out Owl Creek mine is located 18 km northeast of Timmins, Ontario as shown in Figure 2-1. It is located near the west end of the mineral-rich Neo-archean Abitibi greenstone belt, and 4 km north of the Destor Porcupine fault [13]. This region has been the location of a number of base metal and gold mines over the last 100+ years. The open pit mine site is bordered by the small Porcupine River on the west and the Owl Creek drainage system to the east. The area receives on average 830 mm of precipitation annually, 300 mm of which is snow; pan (lake) evaporation is about 530 mm. Average maximum and minimum temperatures are 24^{o}C in July and -23^{o}C in January [11].

Figure 2-1 Location of Owl Creek Mine and Neighbouring Mines, Northeast of Timmins, Ontario [11]



MINE TYPE and HISTORY

Prior to open pit mine development, between 1979 and 1980 an exploration decline targeting base metals was developed [11]. Open pit mining commenced at Owl Creek in 1981 and continued until 1989. The underground exploration ramp and drifts are believed to have been hydraulically connected to the pit bottom.

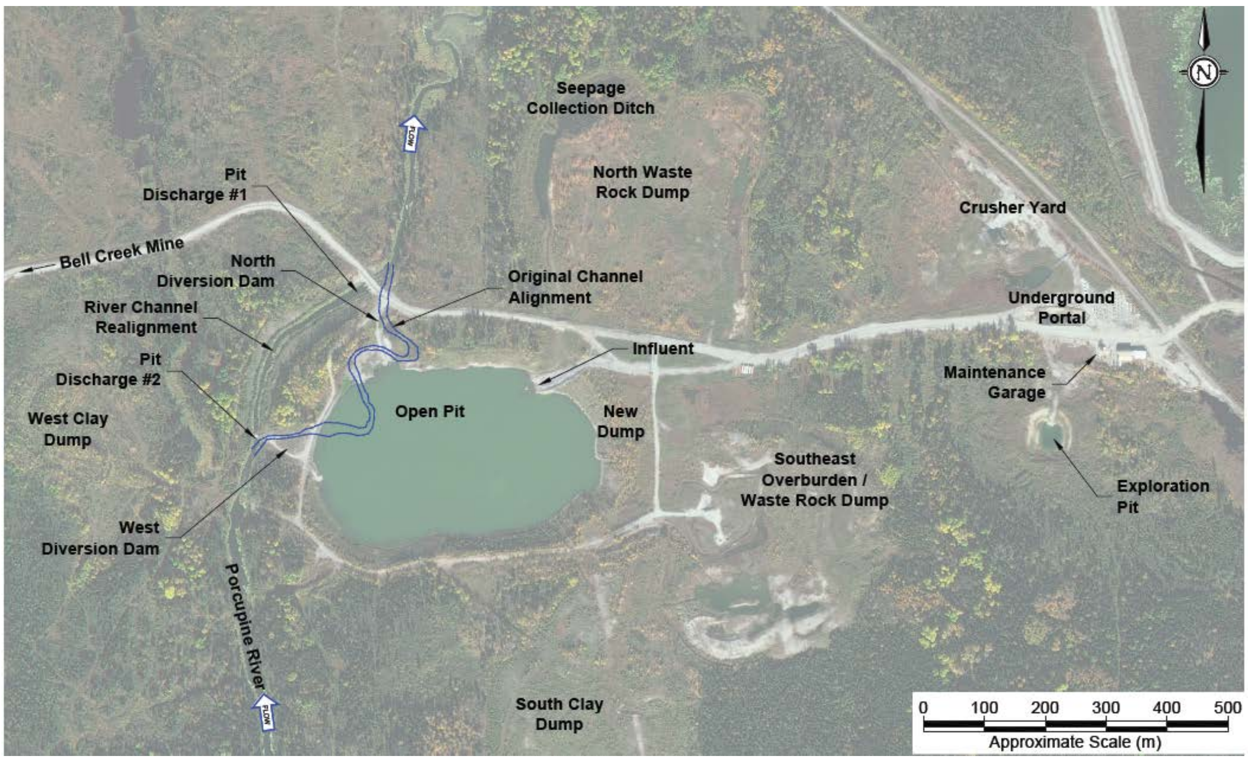
The mine produced 223,671 ounces of gold from 1,623,178 tonnes of ore, or approximately 0.14 oz/tonne [11]. The gold ore was initially processed at the Kidd Creek Metallurgical Plant, and later at the Bell Creek mill [13]. Ore high in argillite graphite was sent to the Pamour Porcupine Mines Ltd., Schumacher Division Mill for processing [11].

During operation, Owl Creek ownership moved from Kidd Creek Mines Ltd to Falconbridge Gold Corporation. After mining ended, ownership was assumed by Kinross, Kinross Gold Corporation which in 2002 amalgamated its Timmins operations with those of Placer Dome Limited. Since 2007, responsibility for the Owl Creek property has been assumed by Goldcorp's Porcupine Gold Mining facilities in Timmins.

To facilitate mining, in 1981 a short diversion of the Porcupine River was constructed by the installation of two diversion dams – one on the north and one on the west side of the pit (Figure 2-2). Dry conditions were maintained in the pit by the usual practice of pumping from a pit bottom sump [1]. Mine effluent was directed to a Mine Water Storage Pond before being released to the Porcupine River.

About 7.4 Mt of material ore and waste were excavated, approximately 3.4 Mt of overburden and 4.4 Mt waste rock [13]. The waste rock from the pit and (underground exploration activity) were stored in the former North Waste Rock Dump (which covered 12 ha), the South Clay Dump, and Southeast Overburden/Waste Rock Dump (also covering 12 ha) (Figure 2-2). Some of the waste rock had been used for road and Porcupine River diversion dam construction.

Figure 2-2 Aerial View of the Owl Creek Mine (August 2012) [11]



In 1990, ARD was detected during a mine inspection in the North Waste Rock dump. The ARD appeared to primarily originate from the graphitic argillite, which contained pyrite, as well as marcasite, pyrrhotite and other sulphides (typically <5% sulphide content) [11]. Parts of all 3 of the waste rock dumps were found to have potential to generate acid (PAG).

Following sampling and testing of the wastes and examination of management options, it was decided that the reactive waste rock would be placed in the pit with alkaline rock supplements and be allowed to flood. This occurred between early 1991 and August 1992. Other activity included backfilling the underground ventilation raise and blocking access to the exploration portal which will later be backfilled and sealed. Following the backfilling of the Owl Creek pit with the acid generation and PAG waste rock, the pit was allowed to flood. (The time of flooding has not been determined in this review of Owl Creek.) Water was also pumped in from the

Porcupine River to accelerate flooding. In recent years, the water levels were maintained slightly below that of the Porcupine River.

Approximately 1.7 Mt of waste rock remains on surface at Owl Creek, some in the mine roads [11]. About 1 Mt is mixed in with an equal amount of acid consuming overburden contained in the South Clay and Southeast Overburden/Waste Rock dumps. These two dumps were regraded during remediation and sloped towards the pit to direct and promote natural drainage to the pit should metal leaching and/or acid generation occur.

An unusual and perhaps significant feature of the Owl Creek situation has been the use of the pit lake as an effluent polishing pond by the nearby Bell Creek mine and ore processing facility (Figure 2-3). Between 2000 and 2011, the Bell Creek facility has discharged up to between 250,000 and 300,000 m^3/yr[14] of effluent to the Owl Creek pit for final polishing. A lesser amount of effluent from surface runoff and tailings pond supernatant was discharged from Bell Creek during a state of mine inactivity from 2002-2009 [11]. From 2002-2003, some water from the Marhill Mine was pumped to Owl Creek. The water level in the Owl Creek pit has been maintained by periodically pumping to the Porcupine River. The objective was to maintain pit lake levels between 277 and 280 masl (metres above sea level) [11]. No water treatment has been required.

Figure 2-3 Aerial Photo of Bell Creek Mine and Owl Creek Pit [13]

The Owl Creek pit lake is no longer required (as of mid-2011) as a polishing pond by Bell Creek. A final closure/remediation activity will be the construction of a robust spillway to the Porcupine River. This is to be constructed on the north side of the pit. This will represent the beginning of the last chapter of the success story of in-pit disposal at Owl Creek. No pit lake water treatment is anticipated to be needed.

[14] Data provided by Porcupine Gold staff, November, 2013.

Figure 2-4 Photo of Owl Creek Pit Lake, May 2010 (Feasby)

Influent Water Pipes (Bell Creek Mine effluent)

The chronology of the Owl Creek operation and closure is shown below in Figure 2-5.

Figure 2-5 Timeline of Activities of the Owl Creek Mine

1981	• Overburden removal and mining commenced at Owl Creek • The Porcupine River was diverted • Waste Rock was used for open pit diversion dams and mine site roads
1985	• Falconbridge Ltd (Falconbridge Gold Corp.) obtained mine ownership
1989	• Owl Creek mining ended
1990	• ARD detected • Assessment of Waste Rock • Remediation options considered
1991	• Pumps for underground exploration decline were shut off • Backfilling of pit & remediation began
1992	• (August) Pit backfill/remediation completed • (September) Natural flooding level was at a 6 m depth
1993	• Owl Creek Mine now owned under Kinross Gold Corporation
2000	• Bell Creek Mine commences use of Owl Creek pit as a polishing pond • Initial documentation of monitoring of Owl Creek Mine & pit began
2002	• Kinross amalgamated Timmins operations with Placer Dome Ltd. Shared interests- 51% Kinross; 49% Placer Dome • Marlhill Mine commences use of Owl Creek pit as a polishing pond • Bell Creek facility commences a period of inactivity (continues effluent discharge)
2003	• Marlhill Mine ends use of Owl Creek pit as a polishing pond
2004	• Bell Creek Mine effluent treatment system improvements completed (monitoring data show more stable parameters)
2006	• Goldcorp Canada Ltd. purchased Placer Dome's ownership share
2007	• Goldcorp Canada Ltd. purchased Kinross Gold Corp. Timmins share (Goldcorp Canada Ltd. now has ownership of Owl Creek Mine)
2009	• Bell Creek Mine restarts mine activity
2011	• Owl Creek pit water quality and effluent are stable and below regulation limits • Pit lake depth measures at ~23 m • Bell Creek effluent discharge ended
2023	• *Anticipated commencement of remediation and mine closure of Owl Creek Mine and pit at or before this time*
2024	• *Mine expected to complete remediation and closure measures* • *Implementation of a 1-10 yr & 11-100 yr monitoring program (stability, biological, & chemical -ground, surface runoff, & pit discharge waters-monitoring)*

PIT DESCRIPTION

The original mined-out pit has a surface expression of approximately 300 m by 500 m and was 100 m deep. The mined out pit is shown in Figure 2-6 [11].

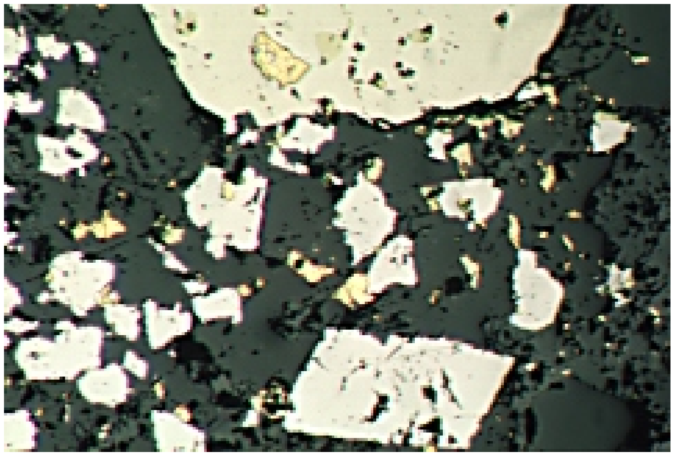
Figure 2-6 Photograph of the Owl Creek Open Pit (looking westward) Prior to the Relocation of Waste Rock to the Pit [13]

The Owl Creek gold deposit occurred in metasedimentary and mafic to ultramafic metavolcanic rocks from the metamorphosed late Archean Abitibi green schist facies [13]. The gold mineralization was principally hosted within stockwork zones of quartz-sulphide-carbonate.

The main ore zone for the open pit was massive/pillowed basalt surrounded to the north, east, and south by metasedimentary rocks (primarily greywacke and argillite) with thin graphitic horizons at the north and south of the basalt [13]. The pit surface stratigraphy was alternating till and glacial fluvial beds (10 m-25 m thick), overlain by 5 m of glacio-lacustrine clay and capped by a 1-2 metre carbonate rich till [14].

The majority of the gold was present in inclusions in pyrite but the highest grade ore was free gold in quartz veins (Figure 2-7) [13]. The dominant sulphide minerals were pyrite (FeS_2), marcasite (FeS_2), pyrrhotite (Fe_7S_8), arsenopyrite (FeAsS), and chalcopyrite ($CuFeS_2$). The sulphide content was generally less than 5% but also found in excess of this amount in the graphitic argillite and brecciated metasedimentary rocks (Kingston, 1987) [13][14].

Figure 2-7 Example Sulphides (white) and Gold (yellow) Content - Owl Creek Mine [17]



WASTE PRODUCTION

An estimated 7.82 Mt of rock was excavated from the pit, 3.4 million of which was overburden (till and varved clay) and 4.77 million tonnes was waste rock [13]. About 4.1 Mt tonnes of waste rock was stored in the North Waste Rock Dump, South Clay Dump, and Southeast Overburden/Waste Rock Dump. Approximately, 0.5 Mt of waste rock was used in road construction and 0.2 Mt for the Porcupine River diversion. Waste quantities are listed in Table 2-1 [11].

Table 2-1 Summary of Waste of the Owl Creek Mine Operations (10^6 tonnes) [13]

	Overburden	Waste Rock		Amount Relocated to Pit
		Greywacke/ Volcanics	Graphitic Argillite	
North Dump	0.5	1.5	0.3	2.289
Southeast Dump	1.00	1.5	0.2	0.97
South Dump	1.05	0.01	0.015	0.02
West Dump	0.85	0	0	0
On-site Roads	0	0.5	0.015	0.369
River Diversion	0	0.17	0	0
Diverse Sources	0	0.555	0.005	0
Total	**3.4**	**4.235**	**0.535**	**3.648**
		4.77		
Total excavated material	**7.82**			

ACID GENERATION IN MINE WASTES

A 1990 mine site inspection detected ARD seeping into the Porcupine River from the North Waste Rock Dump. Samples of the drainage identified a strongly acidic pH of 2.2 and elevated concentrations of Zn, Cu, Ni, Fe, and several other elements [13].

The water sampling results prompted an ARD assessment of all Owl Creek wastes on surface (no tailings were present at Owl Creek). The assessment evaluated the ARD potential of each waste rock location – this included testing of grab and borehole samples from the waste dumps, roads and entire mine site. The evaluation included an assessment of mineralogy, geology, hydrogeology, and hydrology to determine acid consuming and generating potential, and quality of runoff (pH, TSS, and sulphate) of the waste rock, and mine site.

Fifteen bore holes were drilled in the North Dump (ranging from 3 m to 12 m in depth) to determine acid production potential and neutralization potential (kg $CaCO_3$/tonne) of the waste rock as outlined in Table 2-2. The graphitic argillite was the main acid generating rock with a net *maximum acid production* (NAP) ranging from 60 to 289 kg $CaCO_3$/tonne [13]. Sulphide minerals in the graphitic argillite, was determined as anhedral crystals and amorphous nodules. Both were observed to oxidize readily. The iron sulphide content was observed to be to 20% in the argillite.

The *net acid production* results from other waste rock and overburden ranged from 18 to 123 kg $CaCO_3$/tonne. Test results showed that the *neutralization potential* (NP) of basalt (133 kg $CaCO_3$/tonne) and greywacke (105 kg $CaCO_3$/tonne) were significant. They would not be an acid generating problem if they were stockpiled separately from the graphitic argillite. The same was seen for overburden with an acid consuming ability of 91 to 178 kg $CaCO_3$/tonne (an average of 142 kg $CaCO_3$/tonne) [13][11].

The North Dump was well-mixed with PAG and Non-PAG, so the entire dump was considered a potential ARD source. The Southeast Overburden/Waste Rock and South Clay dumps showed strong acid generation from graphitic argillite but minor amounts of ARD was detected, presumably as a result of integral mixing with acid consuming overburden. The overburden in the Southern dumps consisted of glacial till, glacial fluvial sediments, and clay.

Table 2-2 Evaluation of Owl Creek Waste Rock Neutralization and Acid Potential in kg of $CaCO_3$ per tonne [11]

Location	Description	Total Sulphur %	Neutralization Potential (NP) kg $CaCO_3$/t	Acid Potential (AP) kg $CaCO_3$/t	Net Neutralization Potential kg $CaCO_3$/t	NPR	ARD Potential
Former N Waste Rock Dump	**Graywacke/ Buff Mafic**	0.99 - 1.15	44 - 66	31 - 36	8-35	1.2 - 2.1	PAG
	Graywacke/ Buff Mafic	0.56	113	18	96	6.5	Non-PAG
Road Material	**Graywacke**	0.20	111	6	105	18	Non-PAG
Road Material	**Buff Mafic (Basalt)**	0.59	152	18	133	8.2	Non-PAG
New Dump	**Graphitic Argillite**	2.49 - 7.03	18 - 22	78 - 220	-202 to -56	0.1 - 0.3	PAG
SE Overburden Waste Rock Dump	**Graphitic Argillite**	0.8 - 6.21	7.1 - 43	25 - 194	-170 to -15	0.1 - 0.6	PAG
Road Material	**Graphitic Argillite**	9.24	-46	289	-335	-0.2	PAG
Ore Stockpile	**Ore**	1.36	185	43	142	1.42	Non-PAG
S Clay Dump	**Unknown**	3.95	82	123	-42	0.7	Non-PAG

NPR-Ratio of Neutralization Potential to Acid Potential

While elemental analysis determined that the North Dump waste rock contained 1% sulphur or less, the drainage indicated significant ARD as shown Table 2-3.

Table 2-3 1990-1991 Water Quality from the North Waste Rock Dump [13]

Parameter	pH	Sulphate	Zn	Fe	Al	Cu	Ni	As
(mg/L)	2.2	25,000-47,400	11,100-18,800	2,810-8,820	1,240-2,090	36-120	24-44	9.9-37

REMEDIATION OPTIONS

A number of options were considered for the best management of ARD at Owl Creek. These options are outlined as follows:

Table 2-4 Waste Management Options at Owl Creek [13]

Option	Description	Estimated Cost (1991 C $)
1	Continuous ARD collection and treatment	$7,400,00
2	In-situ flooding	$ 10,000,000
3	Disposal in a tailings management area	$ 9,000,000
4	Geomembrane capping	$ 7,600,000
5	Disposal in the Owl Creek open pit	$6,200,000

Option 1 - This option called for a long-term commitment for continued collection, pumping and treatment of the acidic drainage. This would be accomplished by pumping to the existing treatment facilities at the Kidd Creek tailings area. This would require the disposal of 6 Mm^3 of treated sludge produced over a century. Drawbacks to Option 1 included; a) the Kidd Creek tailings area would be decommissioned at some time in the future, necessitating a new sludge disposal area, b) the large volumes of treatment sludge would create a loss of valuable tailings storage capacity, and c) the acid generating process would not be inhibited. The estimated cost in 1995 was $7,400,000 [13]. (This option would not likely be viable today (2013) with changes in property ownership.)

Option 2 - In-situ flooding would require a low permeability dyke to be constructed around the North Dump. The waste rock stockpile would be contoured, capped and maintained as a pond. The waste rock would be kept saturated and treated with a lime slurry to inhibit acid production. This process would require in perpetuity monitoring of pond levels and a seepage monitoring program. The estimated cost in 1995 was $10,000,000 [13]. (This option did not include consideration of PAG wastes located outside of the North Dump.)

Option 3 - Reactive waste rock would be relocated to a nearby active tailings disposal area, then covered by tailings and eventually an elevated water table. The waste rock would be limed until it was covered by saturated tailings. Scheduling could be problematic leading to acidic runoff collection and treatment costs if disposal areas were not available at the time of disposal. The estimated cost in 1995 was $9,200,000 [13].

Option 4 - Geomembrane capping of the North Dump; a composite cover consisting of topsoil, geotextile, a gravel drain layer, a geomembrane, a sand bedding layer, and geotextile overlaying a base layer. Drains would be installed to reduce topsoil and drainage layer erosion and the groundwater would be monitored. There was no certainty that Option 4 would effectively inhibit the acid generating process in perpetuity and did not include long-term maintenance costs. The estimated cost in 1995 was $7,600,000 [13].

Option 5 - Reactive waste rock would be relocated into the Owl Creek open pit where it would naturally flood in 1 to 2 years to an elevation that would submerge the waste rock and expect to inhibit further acid generation and metal leaching. The amount of acidity already released by the rock was unknown but it was estimated that 9 kg of crushed limestone/tonne of waste rock would neutralize it. Dewatering the underground exploration workings was to be discontinued to minimize the hydraulic gradient and interchange of water. In the unlikely event that the water level rose excessively, the water could be controlled and treated with lime, prior to release. The estimated cost in 1995 was $6,200,000 [13].

SELECTION AND IMPLEMENTATION OF REMEDIATION STRATEGY

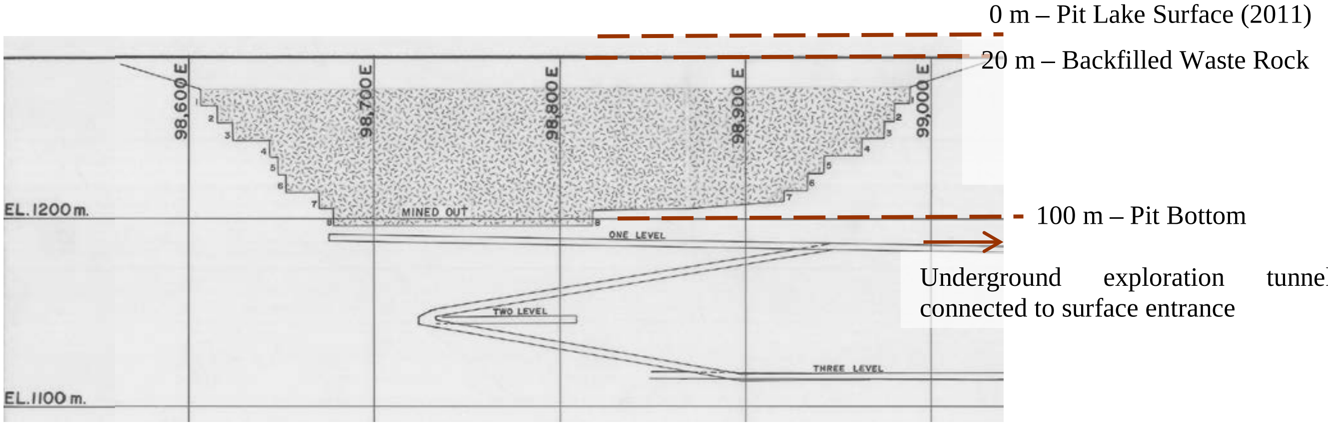
Remediation Option 5 (relocation of waste into pit and pit flooding) was selected because it could be implemented sooner and required the lowest cost. It was perceived to be a walk-away type decommissioning plan whereby the existing acidity could be neutralized and the acid generation would be inhibited. The pit had sufficient capacity to contain all the PAG waste rock.

During pit rehabilitation, the overburden walls were angled back approximately to 30° [11]. The pit slopes contain 3 layers of sandy gravel overlying a clay layer, and subsequently a silty sand layer. The pit walls were stabilized by placing the waste rock at an elevation just a few metres below the original bedrock surface.

In January of 1991, the pumps for the underground exploration decline were shut off, and the underground workings began to flood naturally [13].

Beginning in December 1991, 3,648,000 wet tonnes of waste rock from the North Dump, some PAG from the Southeast Dump, and graphitic argillite from other waste disposal locations were spread across the floor of the Owl Creek open pit in 1.5 m lifts, alternating with 5.4 kg of minus 95 mm (3/8 inch) crushed limestone per tonne of waste rock, which was spread with a loader and a grader. This represented an addition of approximately 15,400 tonnes of limestone [11]. These layers were finalized with a one metre top layer of clay till [11], and the final backfilled height in the pit was at 80 m (Figure 2-8). The pit was then left to flood naturally with some pumping of water from the Porcupine River.

Figure 2-8 Cross Section of the backfilled Owl Creek Pit and Underground Workings, looking north



The general site was also remediated, including the diversion of surface water runoff and seepage into the pit by regrading of the pit waste dumps. This promoted natural flooding which would prevent ARD reaching the Porcupine River.

By September 1992, the remediated pit water level was about 6 m above the level of the waste in the pit. In 1995, prior to Bell Creek effluent discharge, the pit lake depth was approximately 8 m [11]. The water level has been subsequently maintained at about 20-23 m above the level of waste rock (as of 2011), by periodic pumping to the adjacent Porcupine River [11].

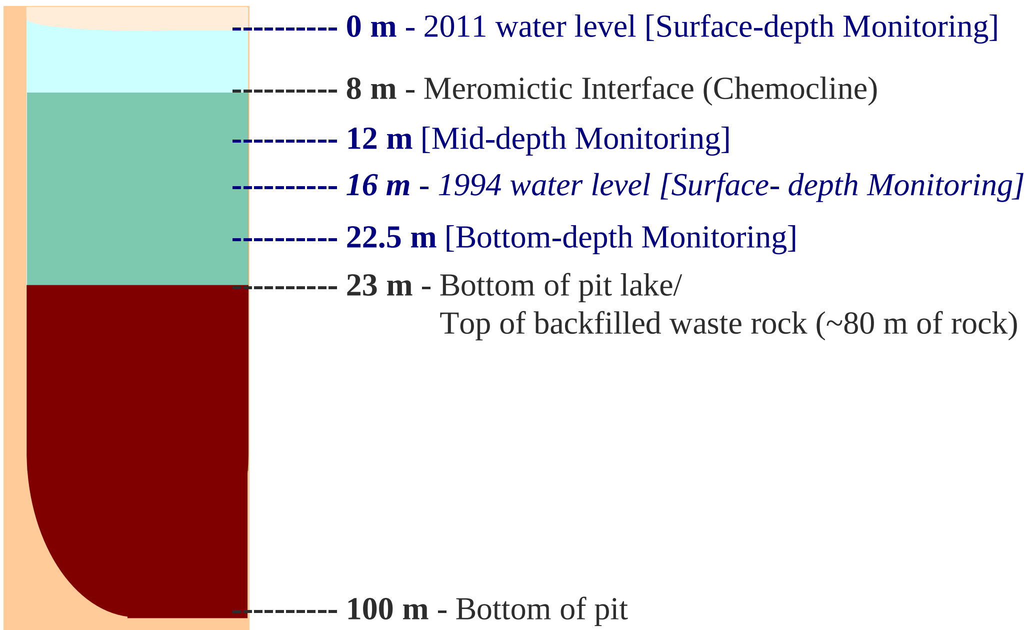
Starting in 2000, the neighbouring Bell Creek Mine facility used the Owl Creek pit as a polishing pond for mine effluent. Eleven years later, in 2011, the Bell Creek Mine operated by Lakeshore Gold constructed a dedicated polishing pond and ceased use of the flooded Owl Creek pit. The Marlhill Mine, northwest of Owl Creek, discharged mine effluent to Owl Creek for an unspecified period of time in 2003.

Available data needed to complete a water balance for the pit are incomplete. What is known is that ground water inflow and net precipitation totals approximately 350,000 m^3/yr [11]. This is slightly more than volume of water received from Bell Creek in recent years.

PERFORMANCE OF IN-PIT DISPOSAL AND PIT LAKE WATER QUALITY

The Owl Creek pit lake is a meromictic lake that is roughly 23 m deep (Figure 2-9) after being backfilled with waste rock. Pit water turnover has not been detected during monitoring since 2000 [11], and recent work by Goldcorp indicates that turnover is unlikely at any time (pers. comm. T. Sulatycky, Goldcorp 12-2014). The upper 8 m of water are responsive to climatic conditions, maintaining between 60-100% oxygen saturation depending on the season, and is believed to mix during winter months [11]; however, waters below the 8 m stratification (the chemocline), have remained anoxic (<20% saturation) [11] and stagnant for many years as evidenced by the records outlined below, including conductivity and temperature profiles depicted in Figure 2-10.

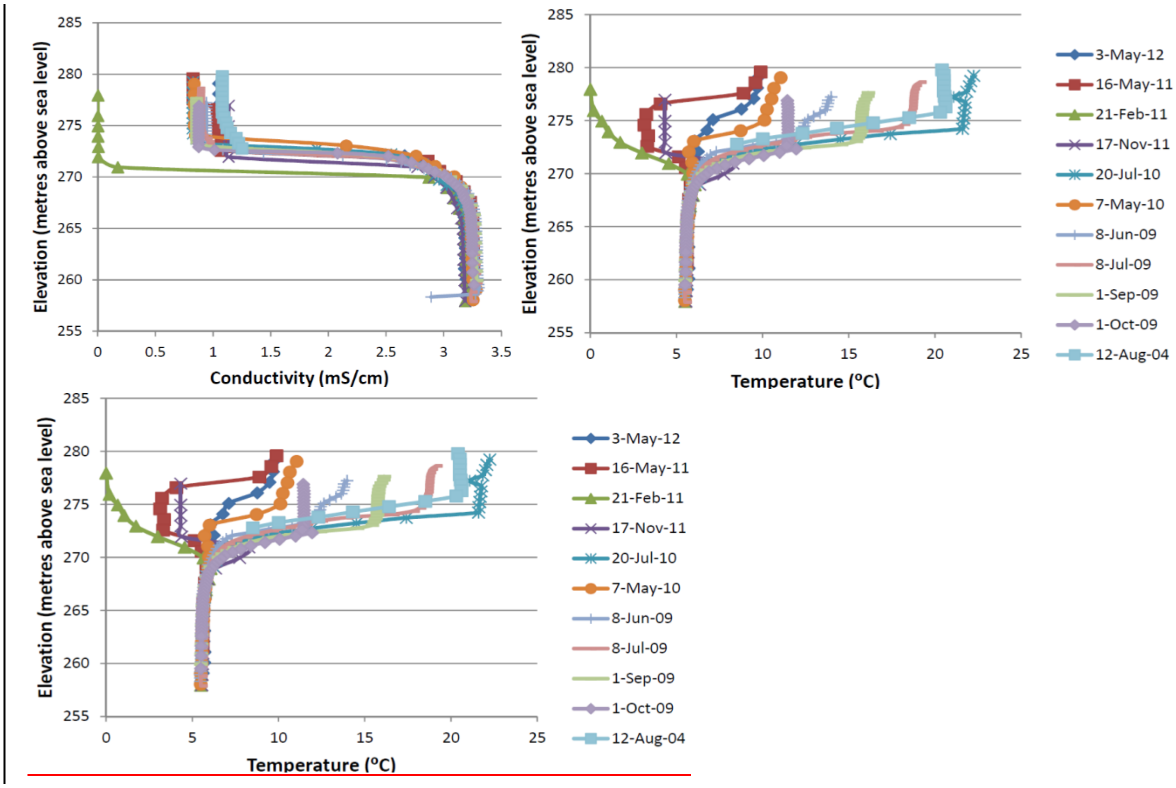
Figure 2-9 Schematic Cross-section Illustration of Owl Creek Pit and Pit Lake[15]



Extensive monitoring data on surface waters, effluent, groundwater seepage and adjacent Porcupine River parameter concentrations indicate that there is containment of the waste rock oxidation products and metal leaching within the lower levels of the Owl Creek pit.

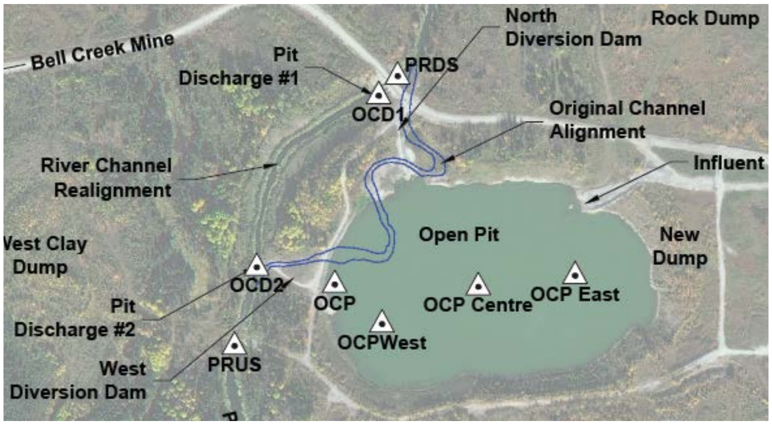
[15] Monitoring depths are estimated depths extrapolated from Goldcorp, 2013 [11] source descriptions**.**

Figure 2-10 Conductivity and Temperature Profiles of the Owl Creek Pit Lake. (Note the chemocline at ~8 m depth (270 masl) [11]



Data collected since 2000 shows the majority of dissolved metals are present in very low concentrations within the pit at all depths. At depth (below the chemocline) there is some increase in metal content and a recorded increase in conductivity. Table 2-5 compares early pit water quality (1994) with more recent sampling (2011) and it shows a significant decrease in most contaminants of potential concern, except chloride and sulphate. The 2011 sampling shows that water above the chemocline generally meets the Ontario Provincial Water Quality Objectives (PWQO) with the exception of silver, cadmium and cobalt, which are slightly above the PWQO. Below the chemocline, metal values are slightly increased and nickel, zinc, silver, cadmium, cobalt and uranium also slightly exceed the PWQO. The PWQO are important as they represent the close out criteria for the pit lake.

Figure 2-11 Monitoring Locations of the Owl Creek Pit Lake Waters. (1999 discharge data is from OCD1 and 2000-2011 data is from OCP) [11]



When comparing 1999 water quality to later years, it should be noted that surface water data from 1999 has been collected from a monitoring location at an Owl Creek discharge point to the Porcupine River (OCD1), and 2000-2011 data (at all depths) has been taken from a location within the pit (OCP; Figure 2-11).

Table 2-5 Owl Creek Pit Lake Annual Average Water Quality Comparison. Note: Concentrations are in mg/L [13] [11]

Parameters	1994 [13]			1999[1] [11]	2011[1] [11]			Target Surface Water Quality (PWQO)[3]
(Depth from 2011 lake levels- Figure 2-9)[2]	**Surface** (16 m)	**Mid depth** (20 m)	**Bottom depth** (23 m)	**Surface** (3 sites)	**Surface** (0 m)	**Mid depth** (12 m; Note: chemocline is at 8 m)	**Bottom depth** (23 m)	
pH	7.2	7.2	7.2	8.03	8.0	6.7	6.5	6.5-8.5
Temp.(^{o}C) [Summer]	18.9	19.7	16.9	-	22.2	9.5	10.3	-
Chloride	1.8	2	2	-	16.7	14.8	13.5	-
Sulphate	690	740	678	-	300.5	611	998	-
Phosphorus	0.08	0.01	0.02	-	0.011	0.003	0.003	0.02
Copper	0.03	0.02	0.02	0.024	0.003	0.001	0.003	0.005
Nickel	0.10	0.14	0.11	0.23	0.02	0.07	0.073	0.025
Lead	0.01	0.01	0.01	0.002	0.007	0.015	0.01	0.025
Zinc	0.13	0.20	0.14	0.005	0.02	0.04	0.04	0.03
Iron	0.18	0.67	1.54	0.077	0.008	0.015	0.008	0.3
Silver	0.01	0.01	0.03	-	**0.0002**	0.0002	0.0005	0.0001
Aluminum	0.14	0.17	0.38	-	0.041	0.035	0.016	0.075
Barium	0.02	0.02	0.02	-	0.02	0.02	0.03	-
Cadmium	0.01	0.01	0.01	0.002	**0.001**	0.001	0.002	0.0002
Cobalt	0.04	0.05	0.05	-	**0.06**	0.03	0.07	0.0009
Chromium	0.01	0.02	0.01	-	0.002	0.0004	0.0004	0.0089
Uranium	-	-	-	-	0.001	0.011	0.078	0.005

Note - Bold values exceed Ontario PWQO.
1 - Monitoring depths are estimated depths extrapolated from Goldcorp, 2013 [11] source descriptions.
2 - Pit Lake depth in 1994 was 7 m, and depth in 2011 was 23 m.
3 - Ministry of Environment Provincial Water Quality Objectives for surface waters, applicable upon mine closure.

The Bell Creek Mine effluent into the Owl Creek pit contained low levels of most parameters including Suspended Solids, As, Pb, Zn, and Cd, with moderate levels of Cu, and Ni as seen in Table 2-6. [11].

Table 2-6 Quality of Marlhill Mine (2003 only) and Bell Creek Mine Effluent sent to Owl Creek Starting 2000 to 2010 [11]

Parameters (mg/L)	Year (2000-2010)										Regulation Limit[1]	PWQO[2]
	'03**	'00	'02	'04	'05	'06	'07	'08	'09	'10		
As	0.03	0.83	0.250	0.020	0.03	0.020	0.030	0.040	0.030	0.020	0.5	0.1
Cu	0.01	1.20	1.800	0.020	0.02	0.010	0.010	0.020	0.040	0.005	0.3	0.005
Ni	0.01	0.70	0.900	0.080	0.05	0.030	0.020	0.030	0.030	0.010	0.5	0.025
Zn	0.003	0.08	0.025	0.004	0.008	0.012	0.004	0.004	0.016	0.003	0.5	0.03
Cd	0.002	0.002	0.002	0.002	0.0008	0.001	0.0005	0.0005	0.001	0.001	-	0.0002
SS	93.7	38.1	20.7	3.1	5.5	2.8	4.4	4.1	9.7	4.2	-	-
SO_4	108.5	-	-	-	-	-	-	-	-	-	-	-
pH	7.5	8.1	8.3	7.8	8	8	8.6	8.1	7.7	8.2	6.0-9.5	6.5-8.5

**Note 2003 data is from Marlhill Mine effluent, only.

SS - Suspended Solids.

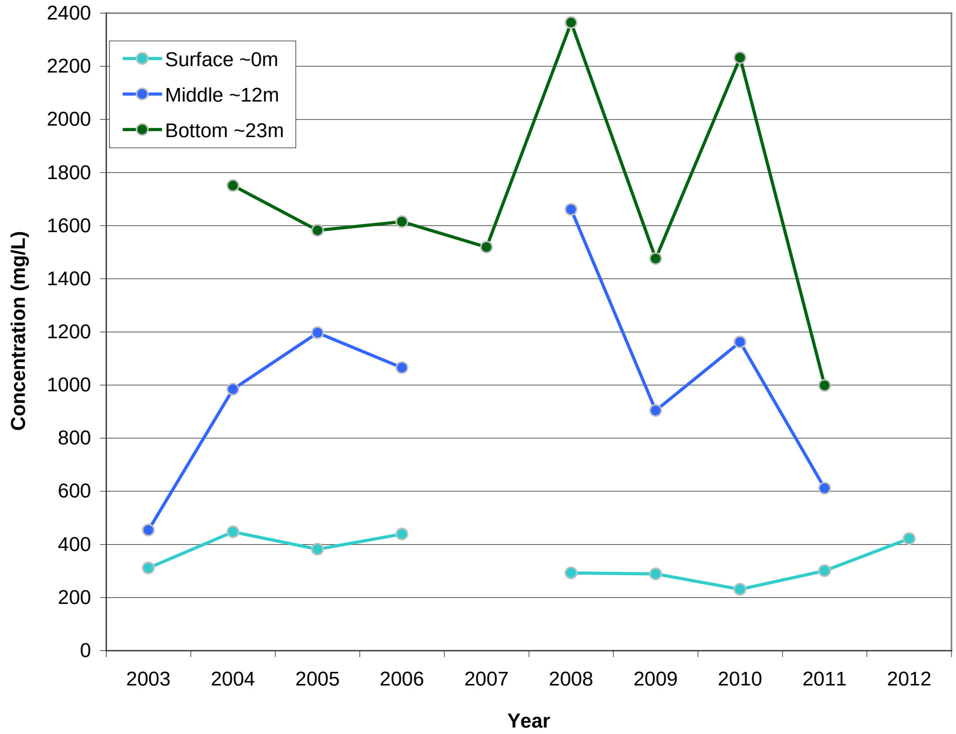
1 - *Metal Mining Effluent Regulations* (MMER) of Environment Canada, in mg/L.

2 - Ministry of Environment Provincial Water Quality Objectives for surface waters, applicable upon mine closure, in mg/L.

Prior to 2004, it appears that the Bell Creek Mine was a major contributor to the dissolved metals detected in the Owl Creek pit lake surface waters as evidenced by a notable stabilization of many parameters in the Owl Creek pit waters since 2004, the year that Bell Creek Mine updated its water treatment system [11]. Additionally, prior to 2004 the Bell Creek Mine effluent maximum suspended solids concentrations were greater than 100 mg/L but were reduced to a maximum of 40 mg/L after 2004 [11]. The Bell Creek mine ceased use of Owl Creek pit in mid-2011 after which Owl Creek pit water quality is expected improve over time until it meets PWQO's.

The bottom of the pit lake is characterized by a relatively low pH and copper levels, and higher amounts of sulphate and suspended solids, and higher levels of As, Cd, Co, Fe, Pb, Ni, and Zn (Figures 2-12 to 2-17). The meromictic conditions of the lake appear to be based on density differentiation of the solids present in the waters. Because of the stratified conditions of the lake, the slightly higher levels of metal in the bottom waters are not expected to report to the surrounding surface or ground waters. It is interesting to note the decreasing trend for the concentration of metals in the surface waters (Figure 2-15).

Figure 2-12 Sulphate Concentration in the Owl Creek Pit Lake[16]



[16] Data from Goldcorp, 2013 [11] source.

Figure 2-13 Concentration of Suspended Solids in the Owl Creek Pit Lake[3]

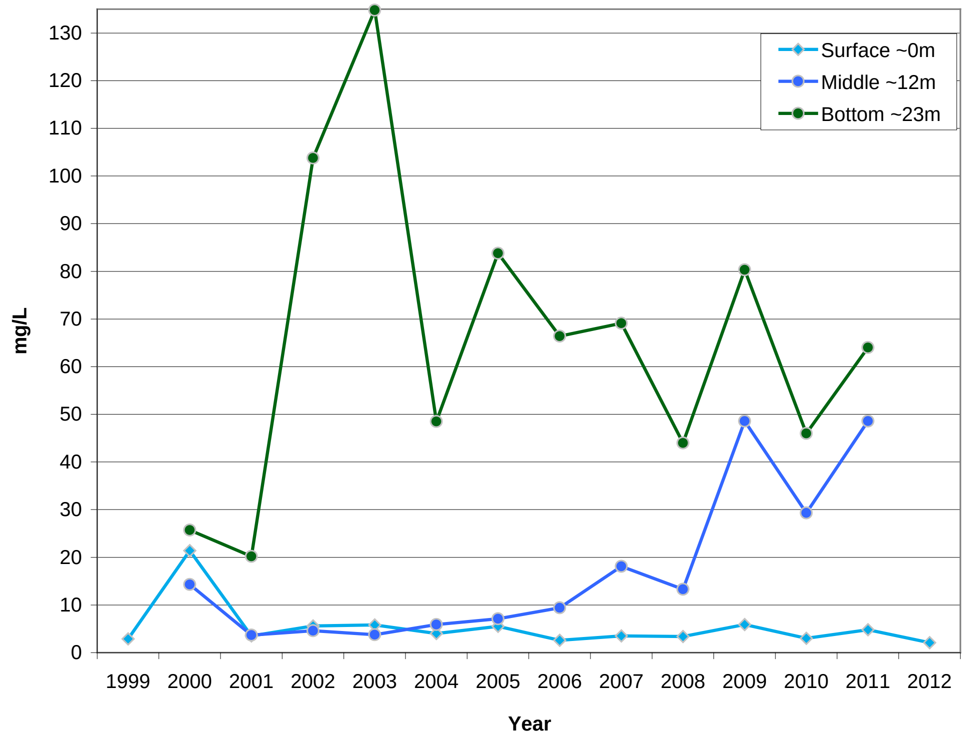
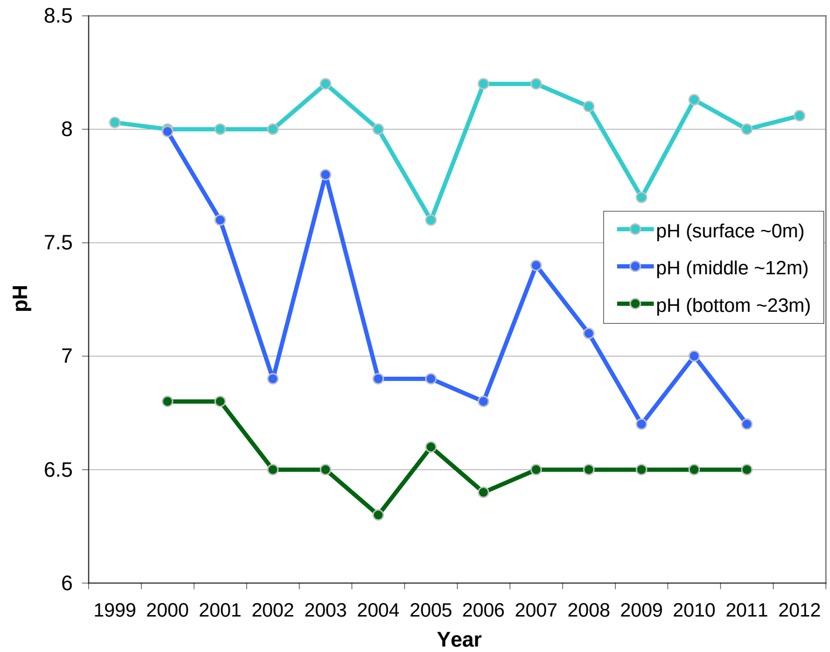


Figure 2-14 pH levels at Three Depths in the Owl Creek Pit Lake[17]



[17] Data from Goldcorp, 2013 [11] source

Overall, metal concentrations have seen a decrease or stabilization over the years as shown in Figures 2-15 to 2-17; nickel has showed the greatest decrease.

Figure 2-15 Surface Water Quality in the Owl Creek Pit Lake [11]

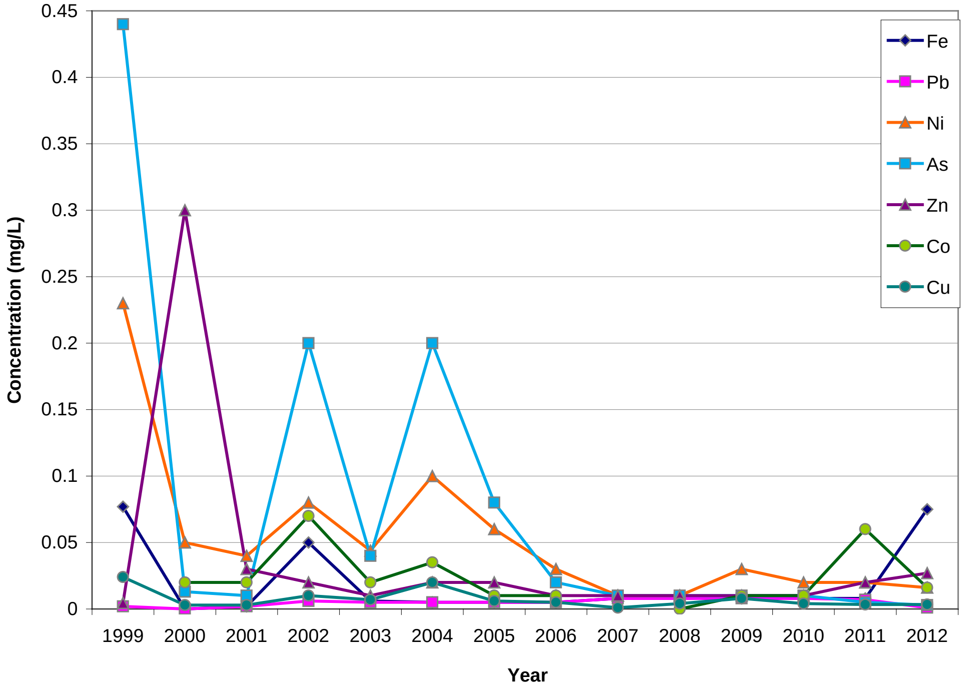


Figure 2-16 Water Quality at a Mid depth (~12 m) in the Owl Creek Pit Lake[18]

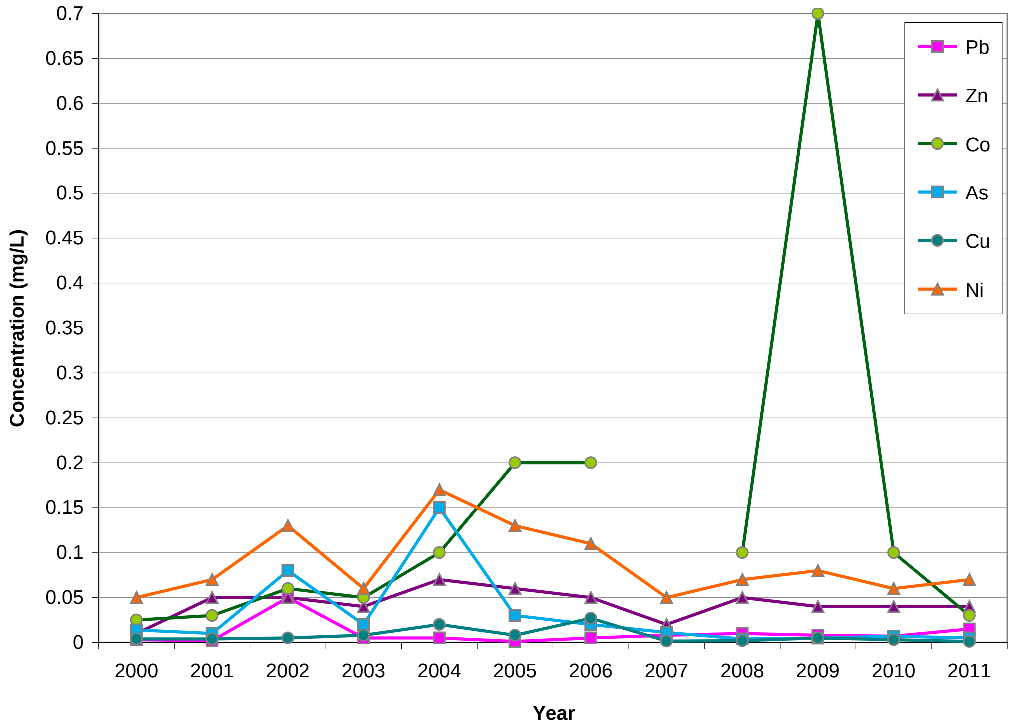
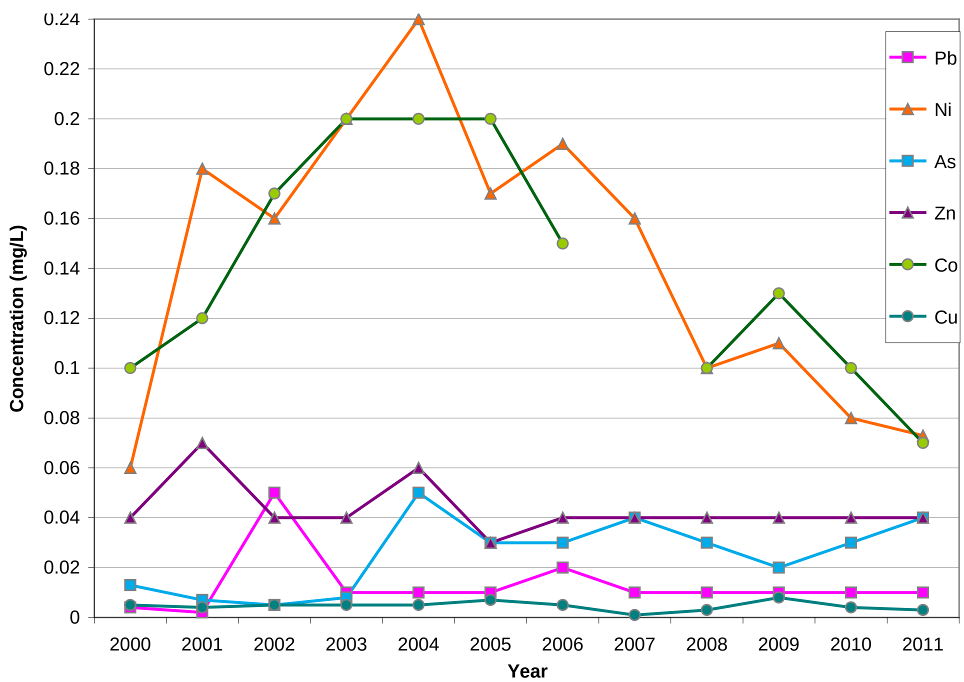


Figure 2-17 Water Quality at a Bottom Depth (~23 m) in the Owl Creek Pit Lake[5]



[18] Data from Goldcorp, 2013 [11] source

The effluent discharge from the Owl Creek Mine continues to meet Canadian national Metal Mining Effluent Regulations (MMER), including toxicity testing. The effluent pH has increased slightly from 7.8 in 2004 to 8.2 in 2011, but remains within the regulatory limits of 6.0 to 9.5 [2]-[9]. The average radium-226 content of 0.006 Bq/L, was far below the regulation limit of 0.37 Bq/L, and the total suspended solids has averaged around 4 mg/L since 2004, which is well below the MMER limit of 15 mg/L.

Environmental effects monitoring of the Porcupine River at the Owl Creek discharge points documented healthy aquatic life indicating the mine effluent has not contained significant levels of toxins.

Groundwater monitoring locations were positioned at various depths around the pit, including within the north and west Diversion Dams. The results from 2011 and 2012 are summarized in Table 2-7 and show generally good quality water for the parameters analysed.

Table 2-7 Groundwater Monitoring Results at the Owl Creek (fall 2011, spring 2012) [11]

Parameters (dissolved, mg/L)	**Location**			
	North (Dam)	**East**	**South**	**West (Dam)**
Fe	2.37 - 7.0	0.013 - 0.53	0.2 - 1.19	1.3
Mn	0.56 - 0.74	0.06 - 0.19	0.07 - 0.18	0.19
As	0.016 - 0.035	0.001	0.0008 - 0.0014	0.0024
Ni	0.0038 - 0.0051	0.0052 - 0.0189	0.0046 - 0.0066	0.0071
Zn	0.001 - 0.0031	0.0004 - 0.0008	0.0017 - 0.0197	0.0015
Sulphate	0.016 - 0.035	107 - 219	12 - 125	32
pH	7.4 - 7.8	7.6 - 7.8	7.6	7.4

REMEDIAL

The only remedial activity planned for Owl Creek will be the construction of a new spillway north of the Owl Creek pit lake to the Porcupine River. A recent water balance done for the site indicates that there is a likelihood that the pit will overflow. As such, a proper spillway is required in order to prevent erosion of the structure and maintain control of the discharge. Water quality is expected to be acceptable in an overflow condition.

Some additional grading of the southeast waste area will be required as part of the closure plan in order to more effectively direct water towards the pit and minimize percolation into the rock, which has some pods of material that are potentially acid generating.

If the meromictic conditions remained in the pit lake as expected, no other remedial activities are anticipated except monitoring to demonstrate the stability of the system.

BENEFITS AND LESSONS OF IN-PIT DISPOSAL AT OWL CREEK

The in-pit disposal of more than 1.6 Mt of oxidized and acid generating waste rock at the Owl Creek Mine has been very successful. Submerging the waste rock in the mined out pit, with alkalinity addition, has demonstrated a robust method of isolating the acid generating and potentially acid generating rock along with oxidation products. The isolation has been assisted by the development of a distinct chemocline at a depth of about 8 m within the pit lake. Current work indicates that this chemocline will be protected from turnover.

In addition, the 20 m deep pit lake has been successfully used as a polishing pond for mine effluent from a nearby facility without compromising the in-pit waste management strategy implemented 22 years ago.

With improving water quality, it is expected that biota would colonize the pit waters. The Owl Creek pit lake apparently supports some active and healthy biological communities, based upon observations of small fish in shallow waters and loons on the pit lake during a 2010 site visit (personal observations, G. Feasby, 2010, Figure 2-18).

Figure 2-18 Loons on Owl Creek Pit Lake [19]

RESIDUAL ISSUES

There are still aspects of the closure plan that need to be completed, including the spillway and the grading of an adjacent waste rock pile. Once these are completed, if water quality trends continue as they are now, it will only be a matter of time before the pit reclamation will be done. In the meantime, monitoring of water quality at the site continues.

[19] G. Feasby, May 2010.

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CASE STUDY #3 SOLBEC MINE

SYNOPSIS

The acid generating wastes and the waste sites at the abandoned Solbec mine site were reclaimed in two phases starting in 1988, first by liming and flooding the tailings, and second by lowering the water level in an open pit, depositing oxidized waste rock and contaminated soils and allowing the pit to reflood. Pit overflow was directed to small ponds and passive biofilters [3].

From 1964 to 1965, 4×10^5 tonnes of ore and waste rock were removed from the pit [1]. During the last years of operations about 10,000 m^3 of acid generating tailings had been placed in the pit. During reclamation an additional 276,000 m^3 of oxidized waste rock and contaminated soil were removed from a 12.1 ha area on surface and placed into the pit. A 5-cm layer of limestone was placed on the waste rock and soil in the pit before flooding occurred.

The pit flooded naturally and in order to inundate all the wastes, a berm was constructed to permanently raise the water level. Overflow has been directed to surge ponds equipped with biofilters to remove any residual metal contamination in the discharge water.

Monitoring results have demonstrated excellent performance of the in-pit disposal of the tailings, soils and waste rock. Water quality consistently meets Quebec discharge water quality criteria. These successful results are in spite of the final waste rock level being 2 metres above local groundwater levels and a probable hydraulic connection to extensive underground workings.

Both the flooding of tailings and in-pit disposal have been successful at Solbec in reclaiming the site and demonstrating acceptable surface water quality. Results showed that the covering waters met all requirements outlined in Guideline 019 (Quebec Environment and Wildlife Department) as well as those of the Quebec drinkable water regulation. The site has now been covered with water for 15 years and has performed according to design. The former tailings pond has become a valued recreation area; land lots have been purchased around the “lake”, and vacation homes have been built. The mine site had become a popular area for walking all year round. The total project cost for both tailings and waste rock has been approximately $5 million Canadian [4].

LOCATION

The Solbec site is about 225 km southeast of Montreal, Québec; Figure 3-1. It was between Lake Aylmer (3 km away) and Lake St-François. Stratford is the closest village [1] [7].

Figure 3-1 Map of Solbec Site Location and Map of Solbec Open Pit Site [5]

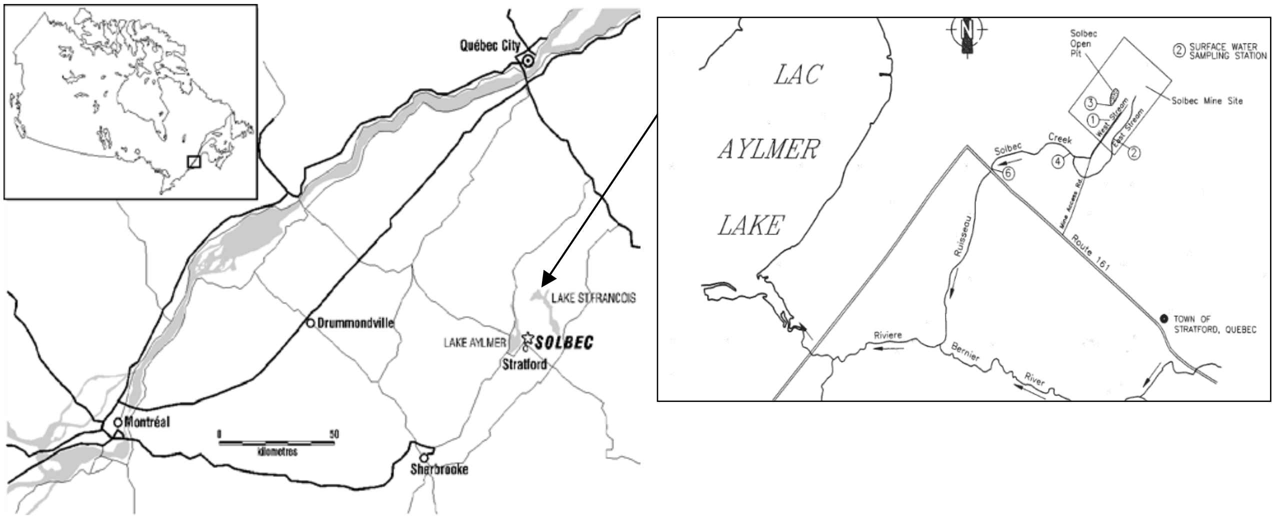
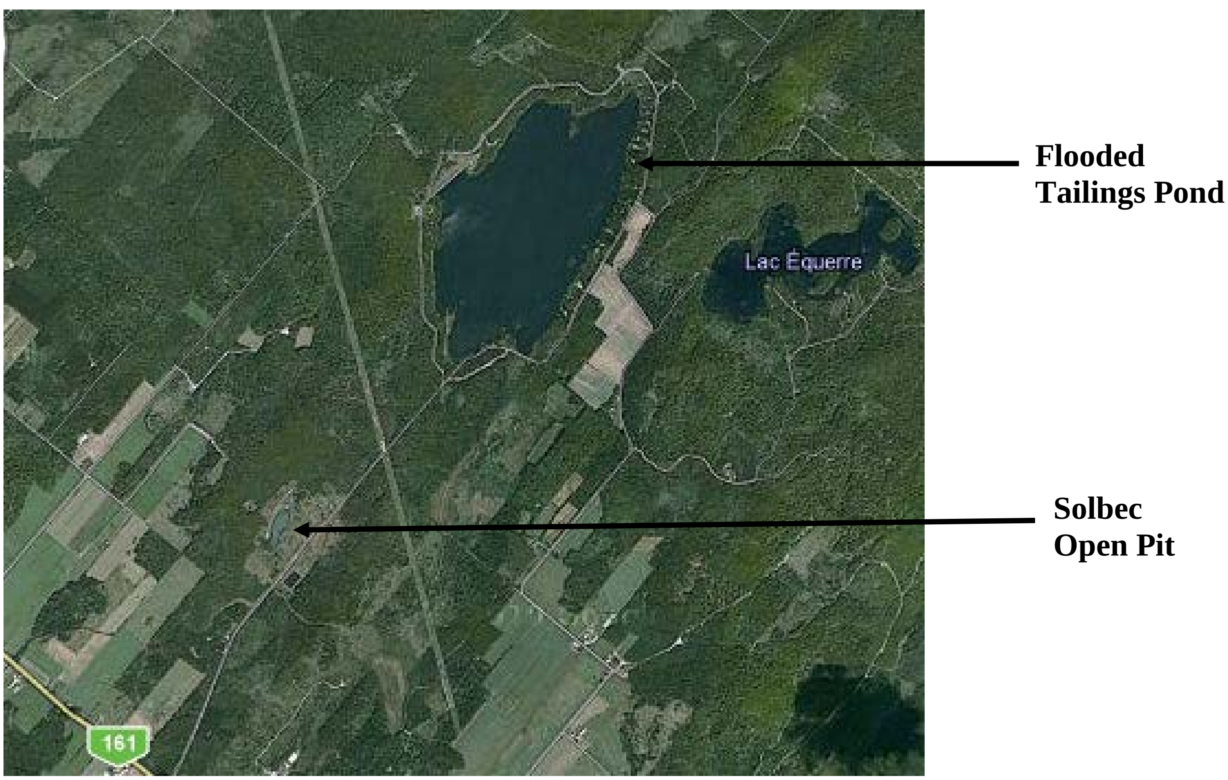


Figure 3-2 Google Earth Satellite Image of the Pit and the reclaimed Tailings Pond (North of the Pit)



MINE TYPE and HISTORY

Sullivan Mines Ltd was purchased and merged with Cambior Inc. in 1987, and as a result Cambior inherited the abandoned Solbec mining site. (In November 2006 IAMGOLD acquired 100% of Cambior shares [1, 2].)

Sullivan Mines Ltd had operated Solbec mine from 1962 to 1970 processing about 1.9 M tonnes of copper, zinc and lead sulfide ores from sericitic schist gangue [1, 2]. Sericite and pyrite-rich mineral complexes are typically strong acid generators due to the expansive nature of the micaceous mineralisation. This was the case at Solbec.

The on-site concentrator operated from 1962 to 1977, processing ore from Solbec and a local mine's ore. A total of 4.8 M tonnes of massive sulphides consisting of pyrite (FeS_2), sphalerite (ZnS), chalcopyrite ($CuFeS_2$), and galena (PbS) [7] were processed [1].

PIT DESCRIPTION

Information from mine section drawings illustrated that the pit had been excavated to a depth of about 40 m, but the depth of the pit to the surface of the tailings and other mine waste at the end of operations was 26 m. Approximately, 10,000 m^3 of tailings had been disposed in the pit during the last few weeks of mill operation in 1977 [7].

The pit surface area was 14,300 m^2 with a total volume (above the tailings) of 365,600 m^3 of which the pit lake occupied 181,700 m^3 in 1988 [7]. The underground workings were at least 10 m below the pit floor. However, some workings extended from the pit floor into the workings providing probable hydraulic connection. These openings may have been filled and covered with tailings before the placement of waste rock. Other shafts and raises extended from the ground surface around the pit to the workings. Although, no evidence of a physical connection to the underground workings was known, a hydraulic connection is likely as evidenced by the volume of water discharging from the flooded pit. Data collected from Station No. 3 Open Pit Overflow showed that there was an average discharge of 1.09 m^3/h during the period from 1989 – 1992 [8].

Ross *et al.* (1994) stated that the pit walls and floor were fractured chloritic schist and sericite. Drill holes in the backfill showed that 6 m of fine material had accumulated in the pit before it was backfilled a small portion (1×10^4) from the last stages of mining and milling.

Figure 3-3 Plan and Section Views of the Solbec Open Pit (1987) [7]

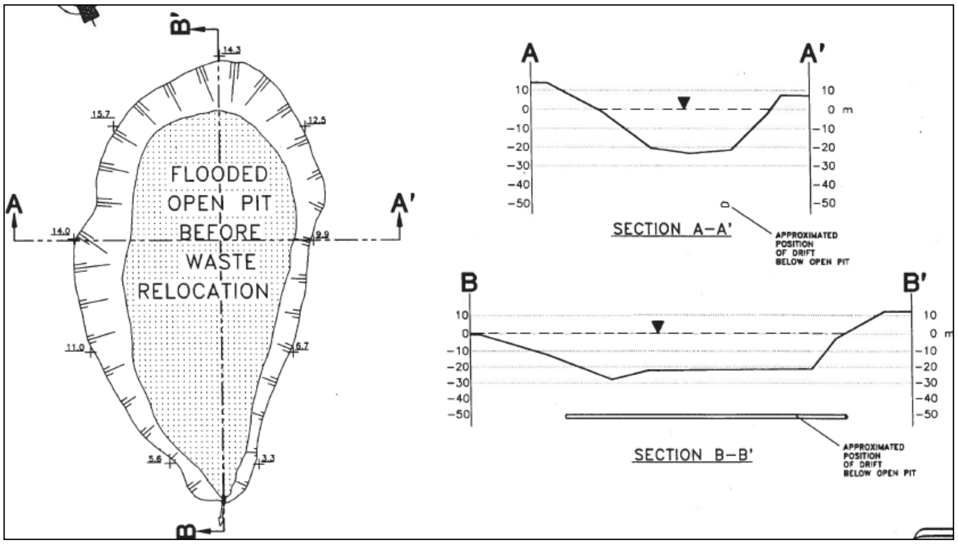


Figure 3-4 Longitudinal Section of the Solbec Pit (1987) [7]

STRATEGIC USE OF PIT

Several options were considered for the use of the small pit at Solbec. A study by (Veillette et Desrochers, 1991) assessed the following disposal options [7]:

OPTION A: No dewatering of the pit prior to waste rock relocation.
OPTION B: Partial dewatering of the pit prior to waste rock relocation.
OPTION C: Complete dewatering of the pit prior to waste rock relocation.

OPTION A - NO DEWATERING

Waste rock would be end dumped into the pit. The pit water pH would be expected to decrease and metal concentrations would increase. Consideration would be required to control and treat pit water [7].

OPTION B - PARTIAL DEWATERING

To reduce the risk of the spread of water-borne contamination, the mine wastes would be placed in a partially dewatered pit. The pit water level would be pumped into a series of settling ponds during in pit waste disposal. About half of the pit water ~100,000 m^3, would be removed at treated [7].

OPTION C - COMPLETE DEWATERING

Complete dewatering might have allowed mobile equipment to enter the pit to compact the wastes. This option would require about 200,000 m^3 of water to be pumped out of the pit and treated [7].

Option B – Partial Dewatering was selected and approved by MENVIQ (le Ministere de L'Environment de Quebec), who authorized the work to proceed in two phases:

- Mine wastes would be placed into the partially dewatered pit and the surface areas would be cleaned and revegetated.
- The pit would be allowed to flood naturally, maintaining a water cover over the wastes [7].

Experimental tests were conducted in 1989**,** 1990, 1991 and 1992/1993. In December 1993, flooding of the pond began after adding the fine limestone [1].

MATERIALS PLACED IN PIT AND METHODS OF PLACEMENT

In August 1988, relocation of the waste rock into the dewatered pit started and the work was completed in 12 weeks. This created a type 2 dump. (The ore is deep or under a significant amount of waste rock which is blasted and excavated. The initial waste rock is placed beside the pit, and as mining progresses the waste rock is backfilled into the pit). A 5 cm layer of minus 5 mm agricultural type calcitic limestone formed a neutralizing barrier. On completion of waste relocation, it was discovered that material was 2 m above the local water table. A low permeability berm was constructed at the south end of the pit to allow a 2 m water cover over the waste rock [1].

A 6,000 m^3 settling pond was constructed downstream of the open pit. A total of 150,000 m^3 of water were pumped out over 6 weeks (beginning in June 1988) lowering the pit level 19 m. The sloped bottom of the pit was then used as a sump/settling pond which was thereafter pumped down on a weekly basis [7].

Ultimately, a total of 276,000 m^3 of wastes (including contaminated soil) were removed from a 12.1 ha area and placed into the pit [7]. No information is available on the waste properties – e.g., acidity, metal leaching etc.

Figure 3-5 Photo of the Pit filling Operation at the South End of the Solbec Open Pit 1998 [8]

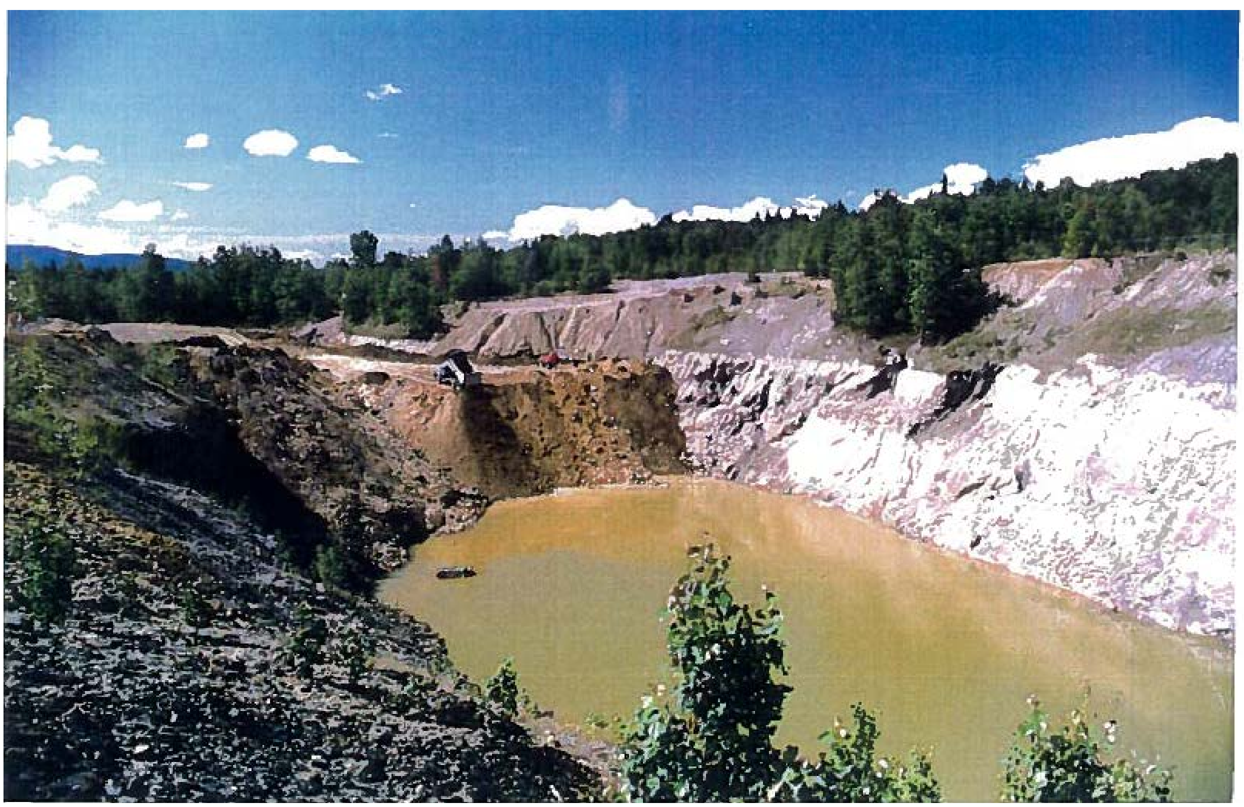
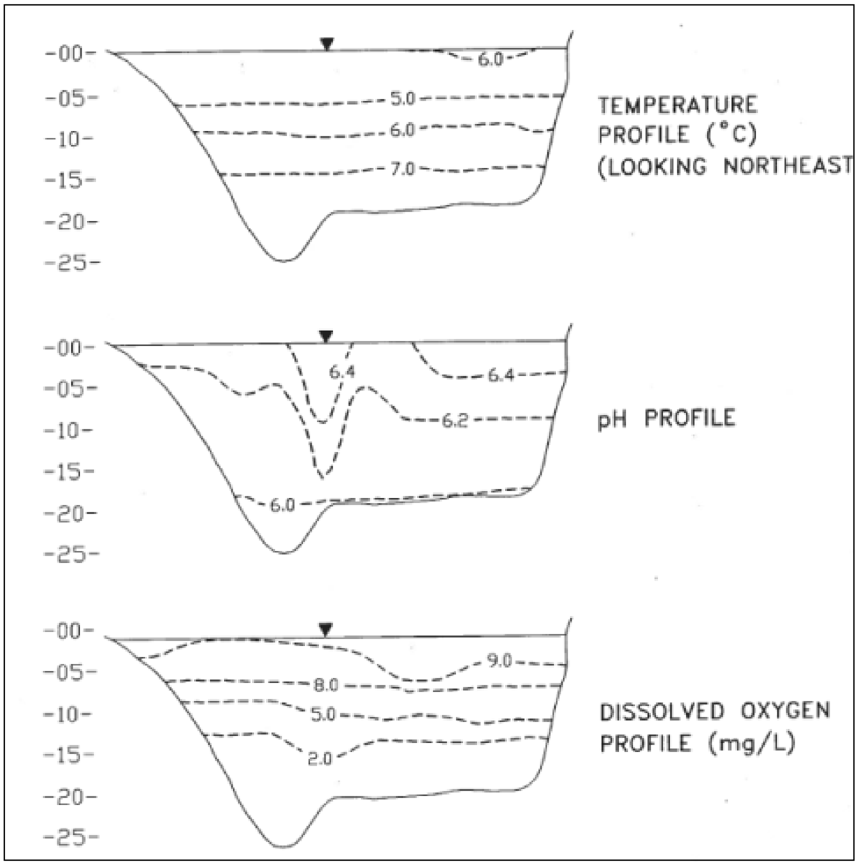


Figure 3-6 Profiles Temperature, pH, and Dissolved Oxygen along Longitudinal Center Section of the Solbec Open Pit (1988) [7]



PERFORMANCE OF IN-PIT DISPOSAL

Ross *et al.* (1994) conducted drill tests and determined that groundwater flowed north to south through the pit backfill. There were also some variables vertical flowpaths. Based on hydraulic conductivities of 1 x 10^{-6} and 1 x 10^{-7} m/s in the surrounding fractured bedrock, which limited flow, a maximum porosity of 0.01, and a hydraulic gradient of 0.03, the average linear groundwater velocity was estimated at 10-95 m/yr. total volume of ground water flow was estimated at 1.2 x 10^4 m^3/yr.

Porewater in the submerged waste rock contained more than 1 mg/L dissolved oxygen; therefore, oxidation of sulfide minerals may be a source of groundwater contamination.

Figure 3-7 Flooded Solbec Open Pit (1993) [7]

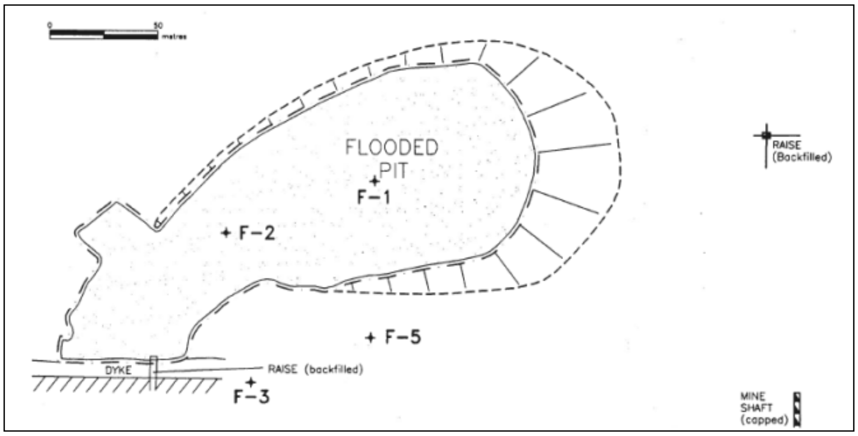


Figure 3-8 Hydraulic Gradient and Groundwater Flow (1993) [7]

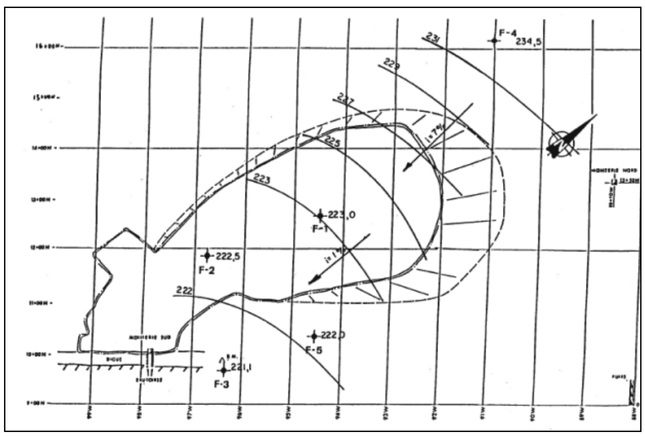


Figure 3-9 Groundwater Flows Within and Around the Solbec Pit (1994) [7]

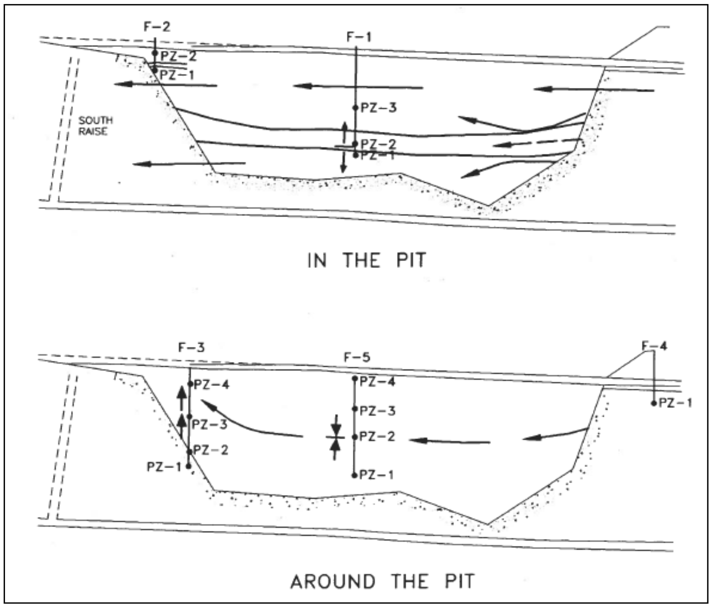
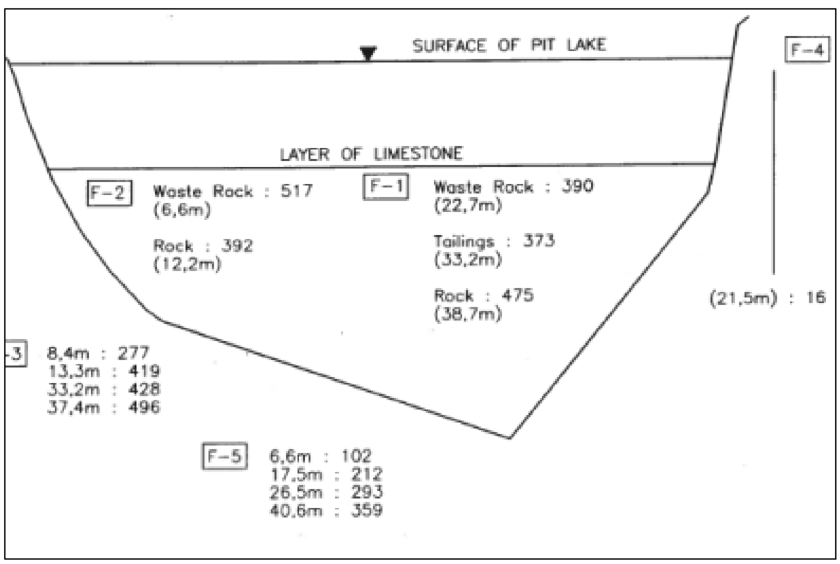


Figure 3-10 Schematic Section of Solbec Open Pit Showing Average Calcium Concentration (mg/L) in Groundwater (1993) [7]



In 1987, a water quality monitoring program was established. Table 3-1 shows the measured parameters (except for zinc and iron) met MENVIQ (le Ministere de L'Environment de Quebec) effluent quality limits [7].

In the fall of 1994 as part of the MEND program an environmental monitoring program for the tailings pond was and is still ongoing [1].

Table 3-1 Solbec Pit Water Quality [a] fall of 1987 [7]

Parameter	Range of Measurements			Measurements According to Depth				Effluent Quality Limits
	Minimum	Maximum	Average	At Surface	5 m	10 m	20 m	
pH	5.7	6.6	6.3	6.31	6.31	6.25	6.08	6.5 to 9.5
Cu (mg/L)	0.05	1.91	0.18	0.21	0.21	0.13	0.18	0.30
Pb (mg/L)	0.00	0.50	0.12	0.12	0.09	0.07	0.05	0.20
Zn (mg/L)	0.64	1.89	1.02	1.02	0.98	1.10	0.96	0.50
Fe (mg/L)	0.32	41.9	3.50	0.69	1.23	2.34	18.53	3.00
Mn (mg/L)	2.3	8.20	4.05	3.42	3.46	4.49	6.18	-
Cd (mg/L)	0.0	0.20	0.04	0.03	0.05	0.05	0.03	-
Temperature (°C)	5.0	7.0	5.7	5.4	5.0	5.7	6.9 [b]	-
Conductivity (µmhos/cm)	1,600	2,050	1,755	1,750	1,694	1,790	1,890	-
Alkalinity, as $CaCO_3$ (mg/L)	134	196	145	143	143	149	170	-
Dissolved Oxygen (mg/L)	1.1	9.8	6.1	9.2	8.2	5.2	1.7 [b]	-
Total Dissolved Solids (mg/L)	1,154	2,819	1,660	1,581	1,588	1,685	2,053	-

(a) Source: Tables 2.2.1 and 2.2.2 Desrochers (1990).
(b) At 14 m depth.

Table 3-2 Open Pit Sediment Analyses [a] fall of 1987 [7]

Parameter	Range of Measurements		
	Minimum	Maximum	Average
Cu (mg/kg)	82	4,087	1,110
Pb (mg/kg)	31	1,304	426
Zn (mg/kg)	165	1,069	318
Mn (mg/kg)	434	2,887	761
Cd (mg/kg)	15	37	20
As (mg/kg)	2.4	3.8	7.4
SO_4 [b]	537	2,154	838

(a) Source: Table 2.2.3, Desrochers (1990).
(b) Likely in form of gypsum $CaSO_4 \cdot 2H_2O$.

Table 3-3 Selected Groundwater Quality (Data Obtained From Piezometer Sampling At Solbec Pit from February to October 1992) [7]

Parameter	Borehole F-4 (Located Upgradient)		Borehole F-1 (Located Within Pit)				Borehole F-5 (Located Downgradient from Pit)			
	Piezometer PZ-1		Piezometer PZ-1		Piezometer PZ-3		Piezometer PZ-1		Piezometer PZ-3	
	Feb 1992	Oct 1992	Feb 1992	Oct 1992	Feb 1992	Oct 1992	Feb 1992	Oct 1992	Feb 1992	Oct 1992
Temperature (°C)	1.3	5.1	5.40	7.80	6.40	7.10	3.0	6.7	2.7	6.9
Conductivity (µmhos/cm)	238	257	3,800	3,710	6,710	6,760	3,590	3,730	2,800	1,620
Redox Potential (mv)	46	-132	-35	-12	-7	1	102.0	47.0	120.0	71.0
pH	7.05	7.70	6.21	6.32	5.88	5.78	5.89	5.9	6.2	6.39
Dissolved Oxygen (mg/L)	9.70	1.20	2.9	3.7	2.0	2.5	5.8	2.9	6.2	4.8
Sulphate (mg/L)	22.7	24.0	2110	2,350	4,330	5,350	3,290	2,200	1,500	725
Zinc (mg/L)	0.12	0.14	18	16	34	24	51.0	49.0	42.0	34.0
Iron (mg/L)	0.09	2.10	36	41	820	1,450	11.00	120.00	0.25	2.60
Copper (mg/L)	<0.03	<0.03	0.15	0.04	0.05	0.06	0.17	0.37	0.46	0.43
Cadmium (mg/L)	<0.02	<0.02	0.02	<0.02	0.02	0.03	0.20	0.28	0.20	0.14
Lead (mg/L)	<0.05	0.07	0.14	0.22	0.22	0.41	0.12	0.14	0.09	0.13
Arsenic (mg/)	<0.20	0.80	0.73	0.30	4.90	6.00	<0.20	<0.20	<0.20	0.60
Calcium (mg/L)	17.0	13.5	600	332	495	330	385	350	285	295

Monitoring of water discharging from the pit has continued. The most recent data (2009) is shown in Table 3-4. Zinc remains the only parameter of any potential concern, but is effectively reduced to well below discharge limits by the installed biofilter.

Table 3-4 2009 Solbec Monitoring Data (Monitoring Data, IAMGOLD, 2010)

Parameter	Effluent Basin 6000	Effluent Basin 11000	Effluent Filters A and B	Effluent Filters C and D	Effluent Final
pH	8	8.3	8.4	8.2	7.7
Copper (mg/L)	<0,01	0.02	0.01	<0,01	0.03
Iron (mg/L)	0.33	0.05	0.09	0.05	0.3
Nickel (mg/L)	<0,02	<0,02	<0,02	<0,02	<0,02
Lead (mg/L)	<0,03	<0,03	<0,03	<0,03	<0,03
Zinc (mg/L)	0.27	0.12	0.06	0.07	0.16
MES (mg/L)	< 4	< 4	< 4	< 4	< 4

REMEDIAL

With the exception of maintenance of the biofilters, no remedial measures have been needed at Solbec.

BENEFITS and LESSONS OF IN-PIT DISPOSAL

In 1993, Dr. Serge Vezina stated that the flooded oxidized waste rock submerged below shallow cover of water was still under evaluation. *"Zinc concentration level in the pit effluent is decreasing but still over effluent limit. Filtration through a multi-media passive filter* (at the outlet of the settlement pond) *is currently being evaluated on a full scale basis and is showing positive results. The use of such a filter is contemplated as a mean to reduce zinc concentration under the effluent limit until such a level is naturally obtained."* [7]

The use of the pit for disposal at Solbec has been beneficial in removing problematic acid generating and metal leaching wastes and contaminated soils from surface and permanently storing them below the pit water table.

Minor amounts of mobile zinc have been leached from the oxidised material submerged in the pit. The levels of zinc in the pit overflow have continued to decline over 20+ years.

RESIDUAL ISSUES

No residual issues have been reported. In-pit disposal with surface water quality polishing has been successful at Solbec.

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CASE STUDY #4 EQUITY SILVER MINE

SYNOPSIS

The Equity Silver Mine had three open pits (Waterline, Main Zone and South Tail) in close proximity of one another and small underground workings (North Zone) that were accessed from the Waterline pit. Equity was British Columbia's largest ever silver-producing mine producing silver, gold and copper from 1980 until 1994. Placer Dome Inc. operated the facility, which is now owned by Goldcorp Canada Ltd. The three open pits produced a total of 34 Mt of ore with an average grade of 183 g/tonne silver equivalent (silver+copper+gold value expressed as silver content). Waste rock production exceeded 80 Mt.

Acid rock drainage (ARD) originating from the waste rock was observed about 2 years after mine operations began and rapidly became a significant issue with its management remaining an ongoing requirement more than 20 years after mining and ore processing has ended. The three pits at Equity perform valuable functions in the management of this ARD.

The Southern Tail pit was the first ore body to be mined; it was subsequently backfilled with waste rock from the Main Zone pit, the second and largest of the three pits. Pit lakes have developed in the Waterline and Main Zone pits. The smaller Waterline pit lake drains into the Main Zone pit lake which also receives treated water and sludge from an HDS (High Density Sludge) treatment plant. In addition to receiving treatment sludge from the HDS plant, sludge from previous treatment operations has been transported to the Main Zone pit. Before closure approximately 0.32 Mt of potentially acid generating (PAG) waste rock were placed in the bottom of the Main Zone pit as a source of alkalinity. No evidence has been observed of contamination of the pit lake by this waste rock. The Main Zone pit provides surge capacity and water quality polishing before discharge on a controlled basis to local receiving waters usually during the spring freshet.

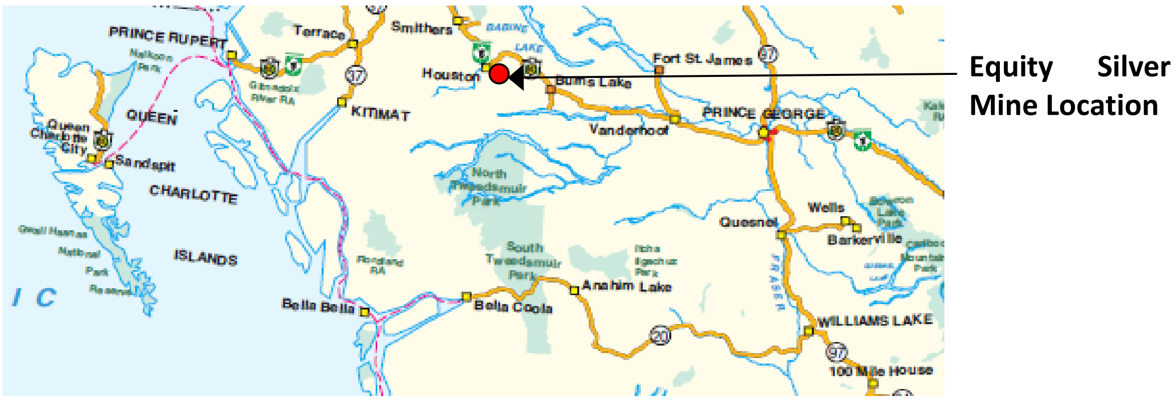
All aspects of acid rock drainage at Equity have been extensively investigated and a wide range of measures have been implemented and maintained to minimize the extent and potential impact of ARD on the surrounding environment. The Equity pits are integral to site ARD management.

LOCATION

The Equity Silver Mine is located on the Nechako Plateau at an elevation of 1300 meters above sea level approximately 35 km southeast of the town of Houston, British Columbia, Canada, and 575 km north-northwest of Vancouver [4]. The annual average precipitation in the area is 670 mm including 250 mm as snowfall, although records of 450 mm (0.45 metres of snow) have been observed [23]. The area is typically frost-free for less than 60 days, with snow usually present from October to April. The spring freshet accounts for 80% of the mine site runoff. This high freshet volume greatly influences ARD management at Equity.

The Equity Silver Mine is at the head of two watersheds that feed the Bulkley River, and is adjacent (to the east) to a segment of Bessemer Creek [7] [2]. Figure 4-1 shows the location of the mine.

Figure 4-1 Location of Equity Silver Mine near Houston, British Columbia [6]



MINE TYPE AND HISTORY

Open pit mining at Equity Silver Mines began in April 1980 and ended in early 1994 [4]. The mine operation consisted of three open pits (Southern Tail, Main Zone, and Waterline), and small underground workings accessed from the bottom of the Waterline pit. The mill processed up to 9,000 tonnes of ore per day and produced silver, gold and copper [3] [5].

Placer Dome Inc. was the major shareholder of Equity Silver Mines Ltd., operating the mine from 1980 to 1994. Placer Dome became the 100% shareholder in 1996 [4] [16]. Since 2006, Equity Silver Mine has been owned by Goldcorp Canada Ltd. [5].

The three open pits at Equity produced a total of 34 Mt of ore with an average grade of 183 g/tonne silver equivalent [4]. Production totaled 72.3 M oz of silver, 498,000 oz of gold and 189.6 M lbs of copper, from ores with an average grade of 2,544 g/tonne copper, 61 g/tonne silver and 0.42 g/tonne gold [9][17]. Antimony and arsenic were temporarily recovered for sale as by-products during operations [17]. In addition, the underground workings produced 450,000 tonnes of ore with an average grade of 363 g/tonne silver equivalent [4].

Equity Silver operations produced 35 million tonnes of tailings and 85 Mt of waste rock [4]. Of this, 7.6 Mt of waste rock was a non-acid generating gabbro/monzonite (mostly used for road and tailings dam construction) and 77.4 million tonnes (or ~91%) was acid generating or potentially acid generating (PAG) pyroclastic volcanic rock [4]. Some of the acid generating rock had been used in the initial tailings dam construction and plant site backfill, before it was determined to be acid-generating.

The waste rock was piled in three locations: initially in the Main and Bessemer dumps, and later in the Southern Tail pit (Figure 4-2(a)). In 1981, some acid drainage was observed to be originating from one waste pile, and in 1982 ARD was confirmed to be a significant issue for much of the waste rock. Initial remedial actions included the installation of a partial containment system to reduce the acid drainage into Bessemer Creek [7]. Tailings were also determined to be PAG and were deposited under a water cover to prevent oxidation and acid generation - a flooded condition that remains in place 20 years after operations ended as shown in Figure 4-2 (b).

Figure 4-2(a) Aerial View of Equity Silver Mine during Operation [9]

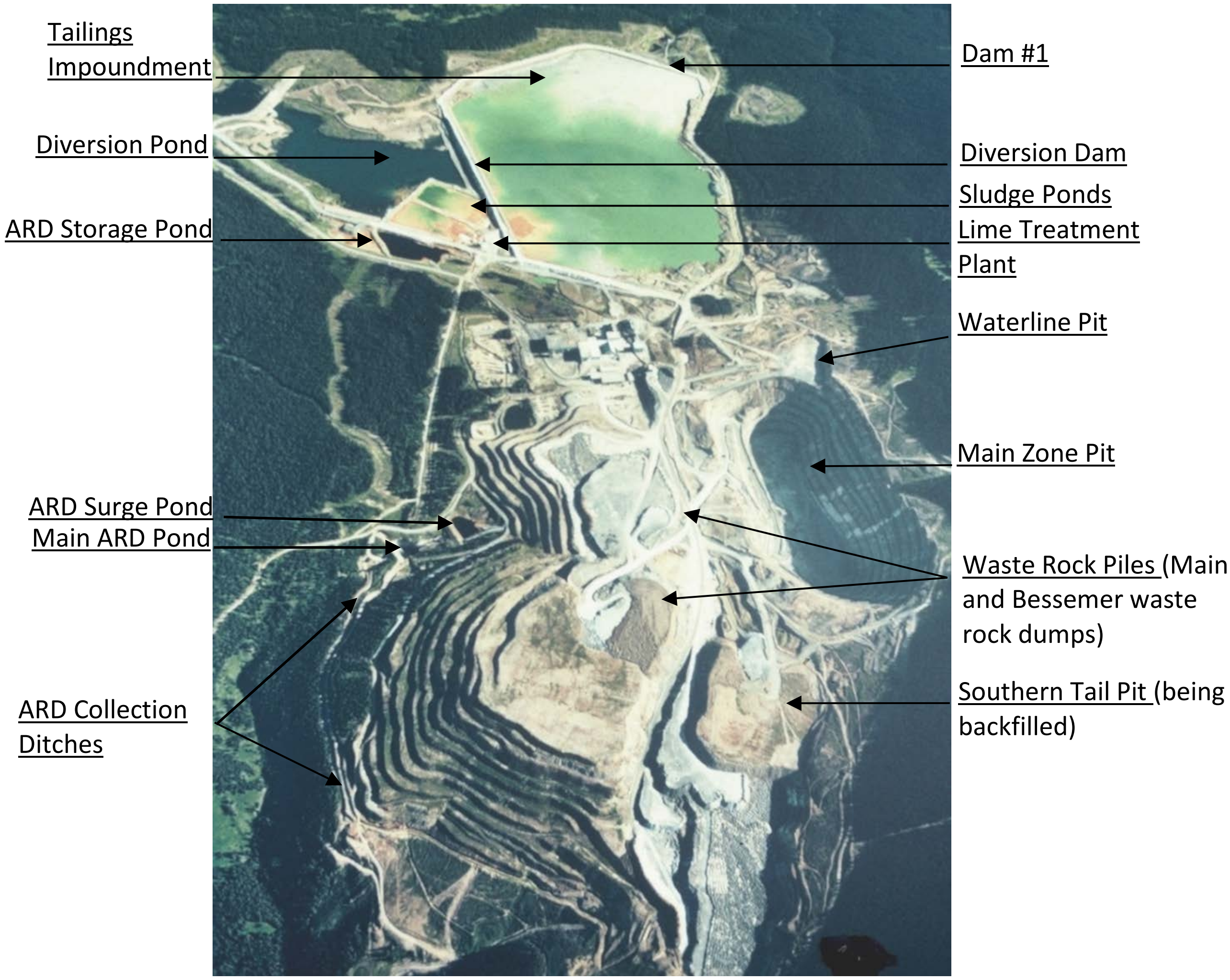


Figure 4-2(b) Recent Aerial View of Equity Silver Mine

The Southern Tail pit contained the highest grade ore (and possibly the most reactive waste rock) and was the first pit to be mined. The Southern pit had the highest sulphide content, lowest alkalinity, and the waste rock was highly modified, making it susceptible to oxidation [9]. Mining of the Southern Tail pit was completed in 1984 followed by partially backfilling the Southern Tail pit with 5 Mt of PAG waste rock from the Main Zone pit [3]. The backfilled waste was then covered with non acid-generating waste rock and the pit was used as Equity's third waste rock dump for the remainder of mining operations.

In early 1983, the mining of the Main Zone pit began, generating substantial quantities of waste rock [23]. The pit was actively mined and dewatered until 1991, after which the pit was allowed to flood with the inflow of groundwater, precipitation, surface runoff and treated water. The resulting pit lake level was allowed to rise to an elevation of 1260 m in July 2000, and has since been maintained at this level by pumping water to the local watershed [23]. The approximate depth of the Main Zone pit lake is 120 m.

In April 1988 a conceptual closure plan was prepared for decommissioning the Equity facility in 1992. A final closure plan was submitted in April 1991 [20]. Progressive measures of reclaiming the waste dumps had already been made in the late 1980's when the flat sections of the terraced waste dumps were covered with uncompacted till. Construction of the waste dump covers started in 1990, initiated with the Southern Tail waste dump [4]. Remediation entailed resloping, placement and covering the waste dumps with 0.5 m of compacted silty clay till topped with 0.3 m of loose silty clay to serve as a vegetation growth medium [3]. The till provided a water and oxygen barrier to reduce ARD production. All three waste dump covers were completed by 1994. The dismantling and capping of the plant site was completed between 1994 and 1997.

From 1985 to 2003, the waste rock produced an average annual ARD volume of 852,000 m^3 [9], or about the volume of 340 Olympic swimming pools each year. Acid drainage continues to be collected and treated to date (2014). The system is composed of a series of ditches and ponds collecting runoff from the covered waste rock dumps, reclaimed plant site, and seepage from the Tailings Pond Dam #1. This drainage collection is pumped to the HDS (high density sludge) plant for treatment (Figure 4-3). A pond termed the Main ARD Pond, receives the majority of the mine site ARD with contributions from the Main waste dump, Bessemer waste dump, and waste rock above the water table in the Southern Tail pit [18][23]. The drainage water chemistry has been relatively stable as seen from water analyses records between 2006 and 2011 for pH (~2.5), Fe (~1200 mg/L), Al (~1000 mg/L), Zn (~800 mg/L), and Cu (~100 mg/L) [18]. As per Figure 4-4, flow and acidity have remained relatively constant for the last 20 years.

Figure 4-3 ARD Collection and Treatment System at Equity Silver Mines [18]

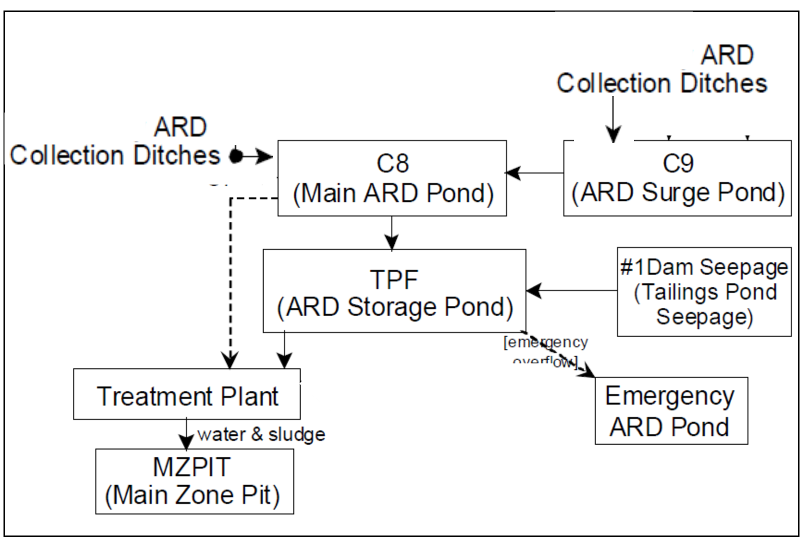
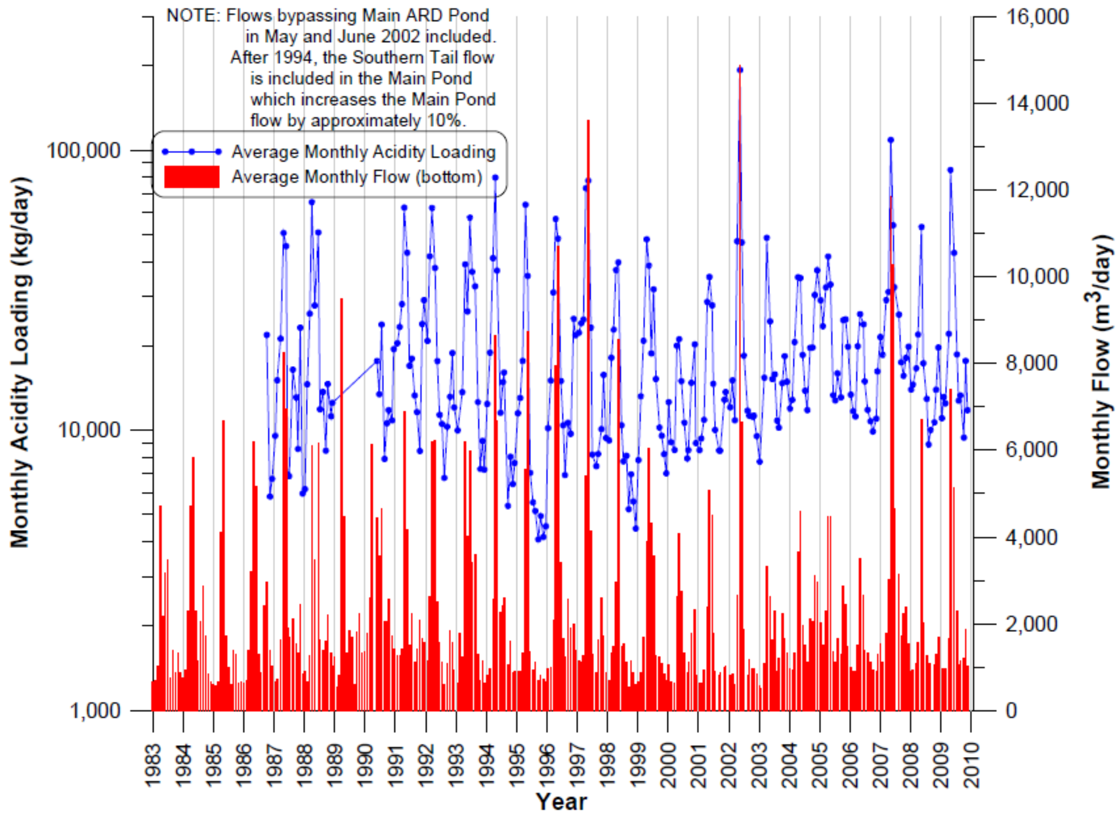


Figure 4-4 Monthly Acidity Loading and ARD Flow from the Main ARD Pond Since Equity Silver Mine Closure [18] (Note: Monthly acidity loading on a logarithmic scale)



The Main Zone pit is a critical component in the management of ARD at Equity. Pumping of treatment sludge to the Main Zone pit commenced in 1993, and continues to be pumped to the pit along with treated water [23]. The Main Zone pit also receives drainage from the adjacent Waterline pit during ice-free months and from the tailings pond. Water is normally pumped from the Main Zone pit to the local watershed during the spring freshet period, [9].

ARD collection and treatment system upgrades were completed between 2002 and 2008. In 2007, the Equity site and surrounding area received a record snow pack and resulting melt, but with treatment system upgrades the mine site was able to handle this event reasonably well. To provide additional assurance, an emergency ARD storage pond was constructed in 2008, which increased ARD storage capacity to 850,000 m^3 [5].

The Tailings impoundment contains submerged PAG tailings, and the pond has a surface area of 120 ha [21]. The tailings pond water is maintained by dams on the north (Dam #1), west (Diversion dam) and on the southern (Dam #2) perimeters of the pond [21]. The depth of the water cover ranges from 1.5 to 8.0 m with higher depths to the north away from the mill. Lime was periodically added to the pond to control pH up to 2007, but has not been required since then

[21] [9]. The tailings pond water pH has been maintained at about 7; copper levels were less than 0.004 mg/L in 2013 and zinc averaged 0.024 mg/l, and showed no measurable level of acidity [21]. Excess Tailings Pond water is decanted to the Diversion Pond or pumped directly to the Main Zone pit [9].

The Equity Mine tailings may have some potential to be re-processed for gold, silver and copper recovery [9] in the future with the right technology.

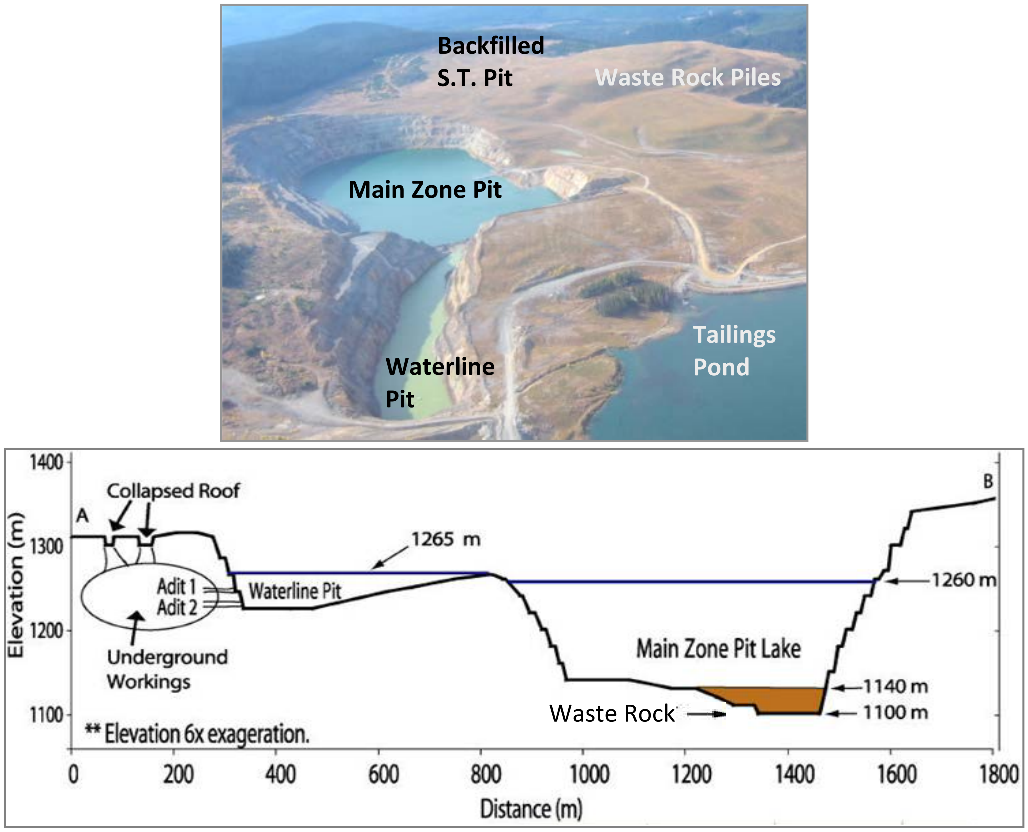
MINERALIZATION AND PITS DESCRIPTION

The silver, gold and copper deposits occurred in tabular zones in homoclinal rock within tertiary pyroclastic volcanics [16]. The principal ore minerals were tetrahedrite and chalcopyrite [16]. Pyrite and pyrrhotite were present along with minor amounts of galena, sphalerite, argentite, minor pyrargyrite and other silver sulphosalts [17]. Pyrite was the most abundant sulphide in all the mineralized zones, averaging between 2-5% by volume in the waste rock [3].

The Southern Tail pit was partially backfilled with waste rock and later used as a location for a waste rock dump that extended above the original pit elevations. The other two pits currently contain pit lakes. The Main Zone pit is 850 m by 500 m across, and 350 m deep (from highest wall which has an elevation of 1350 m) [10][23]. To control water discharge and maintain emergency water storage capacity, water is pumped from the Main Zone pit using an intake 20 m deep in the pit lake, and is discharged into the Bessemer Creek or Foxy Creek drainage systems [23].

The Waterline pit is approximately 150 m by 500 m long (at the top), with an average depth of 18 m and a maximum depth of 43 m [23]. The water level is naturally maintained by an overflow located at an elevation of 1265 m into the Main Zone pit (Figure 4-5).

Figure 4-5 (Top) Looking South at the Waterline and Main Pit Lakes [9]; (Bottom) Profile of the Two Pit Lakes [23]



STRATEGIC USE OF PITS AND MATERIALS PLACED IN-PIT

In 1984 after mining of the Southern Tail pit ceased, the pit was partially backfilled with 5 Mt of PAG waste rock from the Main Zone pit, to one metre below the final accumulated water elevation at that time [3]. At the air water interface of the filled pit, a 2 m layer of non-acid generating gabbro was placed on top of the backfilled waste rock [4]. The Southern Tail pit and footprint was later used as a waste rock dump, which by the end of operations consisted of 12 Mt of PAG rock from the Main Zone pit. The Southern Tail pit zone holds approximately 17 Mt of waste rock. During Equity site remediation (1990-97), the Southern Tail waste rock dump was resloped, and covered with 0.5 m of compacted, and 0.3 m of uncompacted clay till to support a vegetation buffer, as illustrated in Figure 4-6 [3].

Figure 4-6 Profiling and Covering Sequence of the Southern Tail Pit Waste Rock Dump [9]

The Waterline pit lake inflow includes ARD-affected water from the adjacent underground developments, and some contaminated runoff from the pit walls, in addition to net precipitation [23].

The Main Zone pit was backfilled with 0.321 Mt of PAG waste rock from the Waterline pit and underground developments, which filled the bottom 25 m of the pit (Figure 4-5)[23]. The flooded Main Zone pit currently receives HDS treatment plant discharge water as well as approximately 130,000 m^3 per year of ARD sludge. Sludge reclaimed from previous treatment operations has been excavated, slurried and deposited in the pit. The pit walls contain some PAG rock and produce minor amounts of ARD which drain into the pit. The adjacent Waterline pit lake overflows to the Main Zone pit during ice-free months, at a rate up to 20 L/s [23]. On average, 340,000 m^3/yr of tailings water is pumped to the main zone pit.

The ARD sludge from the HDS treatment plant is pumped from the plant and co-discharged into the Main Zone pit with treated water, as shown in Figure 4-7. While this co-discharge may seem to be counterintuitive, this method has proven to assist in moving the sludge into deeper zones of the Main Zone pit lake. The pit lake provides surge capacity needed to accommodate water discharge during the freshet period and for meeting water quality objectives in the receiver.

Figure 4-7 Treated Water and Sludge Discharge to Main Zone pit from HDS [9]

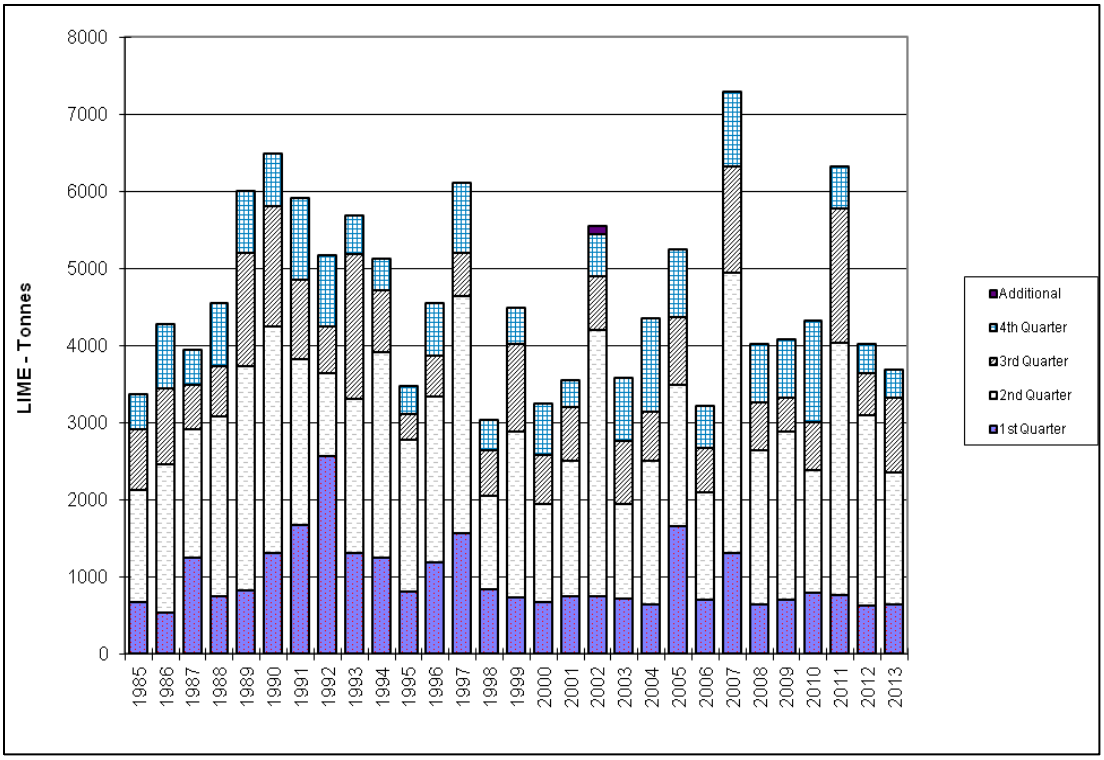
ARD Treatment and Sludge Management

Acid rock drainage emerging as surface seeps from the waste rock dumps flows into collection ditches surrounding the perimeter of the waste rock dumps, and subsequently flows into collection ponds and is pumped into a storage pond near the treatment plant (Figure 4-8) [3]. The waste rock dumps contributes 65% of the mine ARD volume but 94% of the site acidity; seepage from tailings pond Dam #1, accounts for 20% of the volume and 4% acidity, and the former plant site contributes 15% of the volume and 2% of the acidity [9]. Treatment is achieved by adding lime slurry in the HDS treatment plant to a pH of 8.5 and then pumping the sludge and treated water to the Main Zone pit.

Figure 4-8 Movement of ARD from the Equity Silver Mine Site Through the ARD Collection and Treatment System to Disposal (Photographs [9])

The HDS treatment plant produces 90,000-120,000 m^3 of sludge annually, depending upon the volume and acidity of the ARD being treated. Between 2000 and 2009 the average annual volume of ARD collected and treated at Equity was approximately 880,000 m^3. The lime required to treat the ARD averages around 4,000 tonnes per year as shown in Figure 4-9 [22].

Figure 4-9 Annual Lime Consumption at Equity Silver Mine [9]



Treated water achieves dissolved metal concentrations at levels below permit regulation limits. Copper levels in the treated water are typically less than 0.001 mg/L, zinc levels less than 0.02 mg/L, and sulphate levels between 3000 mg/L and 4000 mg/L [9].

The sludge deposited in the Main Zone pit typically has a pH between 8 and 8.5, contains 5-8% solids of largely gypsum and metal-oxyhydroxides [23].

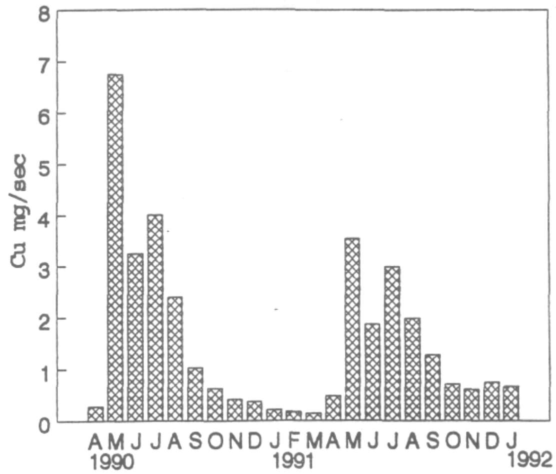
PERFORMANCE OF IN-PIT DISPOSAL

Southern Tail Pit

Early pH data of the Southern Tail pit between 1985 and 1990 (during the pit use as a waste dump), show that the pH increased from 3 to 8.5, then lowered to 7 by 1990 [10]. It is possible that anoxic conditions were rapidly established in the filled pit as freshly broken rock had some neutralizing potential and prevented the initiation of ARD.

A summary of water quality data during the first years of the Southern Tail pit waste dump remediation, showed decreasing copper levels as seen in Figure 4-10 below.

Figure 4-10 Copper Discharge from the Southern Tail Pit during Early Stages of Waste Rock Dump Remediation [20]



Temperature and pore gas data from within the Southern Tail pit, collected between 1992 and 2010, were studied by Morin *et al.* [18], and are depicted in Figures 4-11 and 4-12.

Figure 4-11 Temperature Within the Southern Tail Pit at Various Depths From the Surface of the Waste Rock Dump Cover (Note: Southern Tail data excludes P-6) [18]

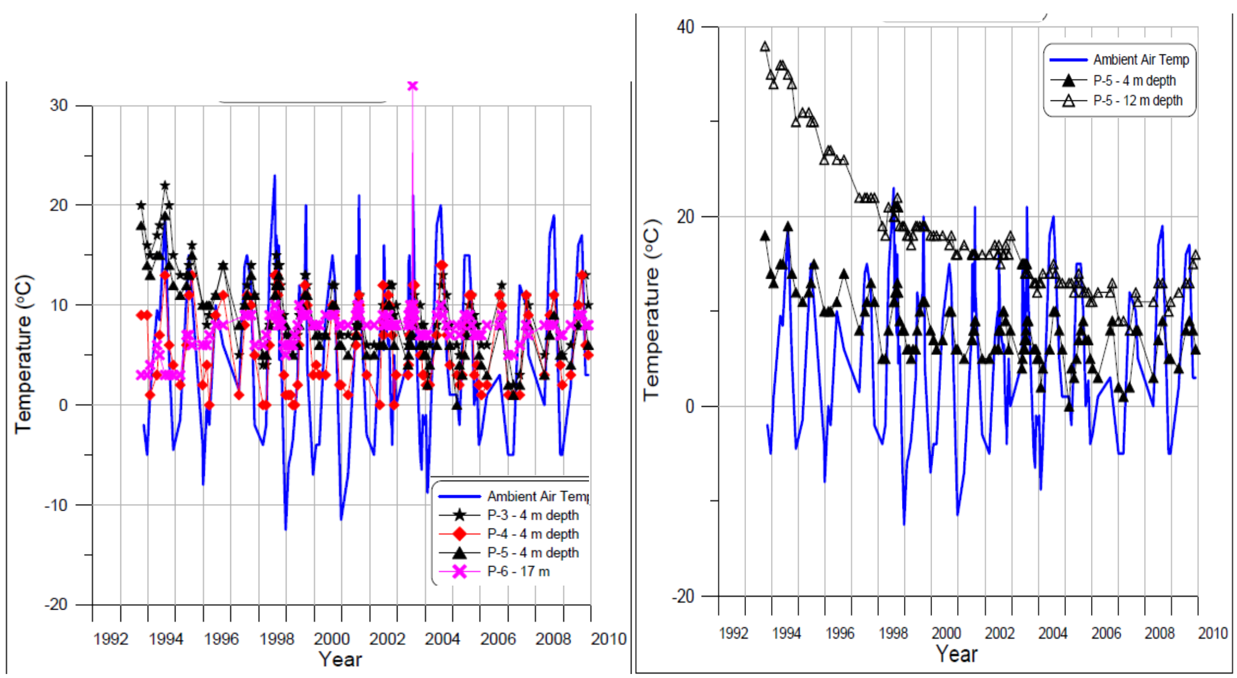
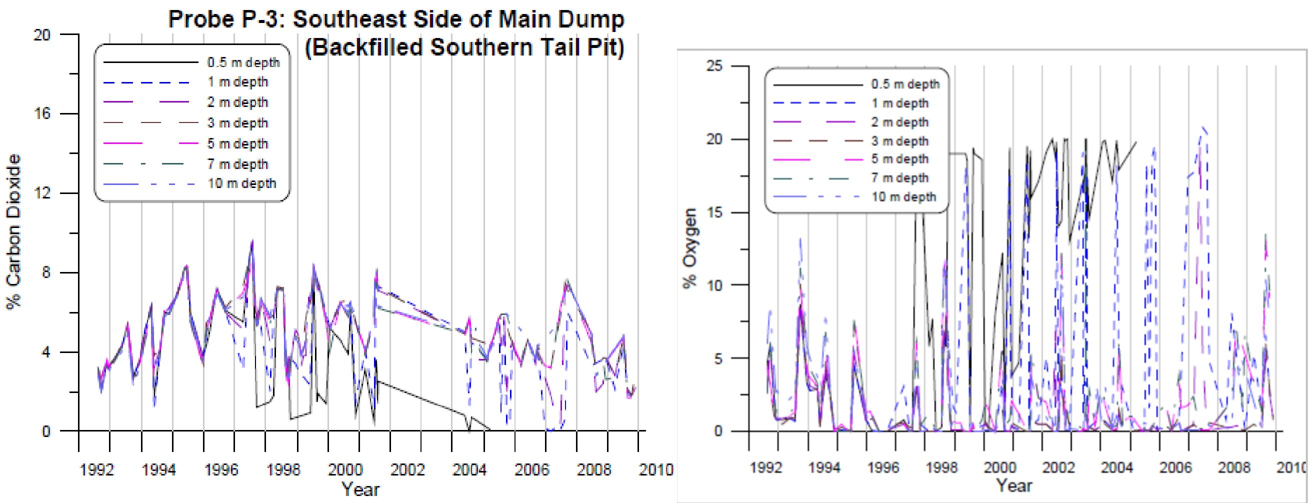


Figure 4-12 Oxygen and Carbon Dioxide Levels within the Southern Tail Pit at Various Depths from the Surface of the Waste Rock Dump Cover [18]

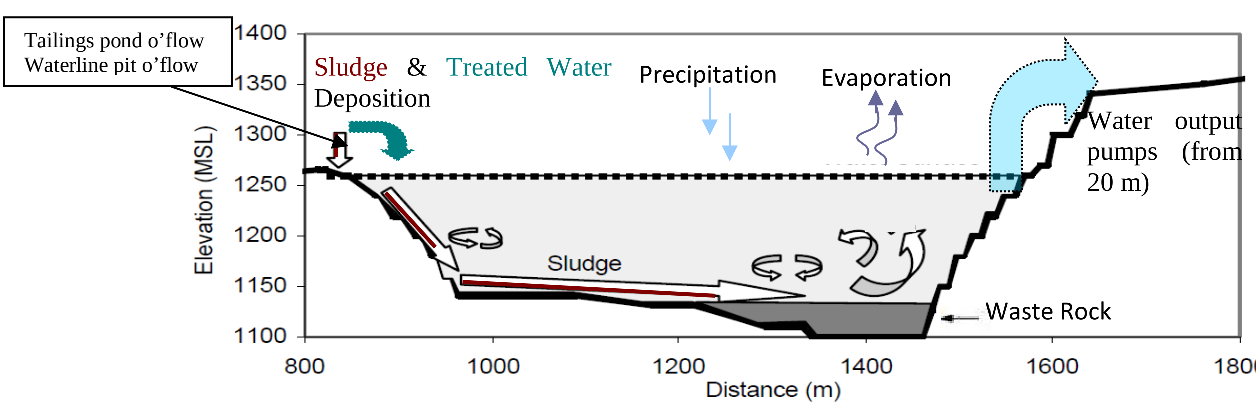


Typically, elevated temperatures above ambient temperature as well as oxygen deficiency within a waste rock pile suggest pyrite oxidation and acid generation. Temperature and pore-gas data did not show a correlating trend within the Southern Tail pit or in the other two acid generating Equity waste dumps evaluated by Morin *et al*. Because of this lack of correlation with reactivity of the waste rock, the data is instead thought to be indicative of a larger-scale air movement within the dump. Based on the long term temperature trends Morin found that the waste rock dumps were generally cooling indicating a slowing rate of pyrite oxidation and acid generation [18]. This trend could be a depiction of the waste rock moving from a stage of active oxidation to a stage of passive flushing of accumulated acidity and metals.

Main Zone Pit

The Main Zone Pit hydrology is illustrated in Figure 4-13 below.

Figure 4-13 Cross Section of the Main Zone Pit [23]



The periodic input of treatment sludge and treated water into the Main Zone pit triggers unique lake mixing observed year-round (Figure 4-13) [12]. Pit lake mixing is indicated by the relatively uniform seasonal temperature, high saturation levels of dissolved oxygen in the water column, and fairly uniform conductivity, as seen in Figures 4-14 to 4-16. The dissolved oxygen content of the upper 10 m of the lake is near saturation levels, while the lower depths varied seasonally, ranging from 68% in early spring to 100% in summer and fall [23].

Figure 4-14 (Left) Seasonal Temperature Trends in the Main Zone Pit Lake [23] and (Right) in the Upper Ten Metres of the Lake [12]

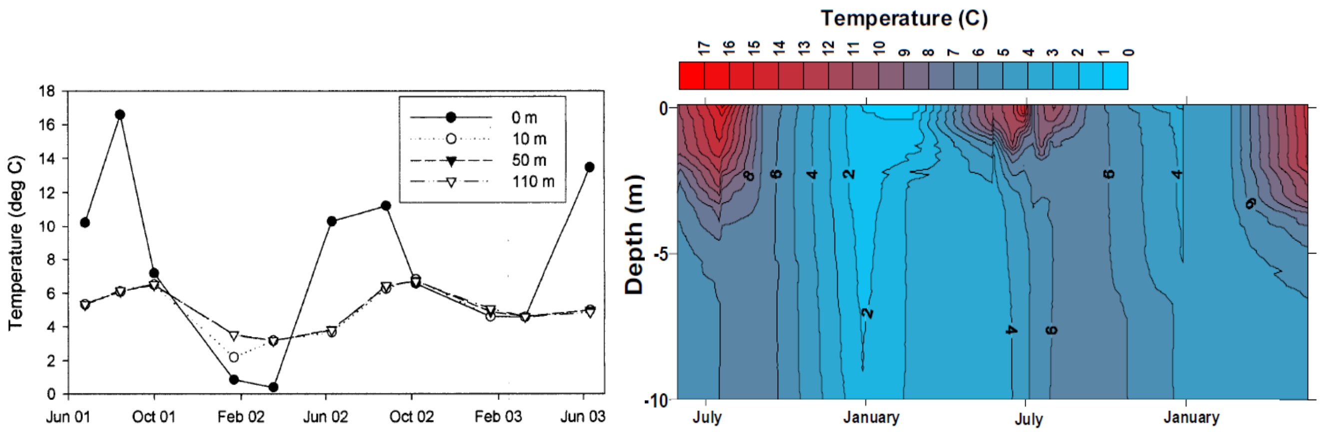


Figure 4-15 Seasonal Dissolved Oxygen Content in the Main Zone Pit [12]

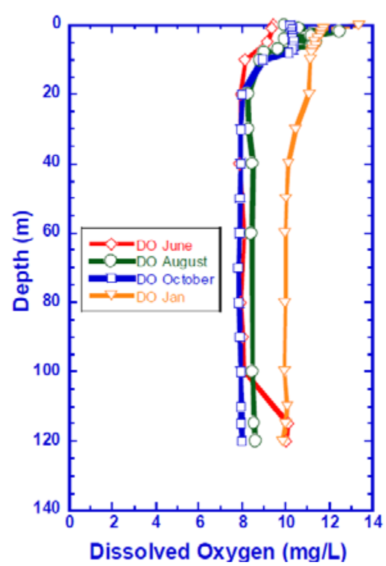
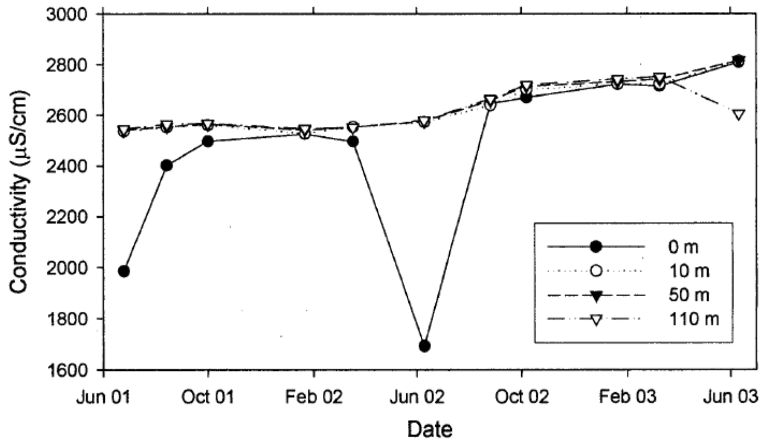


Figure 4-16 Main Zone Pit Lake Conductivity [23]



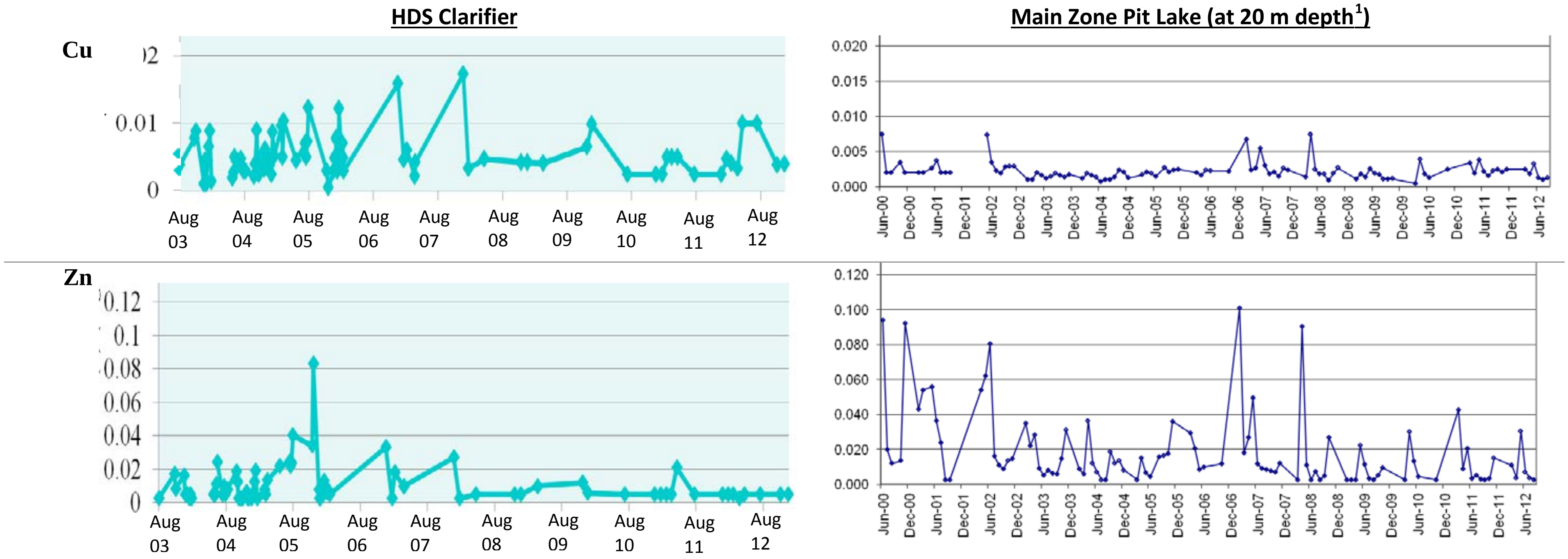
The major anion in the Main Zone pit lake is sulphate, with concentrations of 1,730 mg/L to 2,200 mg/L occurring at the surface to a depth of 40 m [23]. High sulphate levels are due to gypsum and other metal sulphate solubility in the treated water. Moderate phosphate levels have also been noted in the lake; however, these levels are limited by precipitation with Fe-oxyhydroxides [23].

Metal concentrations monitored in the Main Zone pit comprise a full range of metals including Mn, Fe, Cu and Zn, as well as As. Arsenic levels have been very low - at limits of detection based upon 2001-2003 data reported by Whittle [23]. Metal concentrations have been relatively constant in recent years with some temporal peaks in spring and summer surface waters occurring for particulate Fe (0.3 mg/L), and dissolved Zn (~1 mg/L) and Cu (~40 ppb)[23]. The slight metal enrichments appear to be linked with the spring freshet period - which comprises surface ARD seeps, pit wall runoff and input of Waterline pit waters - and not a result of sludge deposition [23]. Figure 4-17 shows a comparison of HDS treated water quality deposited into the Main Zone pit and the concurrent Main Zone pit water quality. Copper and sulphate concentrations in the pit lake have been lower than in treated water, while zinc is slightly higher.

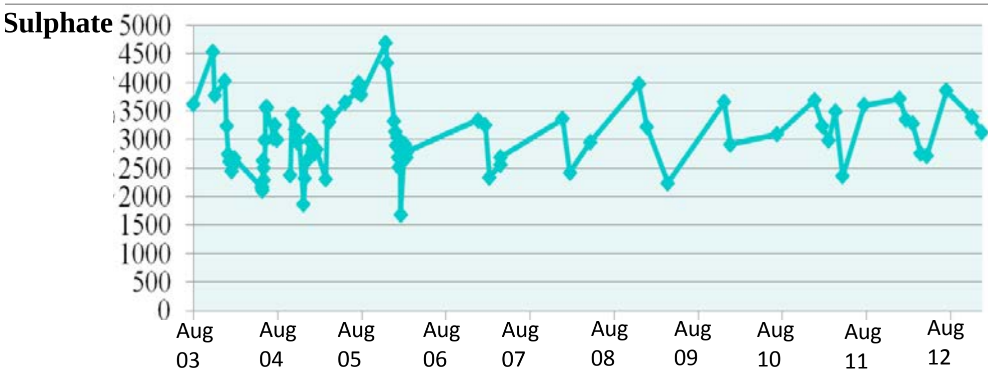
Manganese concentrations on the other hand, have some correlation with sludge deposits as treatment sludge water contains levels around 1.4 mg/L of Mn. In the pit lake, recorded Mn peaks are around 2 mg/L in the summer and fall [23]. Winter records show a reduction in Mn concentrations. Dissolved Fe was near detection limits throughout the water column. Whittle [23] suggested iron may be removed by oxidation or adsorption into Fe-oxyhydroxides.

Metal concentrations collected at depth by Whittle in 2001-2003, for total Fe (~0.1 mg/L or less), and dissolved Zn (~25 ppb), Ni (~10 ppb), Co (<10 ppb), Cu (<5 ppb), and Cd (<14 ppb) showed no temporal activity or trends with depth, although Mn seemed to show slight seasonal variation [23]. The metals water quality data in Table 4-1 shows that the pit lake has generally experienced a further decrease in metals concentrations since 2003.

Figure 4-17 Comparison of Dissolved Cu, Zn, and Sulphate (mg/L) in the Incoming HDS Treated Water and the Main Zone Pit Lake [9]



[1] Note - 20 m depth in the Main Zone pit lake is a standardized depth holding stable parameter concentrations.



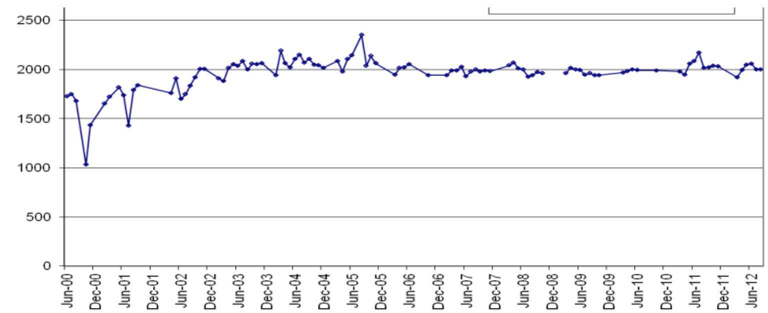


Table 4-1 Main Zone Pit Water Quality @ 20 m Depth

DATE	FIELD pH pH unit	TOTAL ALKALINITY mg/L	DISSOLVED SULPHATE mg/L	SPECIFIC CONDUCT µS/cm	DISSOLVED ALUMINIUM mg/L	DISSOLVED ANTIMONY mg/L	DISSOLVED ARSENIC mg/L	DISSOLVED CADMIUM mg/L	DISSOLVED COPPER mg/L	DISSOLVED IRON mg/L	DISSOLVED ZINC mg/L
2000 avg	7.63	40.3	1497	2556	0.04	<0.2	<0.2	0.00180	0.0032	0.050	0.0766
2001 avg	7.26	43.1	1720	2560	0.09	<0.2	<0.2	0.00251	0.0028	0.130	0.0366
2002 avg	8.12	36.0	1880	1293	0.26	<0.2	<0.2	0.00305	0.0028	0.060	0.0220
2003 avg	7.68	26.5	2049	2841	0.22	<0.2	<0.2	0.00172	0.0016	<0.030	0.0128
2004 avg	7.73	25.6	2062	2070	0.15	<0.20	0.0003	0.00145	0.0012	0.036	0.0130
2005 avg	7.59	46.9	2093	2480	0.21	<0.20	<0.05	0.00138	0.0021	<0.030	0.0182
2006 avg	7.26	32.0	2020	2942	0.08	<0.20	<0.05	0.00179	0.0022	0.034	0.0131
2007 avg	7.20	26.2	1984	2962	0.12	0.10	0.0275	0.00161	0.0029	0.021	0.0189
2008 avg	6.80	35.4	1994	2934	0.10	0.10	0.0380	0.00109	0.0022	0.015	0.0107
2009 avg	7.48	26.6	1978	2961	0.11	0.10	0.0240	0.00098	0.0019	0.016	0.0106
2010 avg	7.31	25.9	1993	2985	0.10	0.10	0.0250	0.00107	0.0021	0.016	0.0130
2011 avg	7.62	23.2	2052	2961	0.12	0.10	0.0010	0.00103	0.0025	0.015	0.0086
2012 avg	7.42	21.4	2019	2994	0.09	0.10	0.0017	0.00138	0.0018	0.015	0.0111
2013 avg	7.65	23.0	1996	3036	0.12	0.08	0.0002	0.00098	0.0011	0.014	0.0062

Apart from Co and Ni present in the pit lake sediment at levels exceeding B.C. Contaminated Sites Regulation standards, the mixing and excess alkalinity from sludge deposition aid in maintaining the stability of sludge deposited in the pit. Possibly aided by pit water oxygenation waters, most are reporting to oxyhydroxide precipitates or co-precipitates. Data indicated that the Main Zone Pit Lake is an efficient sink for metals and is beneficial in keeping the low metal concentrations observed in the Main Zone pit water column [23].

Waterline Pit Lake

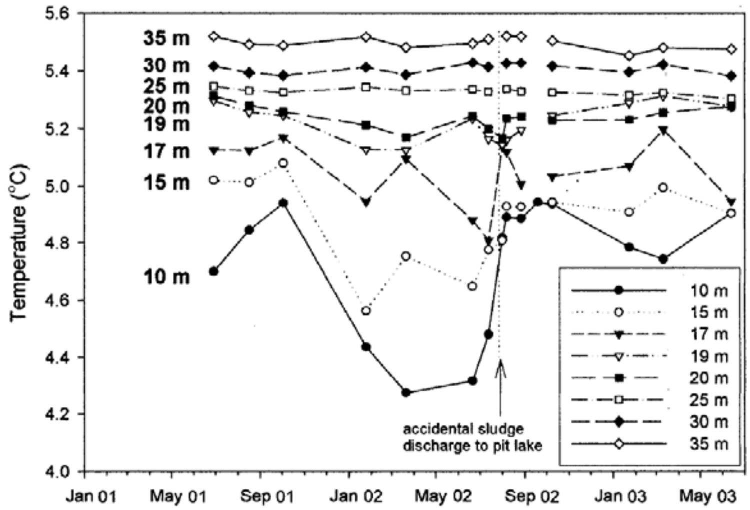
The Waterline pit lake is a stratified, meromictic lake, resulting from a density difference caused by variation in calcium, sodium and magnesium sulphate dissolution [23]. The largest density difference in the lake is between the epilimnion and the hypolimnion. This density layer fluctuates seasonally as noted in Table 4-2.

Table 4-2 Solute Stratified Layers in the Waterline Pit Lake [23]

Epilimnion	**Depth from Surface (m)**		**Conditions, *Notes***
	Summer	Winter	
Layer 1	0-4.5	0-8.5	Dissolved oxygen-close to saturation; well mixed
Hypolimnion			
Layer 2	4.5-17	8.5-17	Sub-oxic to Anoxic; uniform conductivity
Layer 3	17-20.5		Appears to be affected by adit sourced water Conductivity varies seasonally
Adit 1 (15-20 m)			
Layer 4	20.5-28		Anoxic (assumption); linear increase of conductivity throughout, with seasonal variation
Layer 5	28-30.5		Anoxic
Layer 6	30.5-45		Anoxic - increase of conductivity at the top of layer; Highest dissolved Fe and As
Adit 2 (33-37 m)			

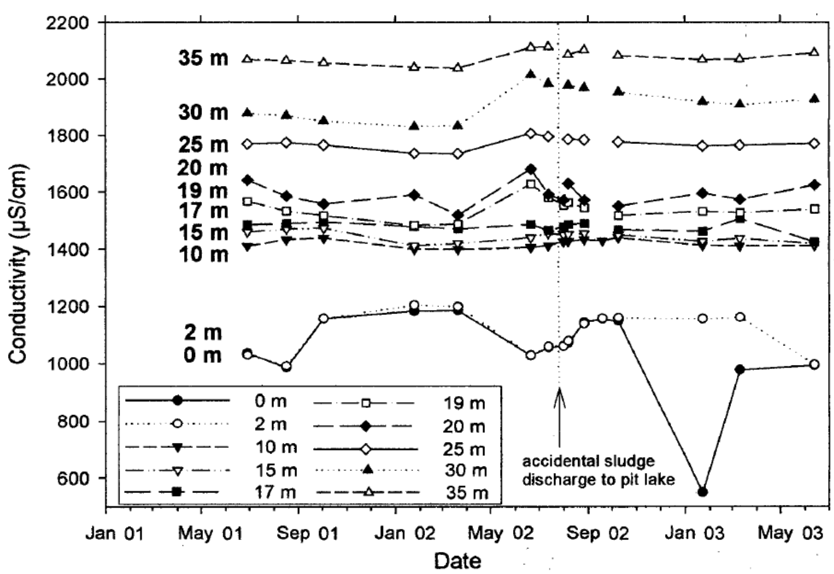
The temperature of the Waterline pit lake waters, in 2001-2003 data collected by Whittle, showed significant seasonal temperature variation detected up to a ~20 m depth (Figure 4-18) [23]. From 20 m to 35 m below surface, the temperature was noted to increase very slightly from 5.2 ^{o}C to 5.3 ^{o}C [23]. This unexpected observance could be attributed to the adjacent underground mine workings which are connected to the pit through two adits, one at a depth of 15-20 m and the second at 33-37 m [23]. Regional precipitation can enter the pit through the adits, adding some salinity and conductivity during the downward and lateral movement of the water. This adit-sourced water appears to be a significant source of iron and arsenic found in the lake depths [23].

Figure 4-18 Annual Temperature Cycle in the Waterline pit (Note: 0 and 2 m data not shown for simplicity; Accidental sludge discharge on 28 July 2002) [23]



The Waterline pit lake waters in the hypolimnion contain 600-1500 mg/L of sulphate [23]. A sulphate-adding source has been determined to be ARD oxidation in the underground mine workings [23]. The lake pH is neutral at 7.5. This neutrality is believed to be a combination of iron-oxyhydroxide or sulphate reduction reactions and aluminosilicate alkalinity [23]. Figure 4-19 shows pit lake annual conductivity trends.

Figure 4-19 Waterline Pit Lake Conductivity, January 2001 to January 2003 (Note: Accidental ARD treatment sludge discharge on 28 July 2002 shown) [23]



From data collected between 2001 and 2003 by Whittle [23], arsenic, iron and zinc were present at elevated concentrations in the hypolimnion, while manganese was slightly elevated throughout the water column. Many metals have been at very low concentrations in the oxygenated epilimnion, particularly in the spring and summer months with a slight increase during the autumn epilimnetic mixing, such as: U (0.6-0.8 ppb respectively), As (~0-1.3 ppb), Mo (1.3-3 ppb), Fe (~1-60 ppb), Co (12-18 ppb), Ni (30-38 ppb), Zn (~400 ppb), and Mn (1300-2000 ppb) [23]. Other metal concentrations were measured to be highest in Layer 1 - Cd (~6 ppb), Cu (1-10 ppb), and Sb (1.7-4.3 ppb) [23].

In the hypolimnion, some metals decreased in concentration with depth, including: Cu (~0 ppb at bottom), Cd (~0 ppb at bottom), Sb (0.2-0.8 ppb), Co (35-28 ppb), Ni (60-40 ppb), and Zn (600-800 ppb), while Mo (3-8 ppb) and As (800-2000 ppb) increased with depth in the hypolimnion [23]. Uranium (1.8-2.3 ppb), Mn (~3800 ppb), and Fe (3000-4000 ppb) demonstrated fairly consistent concentrations year-round in the lower layers. Barium was the only metal that held consistent levels (10-16 ppb) throughout the water column during the 3-year study [23].

A most important aspect of the Waterline Pit lake performance is the quality of water discharged to the Main Zone Pit. With the possible exception of zinc, water quality has been very good as shown in Table 4-3.

Table 4-3 Waterline Pit Discharge Water Quality 2002 – 2013

DATE	pH	Acidity mg/l	As (t) mg/l	Cd(t) mg/l	Cu(t) mg/l	Fe(t) mg/l	Zn(t) mg/l
2002 avg	7.64	<1	<0.2	<0.01	0.07	0.59	0.39
2003 avg	7.47	<1	<0.2	<0.01	0.04	0.92	0.46
2004 avg	7.66	0.5	<0.2	<0.01	0.03	0.71	0.42
2005 avg	7.55	3.2	<0.2	<0.01	0.05	0.68	0.42
2006 avg	7.42	3.3	<0.2	<0.01	<0.01	0.05	0.44
2007 avg	7.46	8.0	<0.2	<0.01	0.05	1.61	0.86
2008 avg	7.56	<1	<0.2	<0.01	0.09	1.21	0.69
2009 avg	7.37	5.6	<0.2	<0.01	0.11	2.66	0.71
2010 avg	7.26	10.5	<0.2	<0.01	0.25	10.25	0.68
2011 avg	7.91	7.7	<0.2	<0.01	<0.01	0.36	0.74
2012 avg	7.66	16.0	<0.2	<0.01	0.019	0.05	0.72
2013 avg	7.47	5.5	0.085	<0.01	0.016	1.21	0.65

The anoxic hypolimnion in the Waterline pit lake is an effective sink for dissolved metals precipitating as sulphides [12] as the anoxic conditions in the hypolimnion result in metal reduction and precipitation. The stratified condition of the lake also provides effective containment of metals in the lower levels of the water column.

REMEDIAL/TREATMENT

Options for mitigation methods have been explored for reducing metal and sulphate concentrations in both Equity Silver pit lakes. Specifically, limnocorral experiments have been conducted in the Waterline pit lake and have resulted in successful development of a pit lake physical-chemical modelling software (PitMod®). This model simulates sulphate reduction and the sequestration of Zn and other metals, as a result of algal productivity (by adding fertilizer) and ethanol deposits in the pit lake [13]. This would result in the collection of metals encompassed on the sediment floor. A whole-lake manipulation of the Waterline pit has been considered.

LESSONS OF IN-PIT DISPOSAL AT EQUITY SILVER MINE

The Equity Silver Mine has had ongoing collection and treatment of ARD since it was first detected in 1982, two years into mine production. Prior to initiation of mining, ABA analyses were completed on only three drill core samples of ore; waste rock was not tested. The consequences and process of mine ARD was not well understood at the time. Cold weather was assumed to inhibit acidification of sulphide-containing rock. Although cold conditions can slow the initiation of oxidation, cool climates often assist in the pumping of oxygen through waste rock dumps as result of a “chimney effect”.

Based on economic considerations, the Equity Silver Mine’s Southern Tail pit was mined first. This pit contained the most reactive rock of the site. The pit would later be filled with less reactive waste rock from the Main Zone pit. A reversal of order in mining of materials (least reactive rock first) could have reduced the sulphide content of the waste rock dumps and ARD production at Equity [9].

Thorough initial investigations of mine site rock may have allowed for appropriate preparations for ARD mitigation during the life of Equity Silver Mine.

“What If” - Potential In-pit Disposal Options

Two of the three Equity Silver Mine open pits have hosted a pit lake after mine operations ended. Theoretically, these pits had the potential to host and isolate much of the reactive waste rock.

Assuming an in-situ rock density of 2.9 t/m^3, the 114 million tonnes of mined ore and waste rock would have occupied a volume of 39.3 Mm^3. The total Equity Silver Mine PAG waste rock is identified to be 77.4 Mt, as 7.6 Mt of the 85 Mt of mine waste rock, was non acid-generating. The 77.4 Mt of broken waste rock has an estimated density of 1.7 t/m^3 and would occupy a volume of 42.6 Mm^3. This waste rock volume is slightly larger than the volume of mined hard rock, and because the main pit has a substantial high wall, some PAG waste rock would not be submerged in water. However, in addition to the volume discrepancy, significant additional practical factors may have been considered in choosing not to backfill the pits, or at least the Main Zone Pit, including:

- Inability to place significant quantities of unoxidized waste in the largest pit;
- Cost – higher costs (Net Present Value) in neutralizing the oxidized waste rock, placing and covering the wastes in-pit, collecting and treating other ARD sources;
- Impracticality of backfilling underground openings and filling pits to levels below water overflow levels;
- The need to treat pit water, hydraulically isolating the wastes in the pit (by pumping);
- Covering the waste rock that would not fit into the pits, and the subsequent collection and treatment of ARD; and
- Absence of a permanent disposal location for treatment sludges.

THE FUTURE AT EQUITY

Ongoing monitoring, collection and treatment, and site maintenance will be required at Equity for many years due to the large volume of PAG waste rock and the associated high acidity ARD that is produced.

The continuing need to collect and treat ARD at Equity has been the subject of many investigations and studies during the mine operations and over the 20 years since mine operations ended. Many valuable lessons have been learned including the beneficial role possible for backfilled pits and pit lakes.

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CASE STUDY #5 LOS FRAILES MINE

SYNOPSIS

The Los Frailes Mine in Spain was a copper, lead, zinc and silver open pit mine facility. On April 25, 1998, a tailings dam failed when the soil foundation beneath the dam shifted, releasing five to seven million tonnes of high sulphide, acid generating tailings slurry into the Rio Agrio valley, the location of a small intermittent stream. The spill covered thousands of hectares of farmland and greatly affected the ecology of the Donana National Park. Clean up action, which included collecting all of the tailings and contaminated soils, was collectively taken by the mine owner (Boliden AB) and Spanish governments. The mine was re-opened on April 6, 1999, but was declared insolvent in 2000, and it closed on September 20, 2001.

The Los Frailes operation included two pits: the Aznalcóllar Pit and the Los Frailes Pit. The Aznalcóllar pit was mined out in 1996 and subsequently used to dispose of scrap and waste rock from the operating Los Frailes pit. After the tailings dam failure in 1998, the Aznalcóllar pit was also used for disposal of the recovered spilled tailings and contaminated soil. In addition, oxidized, acidic waste rock was introduced to the pit lake in 2005 and 2006 from an adjacent waste rock pile, reducing the water quality of the pit lake.

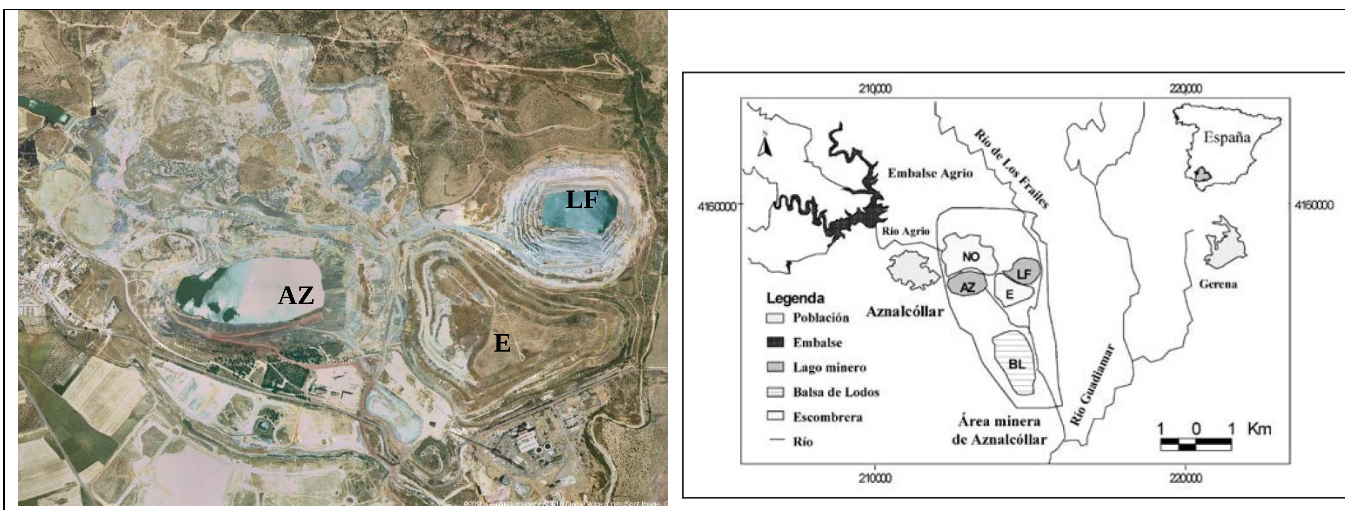
The pit disposal at Los Frailes has been successful in isolating a variety of metal leaching and acid generating wastes, although this long term isolation requires periodic removal and treatment of pit lake waters to maintain this isolation.

LOCATION

The mine site is located adjacent the village Aznalcóllar, 45 km west of Seville in the southern Spanish province of Andalucía (latitude 37^{o} 23'N, longitude 6^{o} 30'W) as shown in Figure 5-1 [5]. The mine site is drained by the Agrio and subsequently the Guadiamar Rivers. The water system extends to the south-southwest and into Doñana National Park [10]. The annual average precipitation of the Aznalcóllar area is 600 mm, with 70% of the rainfall concentrated from November to March. Evaporation varies from 250 mm/day during summer months to almost zero during winter months [18]. The layout of the Los Frailes Mine, including the pits, is shown in Figure 5-2.

Figure 5-1 Location of Los Frailes Mine in Andalucía, Spain (indicated by red star) [7]

Figure 5-2 Closed-out Los Frailes mine, AZ: Aznalcóllar pit lake; LF: Los Frailes pit lake; TI: tailings impoundment; NO: north-western waste rock pile; E: eastern waste rock pile; A:(Google 2010); B:[17]



A B

MINE TYPE and HISTORY

The Los Frailes mine had been the location of metal mining by the Romans and previously to that, by the Phoenicians. The Romans had constructed an aquaduct originating just behind the mine to reach the extensive settlements near the present city of Seville.[21]

The Aznalcóllar ore body was re-discovered in 1956 and brought into production in 1979 by Andaluza de Piritas, S.A. (Apirsa), owned by Banco Central S.A. The mine operated until 1986 and was reopened in 1987 following Boliden's purchase of Apirsa. A second ore body, named Los Frailes with more than 70 x 10^6 t of ore, was discovered in 1987 [10], 1 km east of the Aznalcóllar Pit [9]. When production ceased at Aznalcóllar Pit in 1996 after depletion of its reserves, the pit remained dormant and was used as a disposal location for scrap and waste rock.

The Los Frailes mine facility had a design capacity of 125,000 t/y of zinc, 48,000 t/y of lead, 4,700 t/y of copper and 90.8 t/y of silver from four Mt/y of ore. The mining and processing was interrupted in April 1998 because of a tailings dam failure at 3:30 am on April 25 which released 4 Mm^3 of tailings water and 2 Mm^3 of tailings into the Rio Agrio-Rio Guadiamar-Rio Guadalquivir river systems, contaminating it with tailings solids and heavy metals [18]. The accident resulted from a lateral movement of 60 m along a bedding plane in the Margas Azules (marine Blue Clay) formation that affected a section of the dam more than 600 m long [12]. The wave of tailings was up to 5 meters high, and fortunately there was no loss of human or animal life downstream. Approximately, 2,600 ha of land was covered with tailings along a 40 km stretch of river channels including farmland, and threatened the Doñana National Park, a UN World Heritage Area [6].

Immediately following the spill, the company and the government began recovery of the spilled tailings and contaminated soils. It was decided very early in the reclamation program that the clean-up would be split between the company, which would clean the upper (northern) half, and the government, which assumed responsibility for the lower (southern) half. It was decided that the tailings and contaminated soils would be placed in the Aznalcóllar pit.

The closed out facility is currently managed by an environmental management Division of Junta de Andalucía – the Regional Government [15].

[21] Ironically, the Romans had provided aqueduct foundation stability for the marine clays in the area which led to the tailings dam failure 2000 years later.

PIT DESCRIPTION

The two Los Frailes mine site pits are of considerable size: the Aznalcóllar Pit measuring 1,200 m × 600 m × 275 m deep; and Los Frailes Pit measuring 800 m × 600 m × 265 m deep (Figure 5-3). The mine site is located at the eastern end of the Iberian Pyrite Belt, a 230 km-long east-west mineral belt, well-known for its massive sulphide deposits. The deposits are in rhyolite-hosted pyroclastics as either mineralised pyroclastics or pyritised zones [9].

Before waste disposal in the Aznalcóllar pit, a pit lake had developed. The maximum lake water depth in the pit was approximately 35 m, the surface area roughly 0.3 km^2, for a volume of approximately 9 x 10 Mm^3. The pit wall composition includes several detrital and volcanic lithologies such as felsic epiclastites, dacite porphyries, purple tuffites, black shales and vitric to crystalline tuffs, in addition to the massive sulphide lenses.

Figure 5-3 Los Frailes and Aznalcóllar Pits During Operation early 1998 [9]

STRATEGIC USE OF PIT

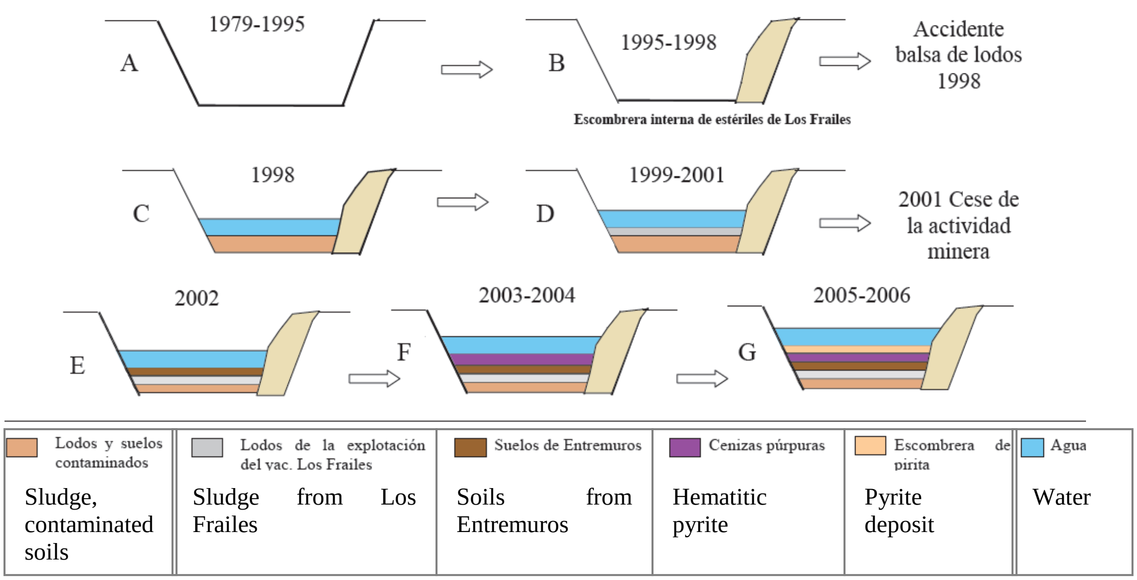
The Los Frailes pit has been flooding naturally and the water has reached a neutral pH with a low concentration of metals [16]. In the future, this pit could provide a reservoir for use in other applications.

Approximately, 7.8 Mm^3 of spilled tailings and contaminated soils were deposited into the Aznalcóllar pit in 1998-9 after the failure of the tailings dam (Figure 5-4). The pit had been used as a disposal site for a number of waste materials including acid seepage from waste rock piles after 1995. Waste rock from the adjacent Los Frailes pit (~ 18.2 Mm^3) was dumped into the pit from 1995 to 1998.

Figure 5-4 Photograph of Tailing Impoundment Wall Breach Looking Southwest [11]

As depicted in Figure 5-5, about 3 Mm^3 of metal-rich muds from the ore-processing facility between 1999 and 2001, metal-polluted soils from the Guadiamar river and Entremuros area (the area of Doñana National Park enduring greatest impact from the tailings spill) again in 2002, ~1 Mm^3 of hematitic wastes from the historic pyrite processing facility between 2003 and 2004, ~1.4 Mm^3 of pyritic wastes from an adjacent waste pile and 55,000 m^3 of treatment plant sludge from November 2005 to November 2006, and deposits of sludge and contaminated soils in early 2007 all went into the Aznalcóllar pit [18][8].

Figure 5-5 Diagram of the Various Waste Deposits into the Aznalcollar Pit from the Start of Mining Activity in 1995 to 2006 [8]



MATERIALS PLACED IN PIT AND METHODS OF PLACEMENT

Before the tailings spill, some small amounts of waste materials including waste rock, contaminated soils and industrial wastes (scrap metal and concrete) had been dumped into the Aznalcóllar pit (Figure 5-6 and Figure 5-7). The major pit filling resulted from the emergency created by the tailings release.

During the six month clean-up operation that followed the dam failure, the collected sediments were deposited in a former open-pit of the mine by end dumping at the pit edge [14]. The authorities approved the use of the open-pit based on a technical assessment from the Spanish Institute for Geomining Technology (ITGE), which concluded that there was an acceptably low risk of groundwater contamination. About 1.3 Mm^3 of tailings were removed and stored in the Aznalcóllar pit [9] [10]. In total, about 10 Mt of soil and tailings (5×10^6 m^3, more than 4 times more than the original tailings) were hauled to the pit during the clean-up operation [11]. The contaminated sludge contained high concentrations of arsenic, zinc, lead, copper, thallium and cadmium. The materials dumped into the mined out Aznalcóllar pit included a large amount of soil because of the necessity of removing underlying and adjacent soils which the tailings solids or tailings waste had contaminated [18].

A large workforce, including hundreds of workers, more than 100 haul trucks from across Europe, loaders, backhoes and graders and other equipment were mobilized as quickly as possible, so that the tailings could be excavated and returned to the mine before the wet season began in October. The heavy rains of the wet season posed a serious risk of reaction products or tailing solids from the high-sulphide tailings would be carried downstream. Fortunately, acid generation was minimal due to residual alkalinity in the tailings added in the milling process [11]. Removal was chaotic because the operation lacked proper co-ordination, adequate equipment, safety measures to prevent the alcohol consumption by the workers, and the use of adequate technology. As a result huge amounts of contaminated aerosols were re-suspended, groundwater pollution was not properly controlled, the flow of contaminated particles to the estuary continued several weeks after the spill (bound to suspended solid matter) and the watershed suffered major modifications (with broad areas of exposed soil and the associated erosion) [3].

Table 5-1 Contents of the Tailings Released by the Dam Failure [3]

Metal	Content g/tonne:
Zinc	8,000
Lead	8,000
Arsenic	5,000
Copper	2,000
Cobalt	90
Thallium	55
Bismuth	70
Cadmium	28
Mercury	15
Pyrite content	68 – 78%

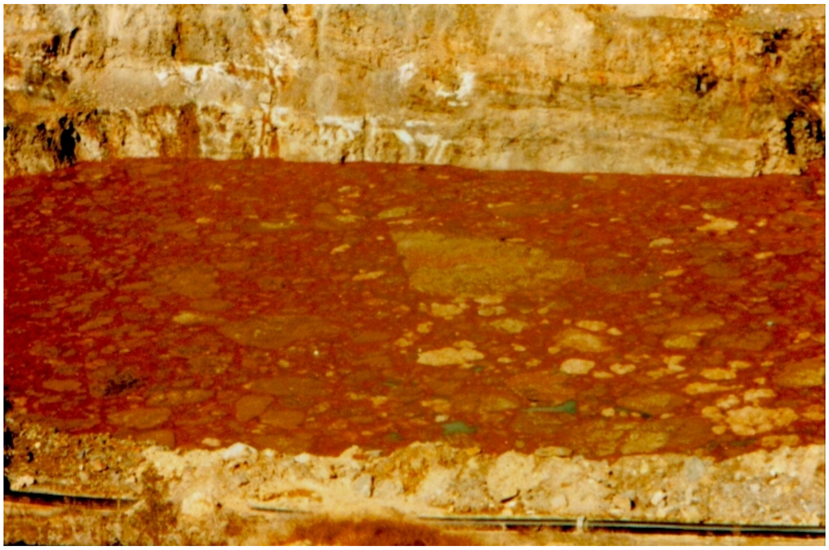
Figure 5-6 The Aznalcóllar Pit Being Filled with Recovered Tailings (G. Feasby, 1998)

When the Los Frailes Mine reopened in late 1999, the Aznalcóllar pit was used for tailings disposal instead of former surface facility which had been closed out [14] [20]. More than a million cubic meters of tailings solids were deposited in the pit by 2002 [18]. Surface runoff and ground water from the remediated former tailings pond, from the former ore processing facility and from a part of the waste rock piles were collected and periodically pumped to the pit lake. This mixture of clean (precipitation and run off) and contaminated water accounted for about 80% of the total superficial inflow into the lake. The remaining 20% was a direct inflow of contaminated water from the waste rock piles (approximately 100 ha footprint) [18].

The pit lake receives two inflows of acid rock mine drainage (ARD): the Northern Drainage (ND) drains the NW pile and enters the pit lake by its northern wall via the Agrio River, and a western inflow discharges pumped ARD from the Acid Mine Drainage Dam (AMDD), a pool which collects all the acidic leachates generated in the mine area [8]. The Aznalcóllar pit lake contains 6.05 hm^3 of highly acidic and metal-rich water, occupies a surface of 284,000 m^2, and has a maximum depth of 37 m and a relative depth of around 6%. Such low relative depth is usually associated to holomictic lakes in which a complete mixing of the water column takes place during some period of the year. Recent research carried out in this lake since 2002 indicates that the homogenization (i.e., turnover) of the Aznalcóllar pit lake water usually takes place during the winter period - November to February [18] [15].

The water level of the pit lake is artificially maintained at a designated level (regulatory level imposed by regional authorities) by periodic pumping. The pumped water is neutralized in an adjacent chemical treatment plant (where the pH is raised to around 9.5 and dissolved metals are removed), and diverted to the Agrio river. The treatment sludges are returned to the pit lake [12].

Figure 5-7 Photo of Aznalcóllar Pit Lake During Dumping (G. Feasby, 1998)



PERFORMANCE OF IN-PIT DISPOSAL

A comparison of the observed water quality with results from other pit lakes in other ore mining districts in the Iberian zone shows that some metal concentrations in the Aznalcóllar pit lake lie within the wide variation of reported data. However, other metal concentrations in the Aznalcóllar pit lake are very high, especially Zn and Cu. This is caused by the different geological settings of the pit lakes and the differences in the dominating ore minerals as well as by the limnological conditions of the pit lakes (e.g., occurrences of permanent chemical stratification, rate of water renewal, application of remediation technologies, etc.) [18]. The acidity of the hypolimnion of the Aznalcollar lake is also the highest in comparison to other pit lakes in the area's Pyrite Belt [8].

The pit lake is partly flooded with approximately 6 Mm^3 of metal and sulphate-rich water and is highly acidic with a pH of 2.7. The wastes placed in the pit included highly acidic ARD (pH 3.6) with high concentrations of heavy metals (800 mg/L Zn, 200 mg/L Mn, 100 mg/L Al, 2000 µg/L Cd) and sludge [18].

The chemical composition of the treatment plant sludges, which are systematically deposited in the pit lake (~50,000 m^3 in 2005) includes, on average, 20.3% CaO, 10.6% MgO, 1.61% Al_2O_3, 1.6% MnO, and 1.26% Zn, in addition to approximately 40% sulphated interstitial water (Santofimia, October 2005 data). Thus, the partial redissolution of these sludges in the pit lake imparts a significant enrichment in SO_4^{2-}, Ca, Mg, and metals like Al, Mn and Zn [14].

Table 5-2 Water Chemistry in the Aznalcóllar Pit Lake between 2002 and 2005 (selected parameters, all concentrations in m/L except pH, n.a. - not analysed) [18]

	November 2002	December 2003	November 2004	April 2005
pH	5.3-5.4	3.7	3.8	4.0-4.5
SO_4^{-2}	4700-4920	5490-5720	8070-8290	7790-8220
Cl^-	106-110	65.9-85.3	84.1-98.0	59.2-63.2
Na^+	84.0-86.0	73.3-75.9	51.4-75.2	70.5-94.5
K^+	33.0-34.0	25.6-26.5	13.2-19.0	17.0-24.5
Mg^{+2}	n.a.	777-807	806-1140	1120-1540
Ca^{+2}	600-620	490-509	418-591	523-730
Al	1.6-3.3	51.5-56.9	108-157	109-130
Fe	130-140	3.8-10.7	1.6-2.8	0.2-2.0
Mn	135-140	161-169	141-201	183-218
Zn	310-320	434-450	638-899	834-1010
Cu	0.03-0.13	7.9-8.2	37.0-40.0	31.8-35.2
Cd	0.65-0.68	1.01-1.05	2.10-2.20	1.90-2.05

Research by Santofimia *et al.* [15] from 2002 to 2007 showed that the reaction of the lake water with the added wastes in addition to mixing with two inflows of acid mine waters coming from a waste pile and a pool which collects acidic leachates generated in the mine area, have strongly modified the water quality of the pit lake. 2002 data showed evenly distributed temperature and EC levels throughout the water column in the winter months, demonstrating a winter mixing of the lake, as seen in Figure 5-8. In 2002, a deposit of soils into the pit lake created a highly anoxic environment [8]. A deposit of pyrite in 2004 increased the EC to 7.4 ms/cm from 5.7 ms/cm and raised the concentrations of all metals present in the pit [8]. Aluminum has created an unusual system by acting as a pH buffer in the hypolimnion of the pit, which is not often found in mine lakes [17]. In 2004, the pH ranged from 4.2 in the hypolimnion to 3.6 in the epilimnion. The summer months presented a slight temperature stratification (20 °C in the epilimnion and 15 °C in the hypolimnion) and the oxygen levels in the bottom layers of the lake were quite low (~60%) [17].

A major and dramatic highlight in the water quality of the pit lake has been the addition of 1.4 Mm^3 of pyritic wastes from an adjacent waste pile during the period from November 2005 to November 2006. The oxidative dissolution of this pyrite has resulted in: a total consumption of dissolved oxygen; a notable increase of total dissolved solids (conductivity 8.6 to 12 mS/cm); and a strong acidification (pH 4.2 to 2.7). In addition, from January to October 2006 an important heating of the hypolimnion was observed. This heating is due to the exothermic nature of pyrite oxidation, and is three times greater than the one observed during a similar period in 2005 (ΔT=5.3ºC *vs*. 1.6ºC, respectively). Despite the virtually total lack of oxidizing agents such as O_2 and Fe(III) (Fe(III)<5% Fe_{total}) in the pit lake, pyrite is being oxidized and dissolved at high rates, as evidenced by the continuous increase of the SO_4^{2-} and Fe concentrations [15].

The two AMD (acid mine drainage) inflows (locally defined as ND and AMDD) represent an important input of acidity and metals for the water quality of the pit lake. The flow rates of both inflows are highly dependent on the rainfall discharge, but their respective water volumes during the studied time interval, November 2005 to November 2006, have been estimated to be around 4,700 m^3 (ND) and 5,500 m^3 (AMMD) [15]. All the toxic deposits into the lake during this period including 55,000 m^3 of treatment plant sludge raised the height of the pit lake by 1.6 m [17].

Figure 5-8 Various Parameter Profiles of the Aznalcollar Pit Water Column from 2002 to September 2007 [8]

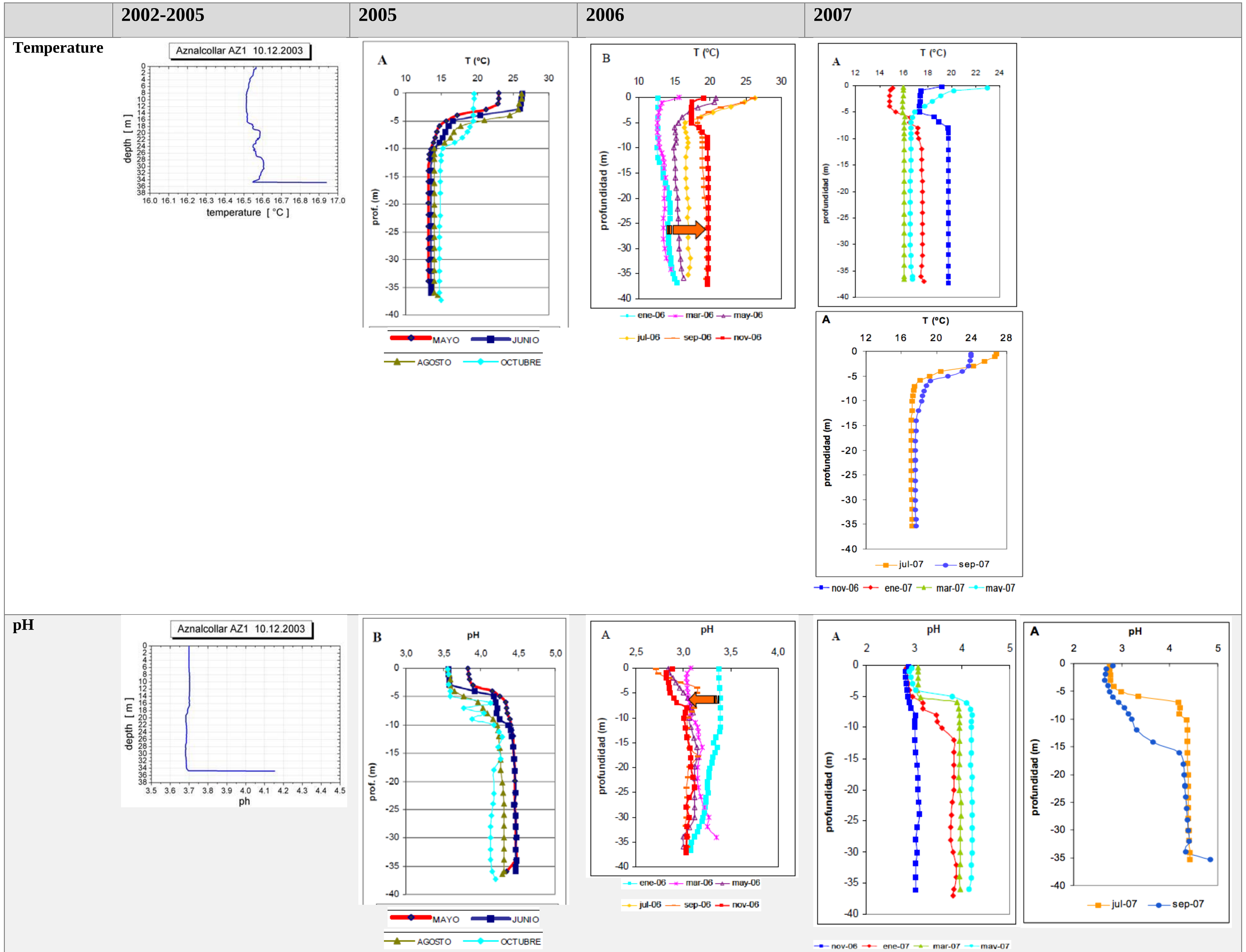


Figure 5-8 Various Parameter Profiles of the Aznalcollar Pit Water Column from 2002 to September 2007 [8] (Cont'd)

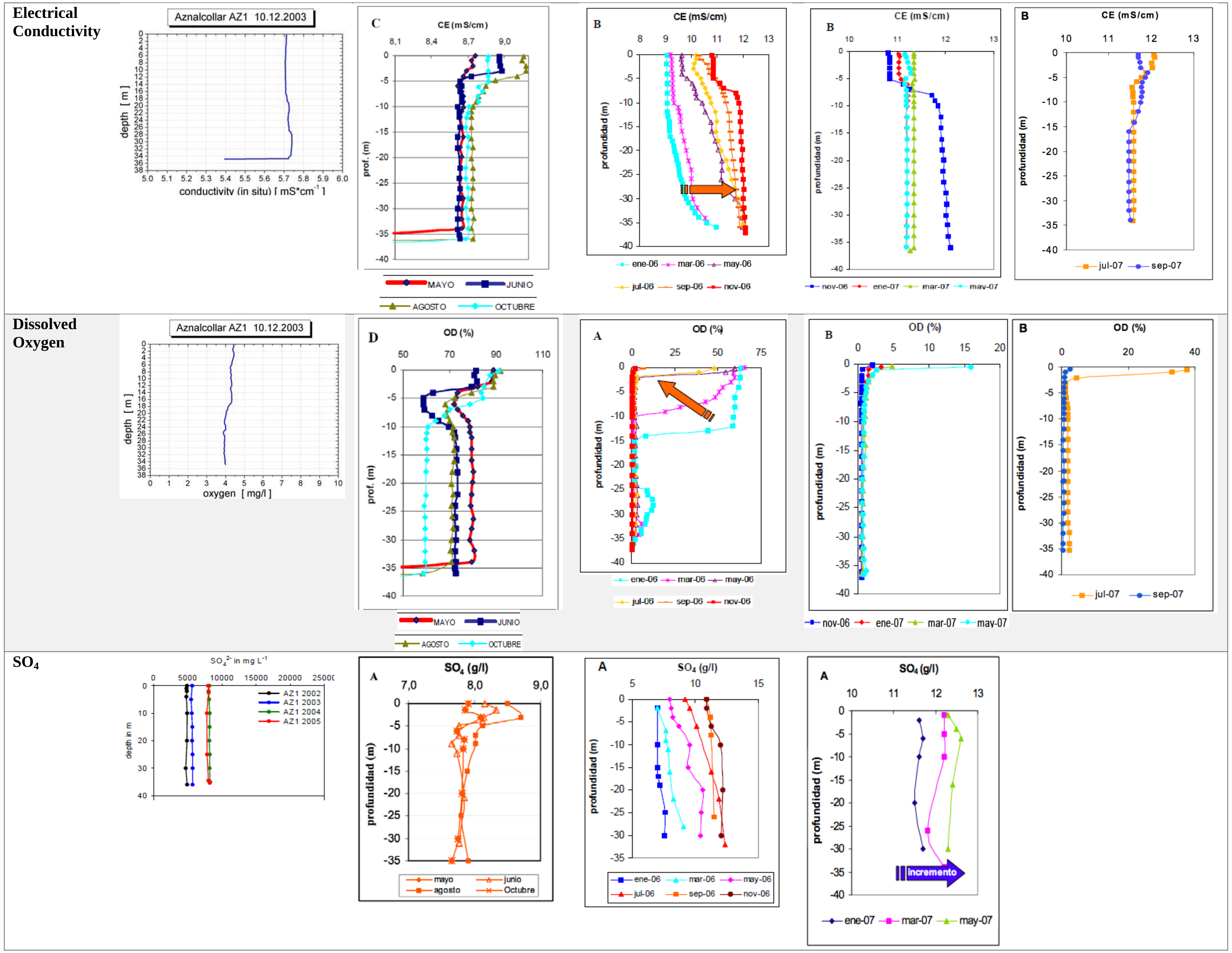


Figure 5-8 Various Parameter Profiles of the Aznalcollar Pit Water Column from 2002 to September 2007 [8] (Cont'd)

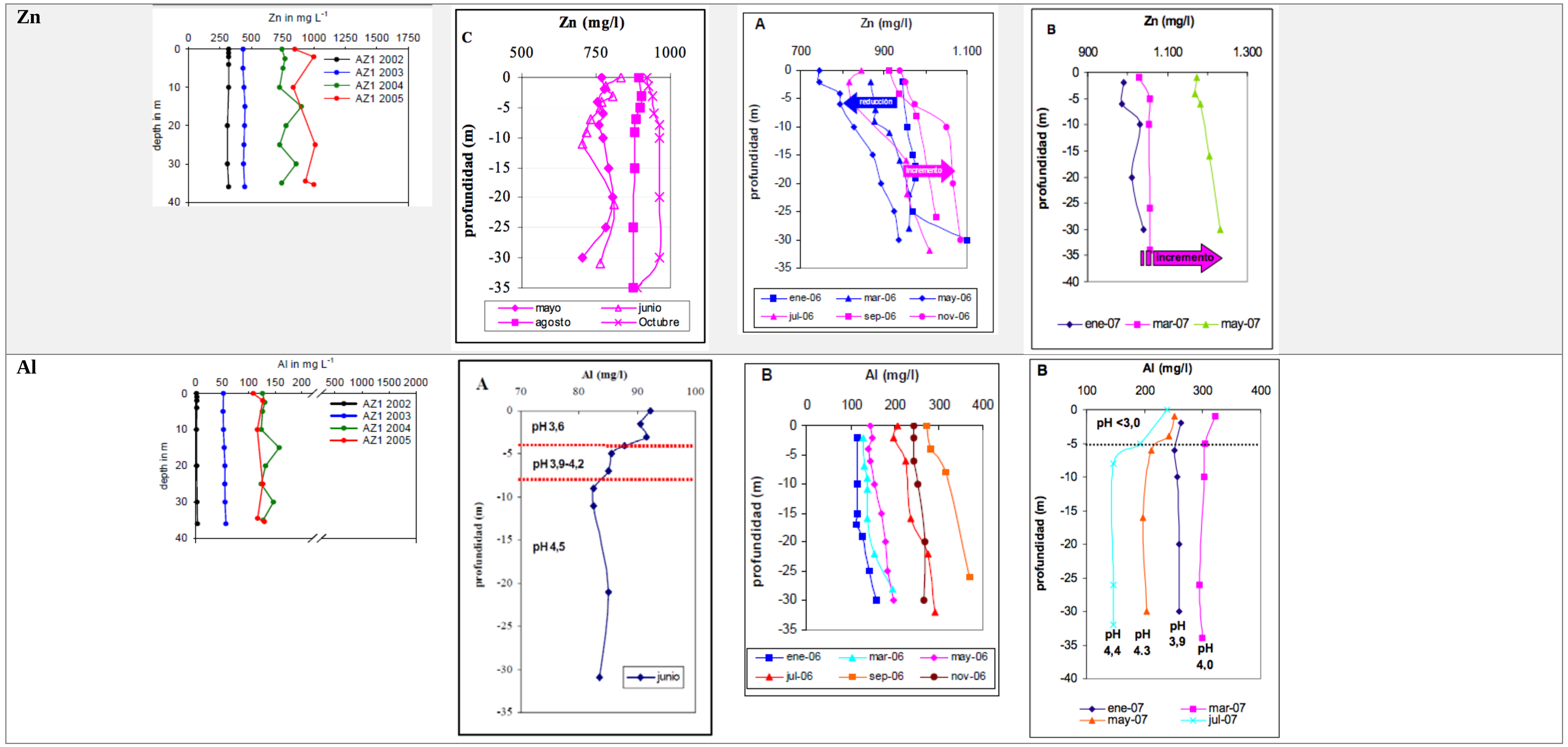
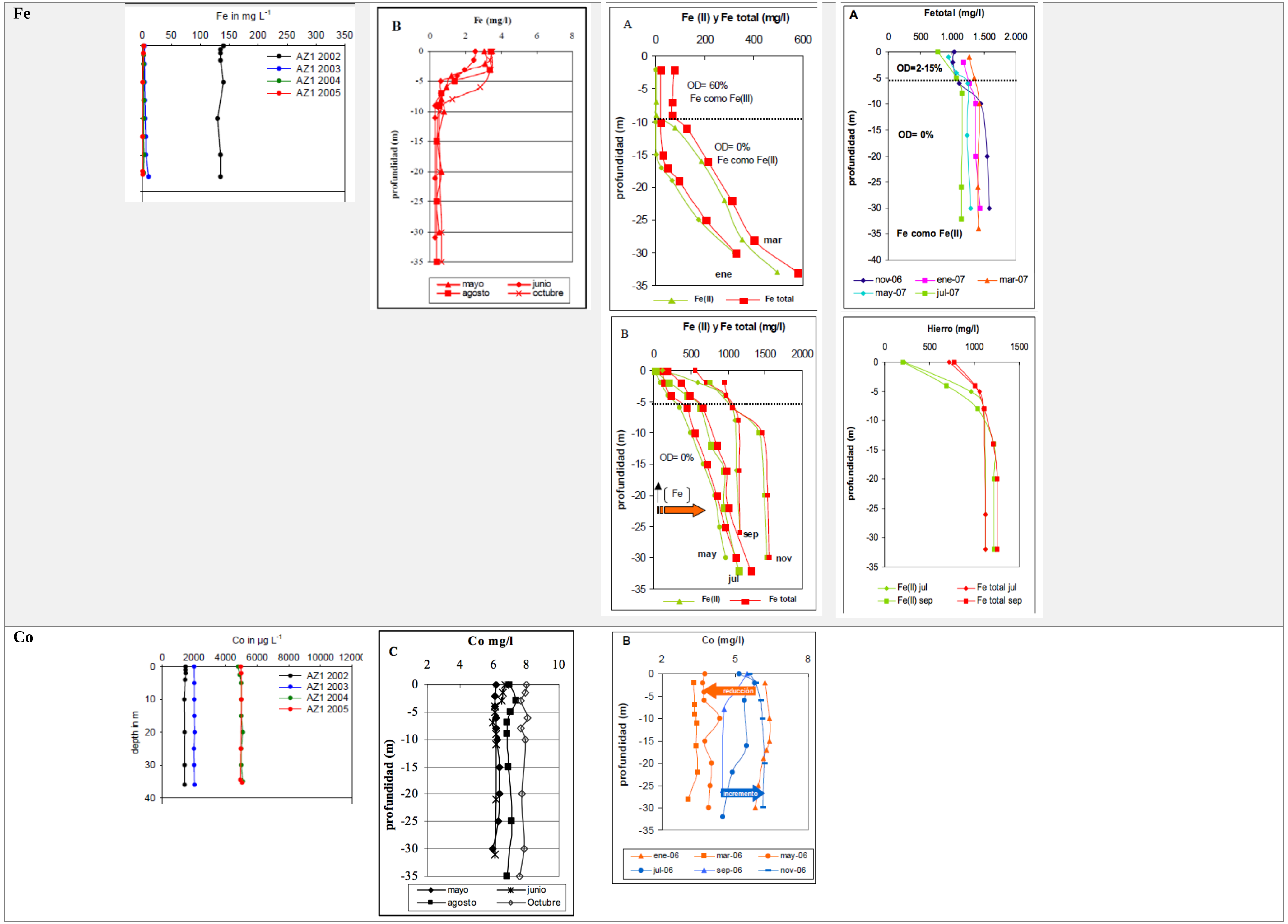

Zn
Zn in mg L-1
depth in m
AZ1 2002
AZ1 2003
AZ1 2004
AZ1 2005
C
Zn (mg/l)
profundidad (m)
mayo
junio
agosto
Octubre
A
Zn (mg/l)
profundidad (m)
reducción
incremento
ene-06
mar-06
may-06
jul-06
sep-06
nov-06
B
Zn (mg/l)
profundidad (m)
incremento
ene-07
mar-07
may-07
Al
Al in mg L-1
depth in m
AZ1 2002
AZ1 2003
AZ1 2004
AZ1 2005
A
Al (mg/l)
profundidad (m)
pH 3,6
pH 3,9-4,2
pH 4,5
junio
B
Al (mg/l)
profundidad (m)
ene-06
mar-06
may-06
jul-06
sep-06
nov-06
B
Al (mg/l)
profundidad (m)
pH <3,0
pH 4,4
pH 4.3
pH 3,9
pH 4,0
ene-07
mar-07
may-07
jul-07

Figure 5-8 Various Parameter Profiles of the Aznalcollar Pit Water Column from 2002 to September 2007 [8] (Cont'd)



The disposal of the extensively oxidized pyritic wastes resulted in a strong and progressive modification of the pit lake water chemistry. The water experienced drastic changes in EC (from 8.6 to 12 mS/cm), T (from 13 to 20ºC), DO (from 75% sat. to 0% sat.), pH (from 3.2 to 2.7), and Eh (from 700 to 500 mV; data not shown), as well as drastic increases in the concentration of most elements, as for example sulphate (from 7.8 to 11.3 g/L), Fe (from 300 to 1,500 mg/L), Al (from 150 to 300 mg/L), and As (from 60 to 2,300 μg/L) (Figure 5-8) [15].

The oxidative dissolution of pyrite provoked a total consumption of the oxygen dissolved in the pit lake. The exothermic reaction is believed to have resulted in the higher temperature of the hypolimnion than that observed during the previous year (ΔT=5.3 *vs*. 1.6ºC). Dissolved iron in the lower (anoxic) layer of the pit lake is >95% Fe (II). However, pyrite dissolution appeared to have continued anaerobically due to the presence of Fe(III), which could be provided by dissolution of sulphates present in the wastes (such as arsenopyrite and tetrahydrite-tennantite), and/or previous Fe(III) precipitates [15].

From November 2005 to February 2006, two opposite and competing processes were simultaneously taking place, namely: the progressive cooling of the surface water, which favours the mixing and homogenization processes; and the oxidative dissolution of pyrite, which gradually increases the density of water in the lower layer impeding mixing and enhancing stratification [15].

In coherence with the dissolution of pyrite, both Fe (II) and SO_4^{2-} are the substances which most importantly have been increased in concentration. However, some other trace elements such as Cu, As and Pb have also been considerably enriched, as a result of dissolution of other sulphides (sphalerite, chalcopyrite, galena, arsenopyrite) and sulphosalts (tetrahedrite-tennantite) which were also common in the mineralization [15].

The increases of Al and Mg (80 to 300 mg/L, and 1,150 to 1,300 mg/L, respectively) are most likely the result of the dissolution of aluminosilicates and/or Al precipitates. Conversely, from January to May 2006, Zn and Cd exhibited a striking behaviour, being reduced in their respective concentrations in the upper (oxygenated) layer of the pit lake (e.g., Zn decreased from 970 to 850 mg/L, but rebounded to its initial content by November 2006). This could have been a result of high concentrations of iron which produces a strong precipitation of schwertmannite, adsorbing trace elements in an anionic form [8]. After May 2006, the pit lake became anoxic, and this reaction could not take place, so that no additional input of Fe (III) occurred, and the formation of schwertmannite was no longer possible. Under these new conditions, both Zn and Cd increased again. Finally, the disposal of oxidised wastes resulted in a significant increase of the total (mineral) acidity of the pit lake (from 2,000 to 6,400 mg/L $CaCO_3$ eq.) [15].

Small deposits of sludges (high in metal sulfides) and other contaminated material worsened the already toxic conditions of the pit lake from January to May 2007. The winter months showed abnormal temperature and electrical conductivity (EC) patterns with an inverted stratification of EC and temperature. This was most likely due to the high density of solids and the dissolution of pyrite in the hypolimnion from remnants of the 2006 pyrite deposits. The oxygenation of the superficial layer caused the oxidation of Fe (II), reducing the EC of the epilimnion. The slightly higher EC of the epilimnion also suggests a partial mixing of the upper waters [8].

The pH of the pit is greater in the hypolimnion possibly due to the dissolution of carbonate materials from the sludge deposits and the lack of the buffer effect typically provided by Fe (III). In May, the pH of the hypolimnion increased to 4 which is a level that facilitates the hydrolysis of aluminum as a buffer, while the hydrolysis of Fe (III) in the epilimnion adds to the acidic environment [8].

The sulfide rich sludge deposits of 2007 increased the concentrations of SO_4, Mg, Zn, Al, Ni and Cd in the pit most likely through dissolution of the minerals, and also presented a non-homogenous level of iron and aluminum throughout. The Fe and Al levels correlate to the stratification of the pH. Due to the anoxic conditions at the bottom of the pit, the mass of sludge settled at the bottom had not dissolved as of 2007, leaving it in a preserved state [8].

In 2009, metal concentrations were monitored in the Agrio and Guadiamar rivers which flow around the tailings dam. The documentation showcased fairly high levels (Table 5-3) although much lower than was seen shortly after the 1998 dam failure [13]. For example, levels of zinc in the Agrio River aquifer were noted to be 300 mg/L directly after the spill but were a maximum of 16 mg/L from the 2009 sampling. Sulphate has followed a similar trend [13]. Higher concentrations of zinc and sulphate were seen in the areas of the river closer to the Los Frailes tailings dam [13]. It should be noted that toxic element concentrations in the Agrio River were already relatively high prior to the Aznalcollar dam failure.

Table 5-3 Field Measurements and Analytical Results taken from Groundwater in the Guadiamar River and in the Agrio River Zone, in September 2009. Note-Samples were taken from piezometers (P) in the Agrio River, wells (W) in the Guadiamar alluvial, and Agrio River surface water (A) [13]

Code	T	Temperature, °C	EC, µS/cm	pH	Eh, mv	Cl, mg/L	SO_4, mg/L	HCO_3, mg/L	NO_3, mg/L	Ca, mg/L	Mg, mg/L	Na, mg/L	K, mg/L	Al, mg/L	As, µg/L	Ba, µg/L	Be, µg/L	Cd, µg/L	Co, µg/L	Cu, µg/L	Fe, mg/L	Li, µg/L	Mn, mg/L	Ni, µg/L	Pb, µg/L	Si mg/L	Sr, µg/L	Zn, mg/L
P1	T2	26.1	908	7.04	216	70.1	199	242	1.4	133	27	42	3.7	0.11	<3	44	<0.3	<1	<3	9.3	0.45	61	0.1	<8	<30	11	1,051	0.03
P2	T1	23.7	1,695	4.93	455	31.1	1,100	<5	4.9	260	109	38	3	1.2	<3	14	0.95	15	44	219	0.38	115	4.2	59	<30	12	858	5.5
P3	T1	24	2,550	5.40	472	68.1	1,789	<5	10.2	328	229	46	7.9	<0.05	<3	15	<0.3	4.8	<3	24	<0.2	109	0.13	15	70	8.3	700	0.81
A1	T0	22.5	2,980	4.54	540	66.4	2,237	<5	15.4	418	259	49	7.3	5.7	<3	18	<0.3	47	74	393	0.46	323	8.3	87	64	14	799	24
P4	T1	25.7	1,114	3.74	617	28.1	714	<5	5.8	132	67	20	4.8	7.7	<3	12	1.1	12	26	357	<0.2	134	1.6	39	43	22	303	3.2
P9	T1	23.7	1,365	3.98	554	31.5	925	<5	7.9	173	93	24	5.1	6.1	<3	13	1.7	16	34	413	<0.2	147	3.6	62	<30	17	376	4.5
P9b	T2	25.3	1,634	4.83	425	34.8	965	<5	<1	220	122	34	5.2	0.88	<3	15	0.57	<1	6.8	67	8.5	48	1.4	18	<30	6.2	672	0.38
P10	T1	22.7	1,447	3.82	551	33.3	946	<5	9.4	219	62	27	6.3	12	<3	11	3.2	35	94	778	4.0	204	8.5	123	76	24	534	13
P10b	T1	22.5	1,701	3.88	582	37.3	1,110	<5	12.5	276	75	29	4.7	15	<3	10	3.6	45	105	1,051	<0.2	226	8.6	144	50	24	679	16
P11	T1	21.4	1,357	6.18	456	113.0	533	89	112.2	210	42	64	3.3	<0.05	<3	15	<0.3	2.9	<3	16	<0.2	32	0.05	<8	<30	7.2	923	0.69
P12	T0	22.8	500	6.22	387	17.9	254	12	3.5	65	30	14	3.6	0.07	9.2	18	<0.3	3	<3	21	<0.2	47	0.24	18	<30	5.2	181	0.86
A2	T0	22.9	2,120	5.29	515	56.3	1,435	<5	15.0	352	151	42	4.1	1.1	<3	24	<0.3	15	22	271	<0.2	128	2.3	38	<30	11	754	4.8
P13	T0	26.2	858	4.09	579	34.5	434	<5	16.6	133	28	27	3	1.9	<3	8.7	<0.3	11	16	255	<0.2	48	1.5	29	<30	10	371	2.9
P13b	T1	24.8	1,292	4.89	491	51.0	683	<5	47.1	223	45	41	2.7	0.97	<3	14	1.2	13	14	251	<0.2	54	1.5	45	73	11	668	3.8
P14	T2	20.9	1,590	4.03	517	31.0	1,080	<5	9.7	241	85	26	6.9	7.2	<3	12	2.9	47	78	871	<0.2	241	8.6	140	<30	22	737	18
P15	T1	23.9	2,520	6.66	404	42.1	1,632	59	17.6	470	137	47	4.8	<0.05	<3	14	<0.3	<1	<3	<8	<0.2	154	0.01	7.7	<30	17	1,966	0.26
P16	T0	24.5	2,490	5.28	415	59.7	1,603	<5	<1	395	173	45	5	0.11	<3	20	<0.3	6.2	<3	18	0.36	99	0.05	19	<30	6.3	849	1.2
W19	T2	23.3	3,460	2.91	749	38.0	2,649	<5	<1	527	132	42	5.6	98	35	7.5	6	49	318	865	15	332	40	291	59	43	1,819	20
W22	T2	21.2	1,400	6.65	341	70.6	644	193	<1	261	42	47	6.9	1	<3	45	<0.3	<1	11	21	0.52	<20	0.87	15	<30	8.5	822	0.63
W30	T1	22.6	1,153	6.90	354	119.9	337	157	26.6	147	50	55	4.8	0.06	<3	30	<0.3	1.2	<3	<8	0.37	<20	0.2	<8	<30	6.2	422	0.12
W31	T1	20.5	1,241	6.70	402	71.3	489	189	4.7	202	42	47	4.9	0.08	18	19	<0.3	<1	<3	12	<0.2	34	0.01	<8	<30	5.7	540	0.03
W32	T1	20.5	1,820	3.54	679	108.0	1,033	<5	6.3	216	54	99	9.8	21	<3	14	2	30	72	665	0.92	79	3.2	71	<30	24	715	6.2
W33	T1	20.4	1,391	6.58	412	71.4	690	120	4.6	252	47	38	5	<0.05	<3	19	<0.3	<1	<3	<8	0.37	29	0.01	<8	<30	8	597	0.03
W35	T1	19.7	1,275	6.85	408	58.6	498	265	2.6	237	40	50	6.1	<0.05	<3	29	<0.3	<1	<3	8.2	<0.2	37	0.01	<8	<30	7.2	546	0.03
W43	T1	19.3	741	7.65	379	82.4	114	239	1.9	84	26	48	12	<0.05	4.2	83	<0.3	<1	<3	<8	<0.2	<20	0.01	<8	<30	1.6	355	0.04
Mean		22.8	1,624	5.30	476	57.0	964	156	16.0	247	87	42	5.5	9.48	16.6	21.0	2.3	20.8	65.3	314	2.8	122	3.8	67.8	62.1	13.3	729	5.08
Min.		19.3	500	2.91	216	17.9	114	<5	<1	65	26	14	2.7	<0.05	<3	7.5	<0.3	<1	<3	<8	<0.2	29	0.01	<8	<30	1.6	181	0.03
Max.		26.2	3,460	7.65	749	119.9	2,649	265	112.2	527	259	99	12.0	98.0	35.0	83.0	6.0	49.0	318.0	1,051	15.0	332	40.0	291.0	76.0	43.0	1,966	24.0
SD		2.0	713	1.35	116	27.6	636	85	24.4	115	64	17	2.2	22.2	13.5	16.1	1.6	17.4	79.8	339	4.8	92	8.1	70.6	12.2	9.1	409	7.16

REMEDIAL

There is the possibility that without the extremely difficult task of sealing the pit walls, the Aznalcóllar pit waters may contaminate ground water. The pit is located within materials which have been strongly folded, thrusted, and shistosed (slates) originating from the Variscan orogeny. Due to the presence and abundance of strongly dipping planar structures (schistosity, thrusts, discloses), there are no impermeable zones (fracture permeability) to prevent migration to the groundwater. The percolating waters will flow through sulphide-rich rocks and within this retention system (an anoxic acidic medium) and arsenic could be solubilised in its more mobile and toxic form (As^{3+}) and percolate down flow [14].

In the 2006 report by Schultze *et al.* [18] there was no chemical, biological or physical treatment option able to serve as a single approach for successful remediation of the Aznalcóllar pit lake. Some options for remediation outlined by Schultze *et al.* include chemical neutralization, neutralization based on sulphate reduction, removal of the metals by artificial eutrophication, the introduction of river water and artificial creation of meromictic conditions. They also suggested additional alternatives which could include control of the runoff from the waste rock piles, submergence of sulphide pit walls, partial and complete backfill of the pit, but these approaches can only be successfully employed if there is no or only a very limited further introduction of acidity, heavy metals and trace elements into the lake. It was also suggested that some measures are necessary for the management of the water quality and solids that enter the pit, such as sludges, partially and untreated non pit waters, and contaminated wastes. Without these actions, they feel that the remediation approach will fail completely or has to be employed frequently, but with a combination of options may provide approaches from the primary rehabilitation of the Aznalcóllar pit lake, as well as for a sustainable long-term management of the lake [18].

BENEFITS and LESSONS

The Aznalcóllar pit at the Los Frailes mine served as an emergency disposal area following a tailings spill. It has also been used to deposit various ARD producing wastes including some highly oxidized materials. No attempts have been made to neutralize the wastes in place and a highly contaminated pit lake has developed.

The level of this pit lake must be controlled to hydrologically isolate the pit waters from local aquifers which are important sources of domestic and agricultural water supplies. Pit water is removed by pumping, combined with seepage from waste rock piles and the closed out surface tailings facility, treated, and discharged to the local environment. The treatment sludge is returned to the pit.

It is expected that this treatment process will continue indefinitely. Treatment requirements (e.g., lime demand) are expected to decline slightly in the near future with treatment sludge blanketing the solid wastes and the occurrence of some sulphate reduction. However volumes to be treated are expected to be approximately constant for the near future, varying only with precipitation and evaporation.

RESIDUAL ISSUES

The long term commitment to continuous treatment is a challenge.

No evidence has been presented of attempts for in-situ treatment of the pit lake. There are two reasons for this: the high levels of acidity and metals in the pit lake; and the continued production of soluble metals and acidity in the pit, as well as from waste rock piles, the Los Frailes pit and the closed out surface tailings facility.

Monitoring of ground water quality to determine if contaminants are migrating in significant amounts will need to be maintained. Should such contamination be discovered, interception of contaminant plumes would need to be considered.

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CASE STUDY #6 STRATMAT MINE

SYNOPSIS

The Stratmat mine in northeastern New Brunswick operated from 1989 – 1993, and is currently owned by Kria Resources (Kria).

The proposed closure plan for Stratmat was to back fill pit openings with waste rock, allowing the pits to flood and cover the pits with overburden. However, after extensive review of the site, it was concluded that the in-pit disposal of waste rock would not be ideal because it would cause a release of stored metals and sulphate which would have migrated into the groundwater.

LOCATION

Stratmat and Halfmile Lake deposits are located in northeast New Brunswick; approximately 155 km north of Fredericton; 48 kilometres southwest of Bathurst and 55 km northwest of Miramichi (47°13’ N and 66°07’ W) [3]. It is adjacent the former Heath Steele Mine (to the north) and 24 km from the Brunswick 12 deposit [1] [3]. The Stratmat Property covers an area of 828.6 hectare on Crown-owned land [3] near the Tomogonops River (a tributary of the Miramichi River), draining into the Little South Tomogonops River. The area has significant relief, extensive bogs, and groundwater discharges [9].

Figure 6-1 Location of Stratmat Mine, New Brunswick [3]

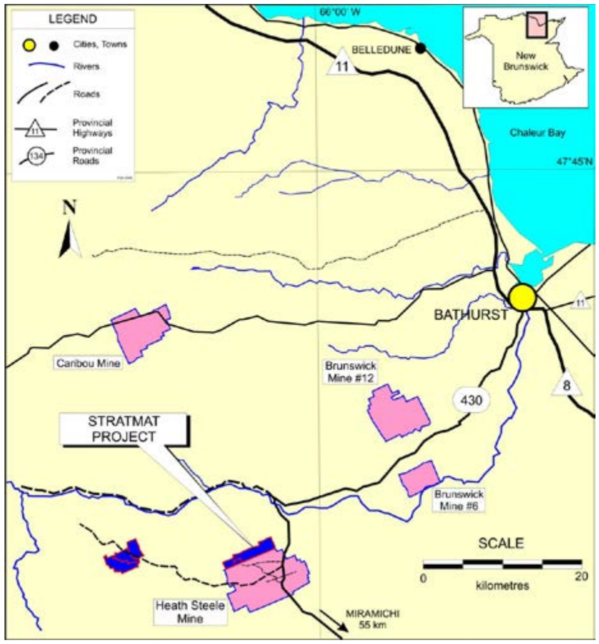
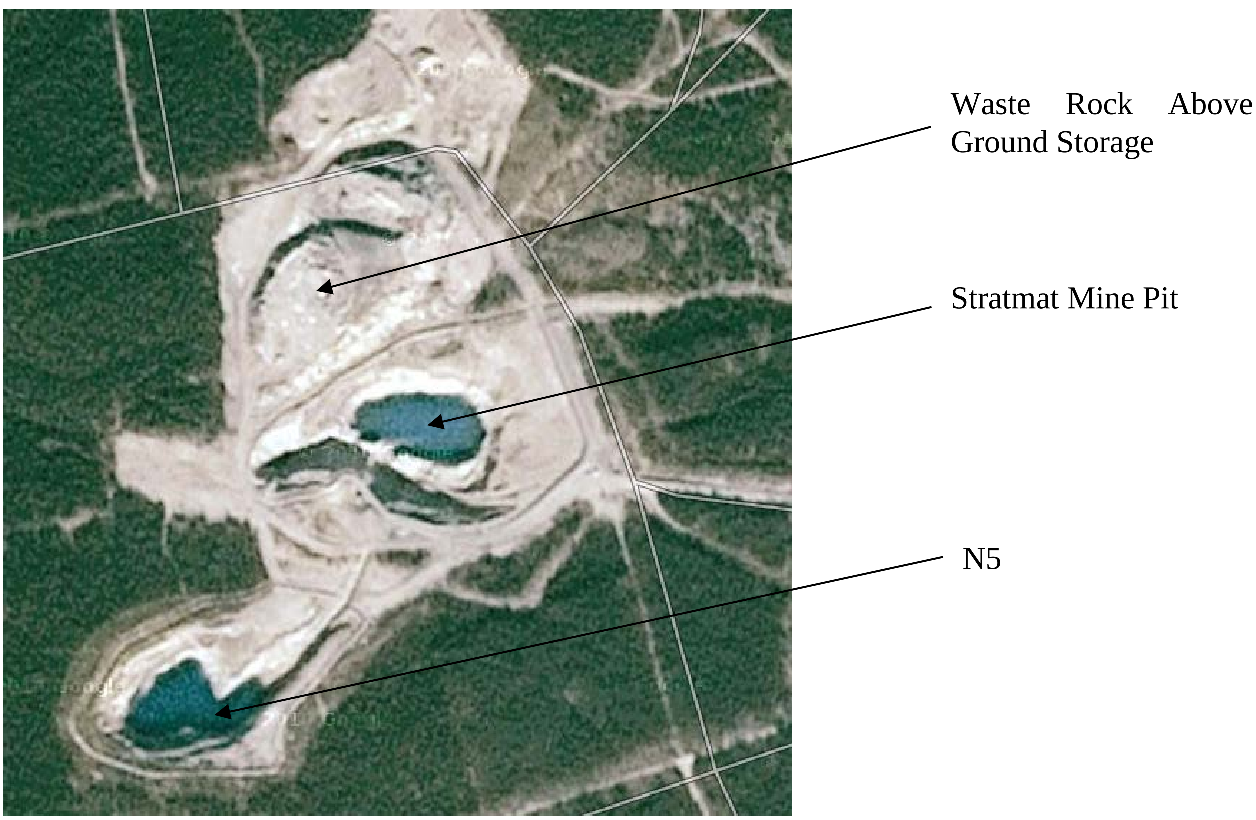


Figure 6-2 Stratmat and N5 Mine (Google 2010)



MINE TYPE and HISTORY

The property was originally staked by Strategic Materials Limited (abbreviated to Stratmat), during the Heath Steele staking rush of 1954. In 1959, Cominco purchased the property, holding the mineral rights until 1985. On January 1, 1986, Noranda optioned the Heath Steele property from Cominco and opened the mine in 1989. The Stratmat satellite mine operated from 1989 to 1993 mining copper, lead, silver and zinc. In 2006 Xstrata took over Noranda [3], and in 2008 Kria gained 100% ownership of the Stratmat and Halfmile Lake projects [3][4].

The Stratmat open pit and underground mine produced about 1,018,000 tonnes of waste rock with a swelled volume of 1,449,000 m. The combined volume of the overburden and waste rock was 497,000 m^3 [9].

In July 2009 Kria completed a Preliminary Economic Assessment (PEA) of the potential resources remaining on the Stratmat property indicating an Inferred Mineral Resource of 5.52 million tonnes.[1].

PIT(s) DESCRIPTION

The surface area for the open pits is estimated to be 62,000 m^2 (6.2 hectare) with approximately 46,000 m^2 having acid generating potential. The average pit depth was 30 m with an overall slope of 50^o (45^o near the top and 60^o near the bottom). A combined volume of 497 000 m^3 of overburden and waste rock was stripped to access the ore. Immediately north and upgradient of the pit there is a 1.449 Mm^3 waste rock pile from the open pit and connected underground mine openings [9].

The sulphide ore varied from massive to disseminated with relatively weak argillite, containing bands of talc, as the host rock. Extensive folding of the mineralized horizon had occurred, resulting in a near horizontal portion of the orebody near the surface. This part of the deposit was extracted by open pit mining. The remainder of the orebody dipped steeply and was extracted by underground mining [9].

The overburden removed from the pit prior to mining was contaminated with base metals. The overburden was exposed to mildly acidic surface and ground water flows prior to development. [9].

The water table grades southeasterly just 3 to 5 meters below the ground surface and the estimated velocity is about 60 m/yr near the pit [9]. The underground mine opens into the pit, thereby collecting the mine water [9]. Hydraulic conductivities in the Stratmat rock formations are assumed to be on the order of 10^{-5} m/s, with higher conductivities in the fractured rock near the surface and lower conductivities at depth. This results in groundwater velocities of approximately 60 m/year in the vicinity of the open pit, and 3 m/year between the existing stockpile area and McCormack Lake (the main groundwater discharge area of the site) [9].

Figure 6-3 Stratmat and N5 Open Pit Mines [2]

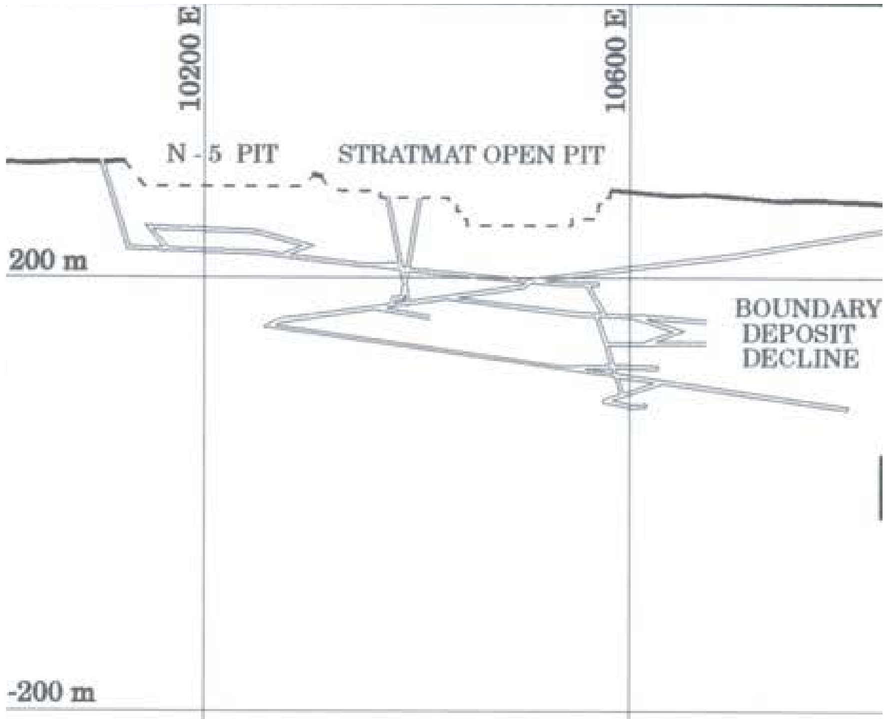
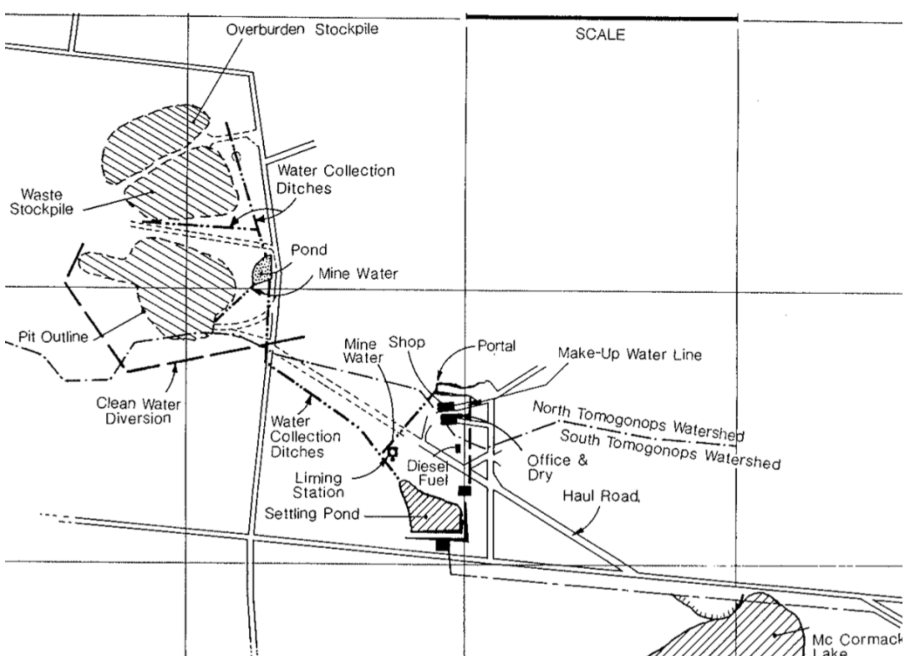


Figure 6-4 Stratmat Surface Plan [9]



(PROPOSED) STRATEGIC USE OF PIT

In 1987-1988 an ecological survey was conducted to establish the ecosystem baselines for the area which is surrounded by wetlands. The Heath Steele-Stratmat Closure Plan (1989) is designed for the restoration of the site environment and the removal of infrastructures. The plan includes placing the waste rock removed from the open pit and underground mines back in the pit with alkalinity addition as needed. It was proposed that the 11.6 hectare stockpile be limed between the shallow lifts. The tailings would be flooded by sealing the decline portal allowing the ground water to re-establish to near pre-mine levels. A cover was to be placed so as to reestablish the vegetation and the wetlands would create an anaerobic layer. Approximately, 179,000 m^3 of overburden was to be stockpiled for use in the cover [9].

The waste rock was determined to contain up to 20% pyrite minerals and less than 0.2 wt. % carbonate as CO_3 [14] with massive sulphide mineralization and sulphide oxidation below the water table. The groundwater total dissolved solids (TDS) were 200 to 1600 mg/l at 84–89 m in depth. Groundwater ranges from low TDS $CaHCO_3$-type water at shallow depth to higher TDS $CaSO_4$-type water within the massive sulphides. With increasing depth through the massive sulphide zone, base metal (Cu, Zn, Mo, Pb, Cd) and Fe concentrations increase (e.g., Fe increases from 40 to 2300 mg/l). The stable isotopes, TDS and straddle-packer recovery time data indicate that groundwater flow is greatest in the upper parts of the massive sulphides [14].

Monitoring stations where installed upradient and downgradient from the open pit mine, allowing for collection of water samples from a depth of 60 m [9].

Table 6-1 Experimental Evidence of Highly Channelled Waste Rock [12]

Channel size	% of total drainage area	% of total drainage flow
Large	<5%	20 – 30%
Intermediate	~ 20%	~ 40%
Background/matrix flows	~ 50%	~ 40%
Column base area	~ 30%	Does not intercept any flow

There were no solubility-controls for the drainage of zinc (Zn), Iron (Fe) and sulphate (SO_4^{2-}). The soluble zinc was generated from the original pore water and not from the dissolution of secondary minerals. The porous media flow rock was differentiated from the channelling flow rock (based on the hydraulic properties) [12].

PERFORMANCE OF IN-PIT DISPOSAL

In 1999, experimental data lead to the conclusion that putting waste rock into the pit would release significant concentrations of stored sulphates and metal [12] [13]. The release of sulphates and metals into the pit lake would have contaminated the groundwater supply as well. Modelling suggested that the concentration control mechanisms through backfilling-flushing-treatment process would last a long time before the pore water would be acceptable for discharging into the groundwater. The pit was not backfilled with the oxidized waste rock [12].

REMEDIAL

In-pit disposal was not carried out due to the extensive underground workings that were open to the pit, porosity/channeling of the rock, and due to the potential for the significant release of minerals and sulphate into the groundwater regime. In-pit disposal did not allow for adequate control of acid drainage and leaching from the waste rock, and water flow from the underground mine workings into the pit further complicated the control of water levels and contaminates. Mainly, there was too much waste rock for storage in the pit, and the piles where left in place.

BENEFITS and LESSONS OF IN-PIT DISPOSAL AT STRATMAT MINE

Upon detailed analyses, Stratmat was clearly a case where backfilling with acid generating and metal leaching mine waste rock was not of net benefit.

RESIDUAL ISSUES

Run-off from the waste rock pile and overburden stockpiles will be collected in ditches that report to a liming station for treatment [9].

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CASE STUDY #7 JUNDEE MINE

SYNOPSIS

Operated by Newmont (recently sold to North Star Mines, May 2014), the Jundee Mine is located in Western Australia in the Little Sandy Dessert. There were several open pits, with open pit mining starting in 1995 and finished by 2007, leaving the two underground mines producing gold. Recently, Newmont sold the Jundee mine to North Star Resources, and as of May 2014, there are an estimated two to three years of reserves remaining [13].

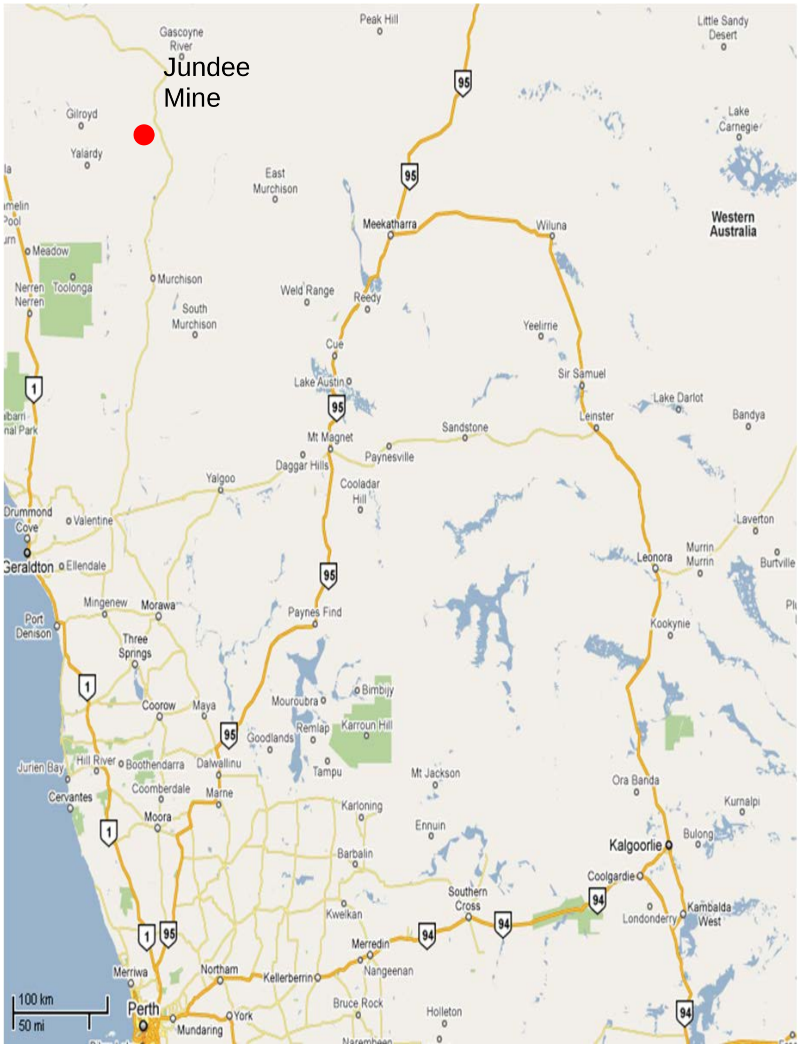
The open pits were filled with waste material: the Fisher pit was filled with tailings; and the Reid, Barton and Main pits were filled with waste rock. The Fisher pit is unlined and has a number of recovery bore sites to intercept seepage and prevent it discharging into the ground water. Many of the pits will have depressions after filling and standing water will remain trapped, preventing contamination of groundwater.

The isolation of contaminants arising in the wastes of the pits is possible because of the very dry climate and the permanent cone of depression in the pits. Currently, groundwater weak-acid-dissociable (WAD) cyanide levels have remained below the regulatory standards, and in the pit lake, the water is slightly alkaline and saline.

LOCATION

Jundee Mine Operation is located in the Eastern Goldfields (the Yandal Goldfield [7]) of Western Australia (NJO), about 800 km north east of Perth and 400 km north of Kalgoorlie. The nearest settlement is the predominately indigenous community of Wiluna, approximately 50 km south west of the mine [4]. It is in the Archaean Yandal Greenstone Belt in the Little Sandy Desert in central Western Australia's desert [10]. The average annual rainfall is 240 mm with an annual evaporation rate of approximately 4,000 mm [10]. Figure 7-1 shows the location of Jundee Mine.

Figure 7-1 Jundee Mine location in Western Australia (Google)



MINE TYPE and HISTORY

Newmont's Jundee Operation (now North Star's) is comprised of two historically separate operations, Nimary and Jundee, both of which were ultimately acquired by Newmont Asia Pacific in 2002. In 1995, open pit mining began at Jundee, and in 1997 underground mining operations began [8]. In 2007, the mine operation consisted of 2 underground mines which produce gold from very thin high grade ore veins [4][8] and the mine was concurrently managing programs of exploration, mining, rehabilitation and closure [4]. The Jundee operation is a fly-in fly-out (FIFO) employing 155 Newmont employees and 373 contactor staff [4][7].

Although closure has been planned to occur in 2013, Jundee has successfully and consistently replaced its reserves for many years and the mine is confident the operation will continue past this time [4]. It was reported in 2014, at the time of sale to North Star Resources, that two to three years of reserves remain [13]. The Jundee pit was closed in 2002 and the final pit was closed in 2007. Open-pit mining will resume if drill tests reveal sufficient yields away from the original mine [7]. In 2011, underground and surface drill programs added new mine life with identification of additional ore bodies, which is anticipated to extend Jundee's mine plan beyond 2014 [6][3][13]. An excess of $19 million was spent on exploration in 2011 [3].

Open-cuts are being backfilled with waste rock, and tailings ore are being left to fill with water [10]. Closure of open pit mines at Jundee occurred as production moved to underground operations. This resulted in a large reduction in the volume of material mined at Jundee, especially waste rock [4]. The mine is currently producing gold from low-grade ore and continues to be one of the top 5 gold mines in Western Australia. In addition, specific actions were carried out in 2007 to reduce impacts on local and regional water bodies.

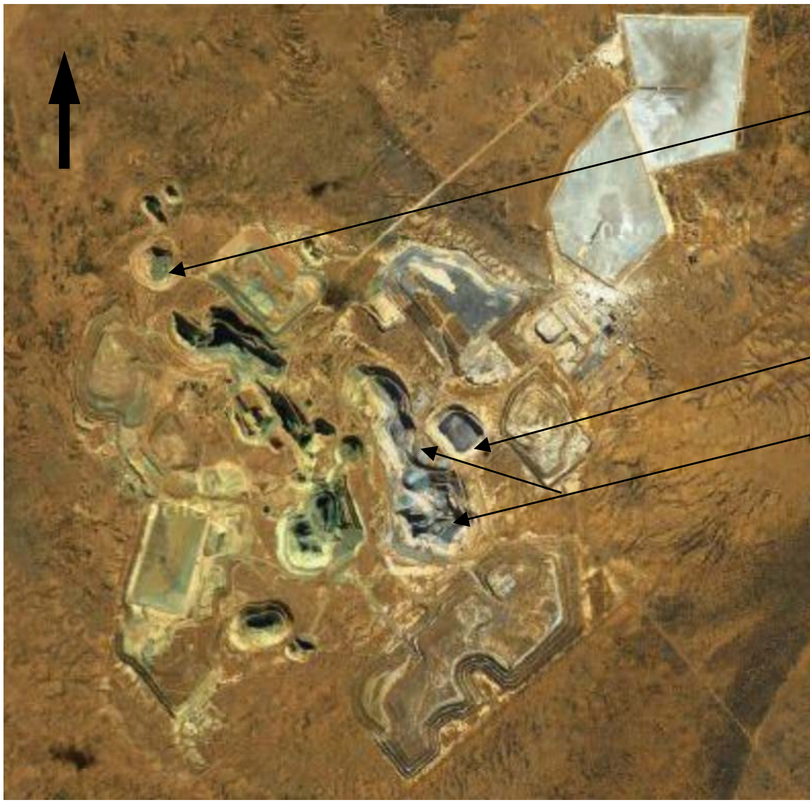
About 5.5 million ounces of gold has been mined at Jundee from 1995 to 2011 [3]. In 2006, the operation produced 313,000 oz. of gold and reported 1.48 million oz. of gold reserves at year-end, and produced 378,000 ounces of gold in 2008. It has proven and probable reserves of 1.3 million ounces [4]. In 2011, the mine had proven and probable reserves of 650,000 oz [3]. In 2009, Jundee produced 412,300 ounces of gold [11]. High-grade ore contributed to an overall reduction in cyanide use at Jundee of about 40 percent in 2008 [4] and a 23 percent reduction in cyanide consumption in 2011 due to a reduction in ore processed [3]. Table 7-1 lists the production overview for Jundee Mine.

Table 7-1 Production Overview for Jundee Mine [4][3]

Jundee	Average Ore (oz. /tonne)	Gold produced (000 oz.)	Revenue ($)	Revenue ($)	Material Mined (000 dry short tonnes)	Material Milled (000 dry short tonnes)
2007	0.174	291	298	253,247	7,436	1,827
2008	0.241	378	377	399,942	1,145	1,636
2009	0.270	411	413[1]	-		1,603
2010	0.231	335	416[1]	-		1,581
2011	0.259	333	529[1]	-		1,322

1-In US $millions

Figure 7-2 Aerial View of Jundee Mine (Google)

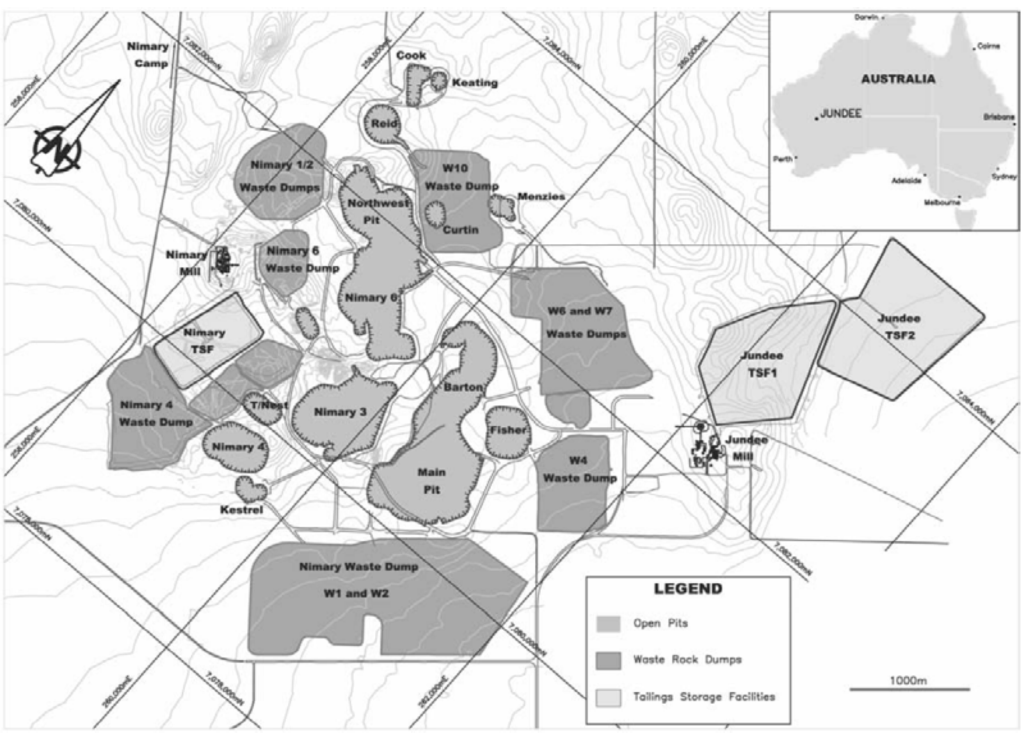


Reid Pit – filled with waste rock

Fisher Pit – filled with tailings

Barton Pit and Main Pit – filled with waste rock

Figure 7-3 Location Map and Site Plan [10]



PIT DESCRIPTION

Host mineralisation identified by XRD consisted of albite, anyhdrite, ankerite, aragonite, bassanite, calcite, clinochlore, gypsum, halite, hematite, hexahydrite, kaolinite, kieserite, microcline, muscovite, quartz, and siderite. Secondary minerals identified on many of the pit walls consist of a mixture dominated by hexahydrite and halite [2].

Generally, the gold ore occurs within shear-hosted quartz-carbonate veins and in the surrounding altered and strained wall rocks (including basalt, dolerite, dacitic volcaniclastics, carbonaceous shale and dacitic porphyry). The larger deposits on the site were developed in, or between, areas where two or more shear corridors intersect exhibiting strong structural control [10].

Hydrothermal alteration associated with Au mineralization is laterally zoned and overprints the metamorphic phases. The alteration zones are tangentially not extensive, generally less than ~ 10 m from the mineralization but can be up to 25 m. A distal alteration zone is dominated by a chlorite-quartz-calcite alteration assemblage and the proximal alteration assemblage consists of a quartz-ankerite-calcite-muscovite-pyrite-arsenopyrite (-chlorite-rutile-leucoxene) [10].

The entire thickness of the mine sequence is about 2.2 km and is dominated by two basalt units that contain massive and pillowed flows that are locally amygdaloidal and variolitic. Dacitic intrusives that are aphanitic to weakly porphyritic and grey-green in color with plagioclase (-hornblende-quartz), host significant primary gold mineralization in several of the deposits. Late stage porphyritic granodiorite dikes are also prevalent. The entire sequence has been metamorphosed to a greenschist facies assemblage with partial to complete replacement of primary pyroxene by actinolite and igneous plagioclase by albite [10].

The regolith overlays the crystalline rocks with complete oxidation as observed in drill cuttings ranging in depths from 40 to 120 m. The oxide zone is a combination of two clays; red mafic-rock-derived clay and purple ashen clay, with oxidised relict silicified fragmental material. There is a transitional zone, approximately 15 m thick consisting of partially oxidized, silicified material. Below the transitional zone, occasional isolated fractures with oxidised surfaces extend further into the basement or primary rock [10].

The primary sulphides in the waste rock and wall rock are pyrite and arsenopyrite. Carbonates are extensive and in the form of calcite and ankerite [10].

Figure 7-4 Jundee Pit [4]

Figure 7-5 Jundee Pit [8]

STRATEGIC USE OF PITS

Robertson GeoConsultants Inc., in partnership with O'Kane Consultants Inc., carried out a waste rock characterization program and determined the implications of pit backfilling with waste rock on pit water quality for Newmont Jundee Operations. The work included a waste rock sampling program and the design of a laboratory testing program. Recommendations were made for mine waste management and on the suitability of materials used for rehabilitation works [8].

In 2004, Robertson GeoConsultants Inc. completed a comprehensive hydrogeological review of the site and an assessment of current and potential future impacts of mining operations on the local groundwater system. The review also provided recommendations for minimizing future impacts on the groundwater during operations and also outlined future hydrogeological studies that could assist in closure planning. The work included development of a 2D seepage model (HYDRUS-2D), development & calibration of a 3D ground water flow model (MODFLOW), design and supervision of waste rock characterization program, mine waste management strategies, hydrogeological review and assessment, review of ground water monitoring plan [8].

MATERIALS PLACED IN PIT AND METHODS OF PLACEMENT

Jundee sources water from groundwater wells, the underground mining operation and from mine pit water collection. Newmont recycles as much water as possible within the processing portion of the operation, reducing the need to pump and use groundwater [7]. In 2007, approximately 3500 ML of water was withdrawn from the environment but in 2008 it was 1,500 ML [4]. To reduce process water consumption, they designed and installed a seepage interception system around the tailings impoundments to allow water recovery back to the processing plant [7].

There is no discharge of process or tailings water at Jundee. The Fisher in-pit facility is unlined, but has several recovery bore sites down gradient to intercept any seepage [1]. In 2007, the Fisher Pit Recovery Bores where re-drilled to increase potential water recovery from the Fisher Pit. Jundee incorporated a management practice of bore rotation in the village potable and mine water supply system to ensure specific areas of the aquifers are not being over-stressed [7].

Figure 7-6 Fisher Pit Partially Filled with Tailings (R. Hague, 2005)

PERFORMANCE OF IN-PIT DISPOSAL

The Department of Environment and Conservation has established a ground water quality standard of 0.5 mg/l WAD cyanide at Jundee. Monitoring results show WAD cyanide levels below 0.04 mg/l [1]. Jundee was recertified in 2012 with full compliance with the International Cyanide Management Code (ICMC) [12]. Salinity values reported in the groundwater data associated with the open pits from exploration drilling, range from 3470 to 4510 mg/L, but water quality data for groundwater associated with the open pits is limited [10].

The best estimates of pit water quality suggest a slightly alkaline, saline pit lake with elevated TDS values dominated by Na, Cl- and SO_4^{-2}, with lesser amounts of Ca, Mg, HCO^{3-} and NO^{3-} and negligible metals with the possible exception of Cd, Co, and Ni +/- As. Evaporation is anticipated to play a fairly dominant role in the chemistry within the pit lake system given the climate at Jundee. This will likely result in gradual increases in concentration of the constituents within the lake system and potentially the development of a density stratified lake 'trapping' the poorer water quality at depths in the form of a stratified hydraulic cage post closure [10].

Within the pits themselves, much of the backfill material (primarily waste rock) and primary pit wall rock contained substantial neutralization in the form of carbonate veining. It is expected that these materials will add net alkalinity to the pit systems. The assessment of hydrogeological work conducted by others indicates that a cone of depression will remain towards the open pits and underground workings post mining. As such, there is little anticipated risk to regional groundwater from interactions with pit waters as the water will remain 'trapped' in the local depression, or 'hydraulic cage'. The groundwater table will however likely rebound to some level resulting in standing water in the open pits. The water quality in these 'lakes' will be dependent on a number of factors:

- The water quality and rate of the rebounding groundwater;
- Interactions of the rising water with the pit walls;
- Precipitation and run-on (over exposed pit walls);
- Solubility constraints due to secondary mineral precipitation; and
- Evaporation (i.e., concentration due to evaporation). [10]

The characterization of solid waste material of the site indicates that there are only very minor, localized areas on the site that show a depressed pH (i.e., acidity) and the potential for acid formation (only 5% of samples tested had a pH value less than 5 [2]). The vast majority of material on site is not acidic and is considered to be non-acid forming. There are indications of neutral metal leaching in particular areas on site and electrical conductivity (salinity) indicating that there are stored soluble constituents in much of the material, albeit largely non-acidic. Seepage indicators occurred where moisture contents were higher near pit wall seeps (through fractures, cracks and drill holes), where secondary mineral crusts had formed [10].

The results for the seep and pit pond water samples ranged in pH from 6.7 to 8.0 and EC values ranged from 5.3 to >20 mS/cm (note 20 mS/cm is the upper limit of measurement for the field meters utilized). In general the pit seep and pond waters were well buffered with moderate to high conductivities and similar to the groundwater values available on record [2].

Acid neutralisation capacity (ANC) values ranged from <1 to 360 kg $CaCO_3$/t equivalent. The average was calculated to be 63 kg $CaCO_3$/t equivalent with a standard deviation of 89 kg $CaCO_3$/t equivalent. The majority of samples reported relatively low ANC values; however 24% of the samples (33 samples) had ANC values greater than 100 kg $CaCO_3$/t equivalent. The samples with higher ANC values are all primary material, most often with carbonate veining evident and largely identified as basalt [2].

By and large, most of the material sampled plotted above the 2:1 ANC:MPA (maximum potential acidity) line in the non-acid forming (NAF) zone. A number of samples cluster right at the apex of the figure (i.e., have low ANC and MPA values) and a few plot well within the potentially acid forming (PAF) field (i.e., elevated MPA values and negligible ANC values). Of those that plot within the PAF field, the majority are from two specific dumps and one stockpile that reported higher sulfide values. The ANC/MPA ratios ranged from 0 to over 1000 with an average of 35 and a standard deviation of 103. Twenty-five samples (~18%) reported ratios less than 2:1 [2]. The trends can be seen in Figure 7-7.

The Net Neutralization Potential (ANC minus MPA; or NNP) values for these samples ranged from –70 to 360 kg $CaCO_3$/t equivalent (as seen in Figure 7-8), with an average of 56 and a standard deviation of 87 kg $CaCO_3$/t equivalent in the dataset. Fifteen samples or 11% of the dataset reported negative NNP values (or greater MPA than ANC) [2].

Figure 7-7 Maximum Potential Acidity (MPA) versus the Acid Neutralisation Capacity (ANC) [2]

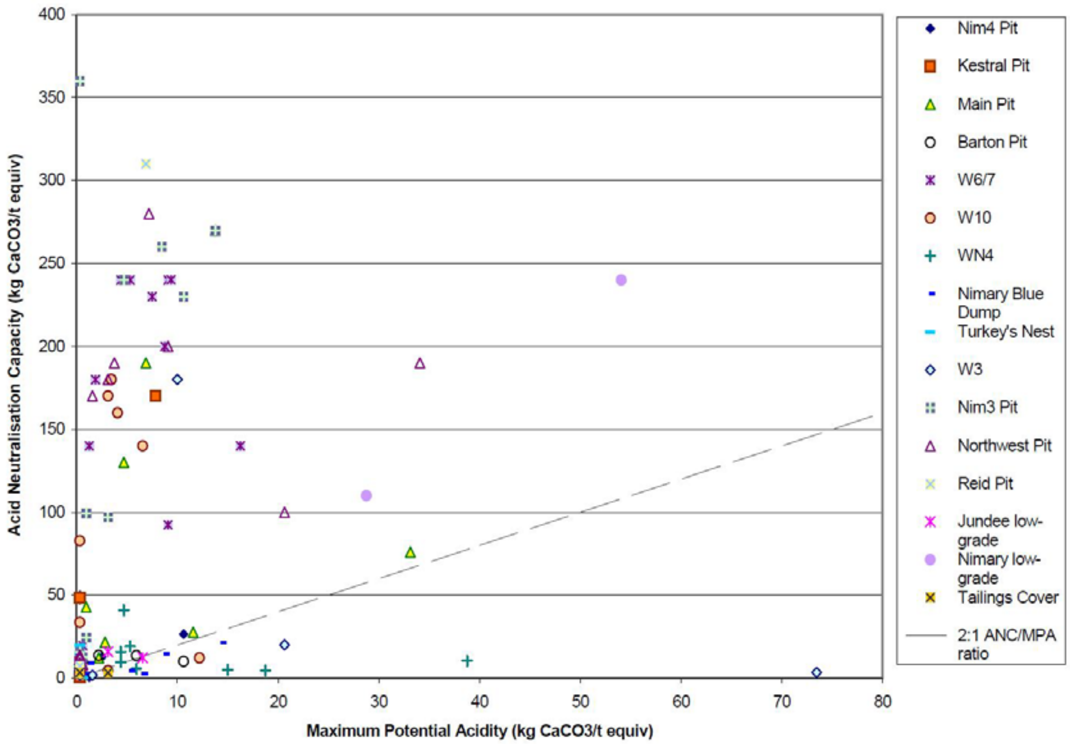
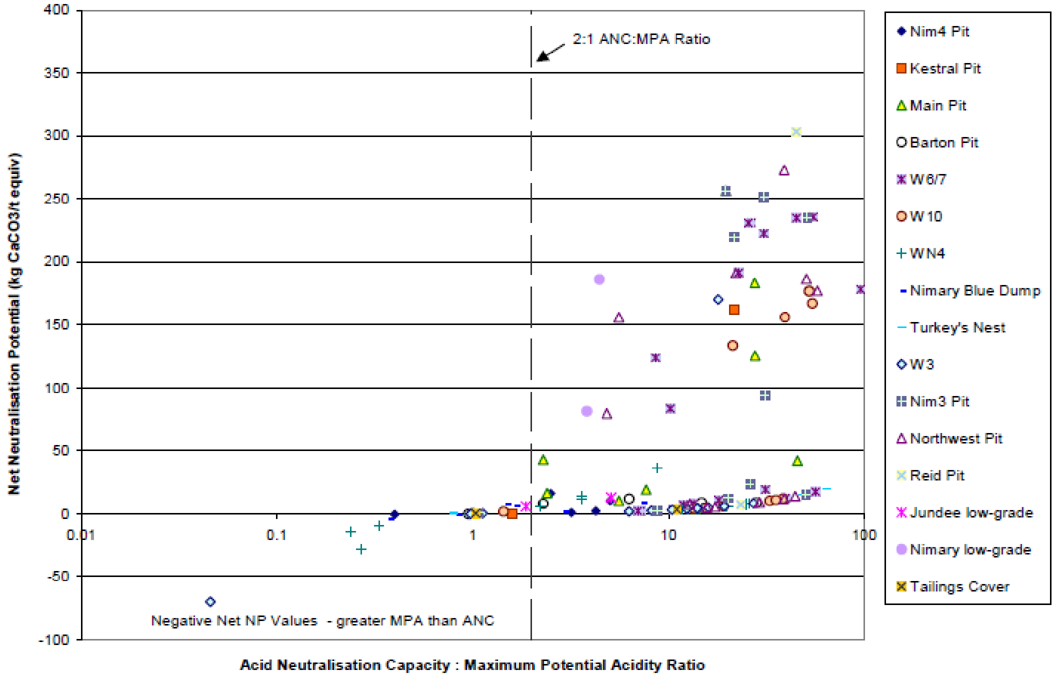


Figure 7-8 Ratio of Acid Neutralisation Capacity (ANC) to Maximum Potential Acidity (MPA) versus the Net Neutralisation Potential (NNP) [2]



REMEDIAL

In line with Newmont's approach to environmental stewardship and social responsibility, Jundee has a well-advanced closure plan and has already committed to an ongoing work program that encompasses both closure and progressive rehabilitation at several exhausted open pits and the recently decommissioned Nimary operating facilities [4].

In 2009, work on the rehabilitation program surrounding the immediate mine area continued, as well as four major segments of the southern open pit mining area. Rehabilitation plans are made well into the future, with the objective that all available mine rehabilitation is completed before ultimate mine closure [7][4].

BENEFITS and LESSONS OF IN-PIT DISPOSAL AT JUNDEE

The Jundee facility has successfully taken advantage of mined out pits for tailings and waste rock disposal. The Jundee operation was audited in late 2008 and was conditionally certified to meet state and company performance criteria. In 2012, the mine was recertified in compliance with the ICMC.

RESIDUAL ISSUES

Other than long term monitoring of water quality in the pits and in the surrounding groundwater, no long term requirements are reported.

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CASE STUDY #8 MARYMIA MINE

SYNOPSIS

The Marymia Mine is located in Western Australia in the Murchison Desert. Mining began in 1992 by Resolute Resources and operation stopped in 1998. Currently, the mine is owned by Barrick Gold through Dampier Gold, and there are plans to resume operations, although there is no current mining of the larger of two deposits (K1 deposit). The smaller mined out K1SE pit was continuously used to deposit tailings from the Marwest pit from December 1995 until October 1996 and then intermittently from August 1997 to November 1997. The mined out K1 pit was used intermittently from June to November 1997 for tailings deposition.

A total of 1,027,688 tonnes were deposited in the K1SE Pit, and 268,406 tonnes were deposited in the K1 pit by the use of spigots. The groundwater quality results from the monitoring bores surrounding K1SE have remained within the Department of Environmental Protection (DEP) license values: pH has remained between 7 and 8; the weak acid drainage of cyanide (WAD CN) has remained below 0.5 mg/L; and the total dissolved solids have remained below 1000 mg/L (although there was a spike in the southern end of the pit in 1997). Currently, the Marymia Mine is in care and maintenance mode.

Tailings at Marymia contained fibrous minerals that were considered to be hazardous and the in-pit disposal with a water cover provided complete containment. Also, in-pit disposal reduced evaporation and resultant raw water demands in a very dry location.

LOCATION

The Marymia gold mine site is located approximately 1000 km northeast of Perth, and 200 km northeast of Meekatharra in Western Australia [1], in the North Goldfields of the Murchison Region. The climate is semi-arid to arid with an annual rainfall of 230 mm a year (generally due to tropical depressions or cyclone activities in the summer and autumn months) and an annual evaporation rate of 3800 mm per year [2]. The mine site topography is an area of low-relief alluvial plain with minor ephemeral waterways draining into Lake Gregory [1][2]. Figure 8-1 shows the location of Marymia Mine, Figure 8-2 shows the site layout for the mine and Figure 8-3 shows the three Marymia mine open pits: Marwest, K1 and K1SE.

Figure 8-1 Location of Marymia Mine in Western Australia [2]

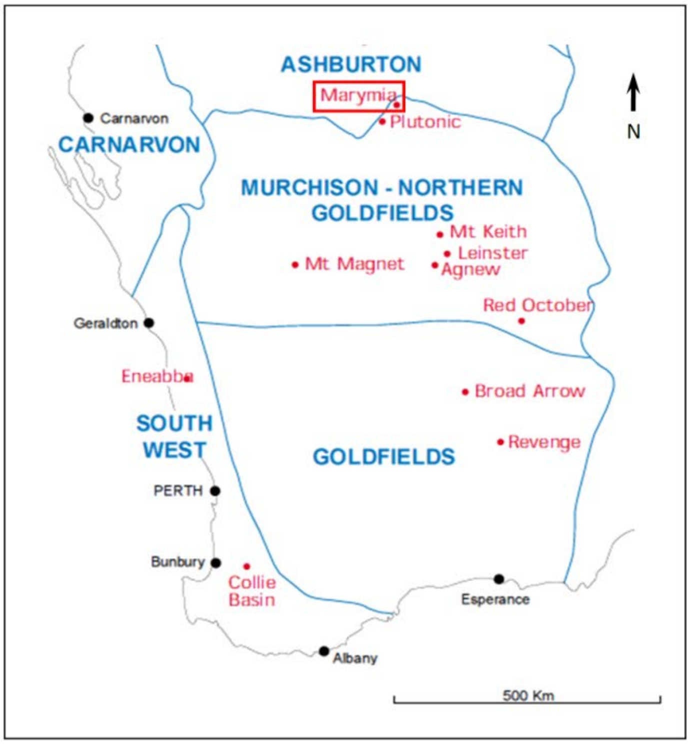


Figure 8-2 Marymia Mine Site Plan [2]

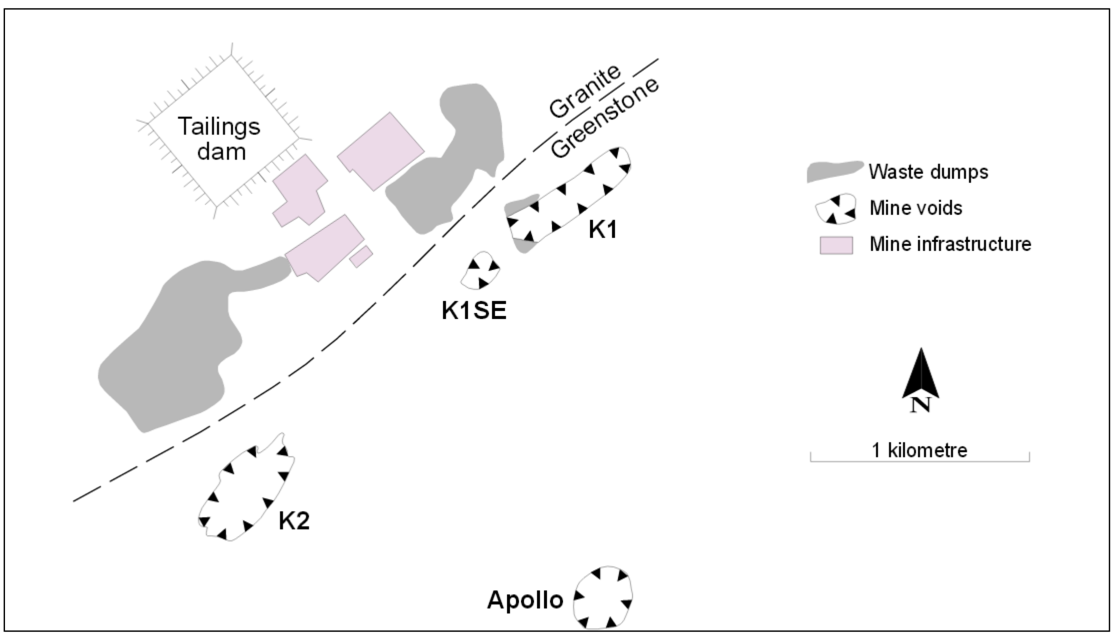
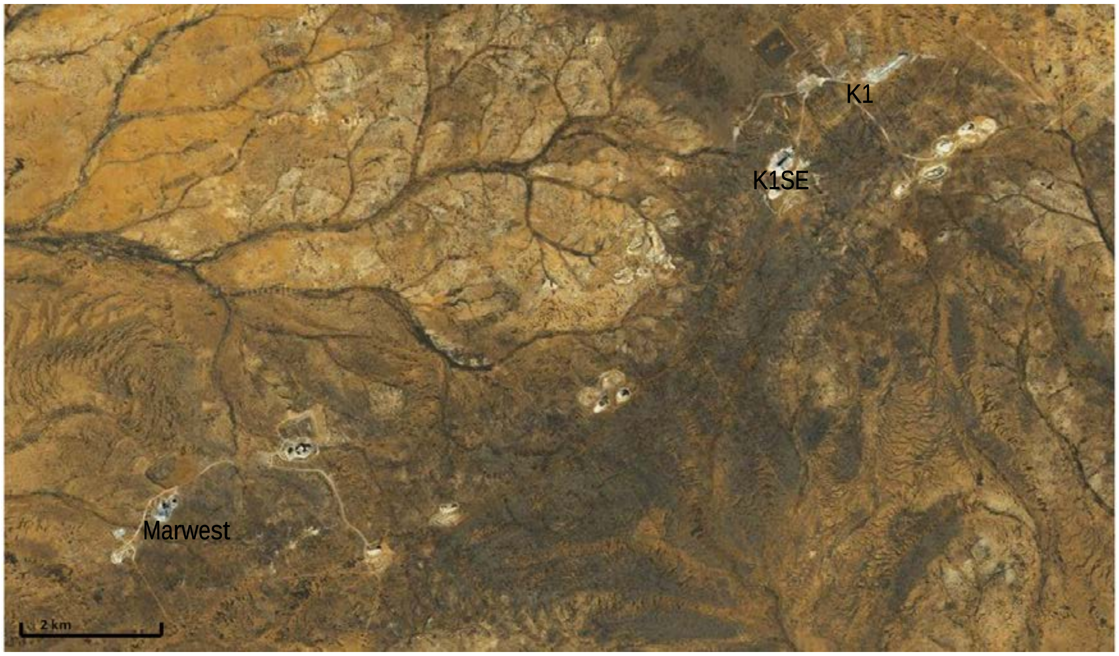


Figure 8-3 Location of Marwest, K1 and K1SE Pits (Google)



MINE TYPE and HISTORY

The Marymia Gold Mine operated from 1992 to 1998 under the ownership of Resolute Resources Ltd. The property is currently owned by Barrick Gold. Several known deposits are currently being considered for development [1][2]. Both the K1 and K2 deposits are currently held by Dampier Gold and based on drillings, results a two million ounce reserve grading between 1.9 g/t and 2.2 g/t Au [6].

The mine had a capacity of 650,000 tonnes ore per year [1]. Since the initial discovery in the late 1980s until the end of 1998, when the property was placed in care and maintenance, 10,637,362 grams of gold were produced by Marymia mine [5].

Tailings from the Marwest pit (12 km southwest of the K1 deposit) were placed in the mined-out K1SE pit starting in December 1995 and the K1 Pit starting in June 1997 with the permission of Department of Minerals and Energy of Western Australia (DMEWA) [1][2]. Tailings deposition operated continuously from December 1995 to October 1996, and then intermittently from August 1997 to November 1997 [1].

The cost of construction to handle tailings was $220,000 for the K1 Pit ($0.06 per tonne), and the cost of rehabilitation was $139,500 ($0.038 per dry tonne of tailings) [1].

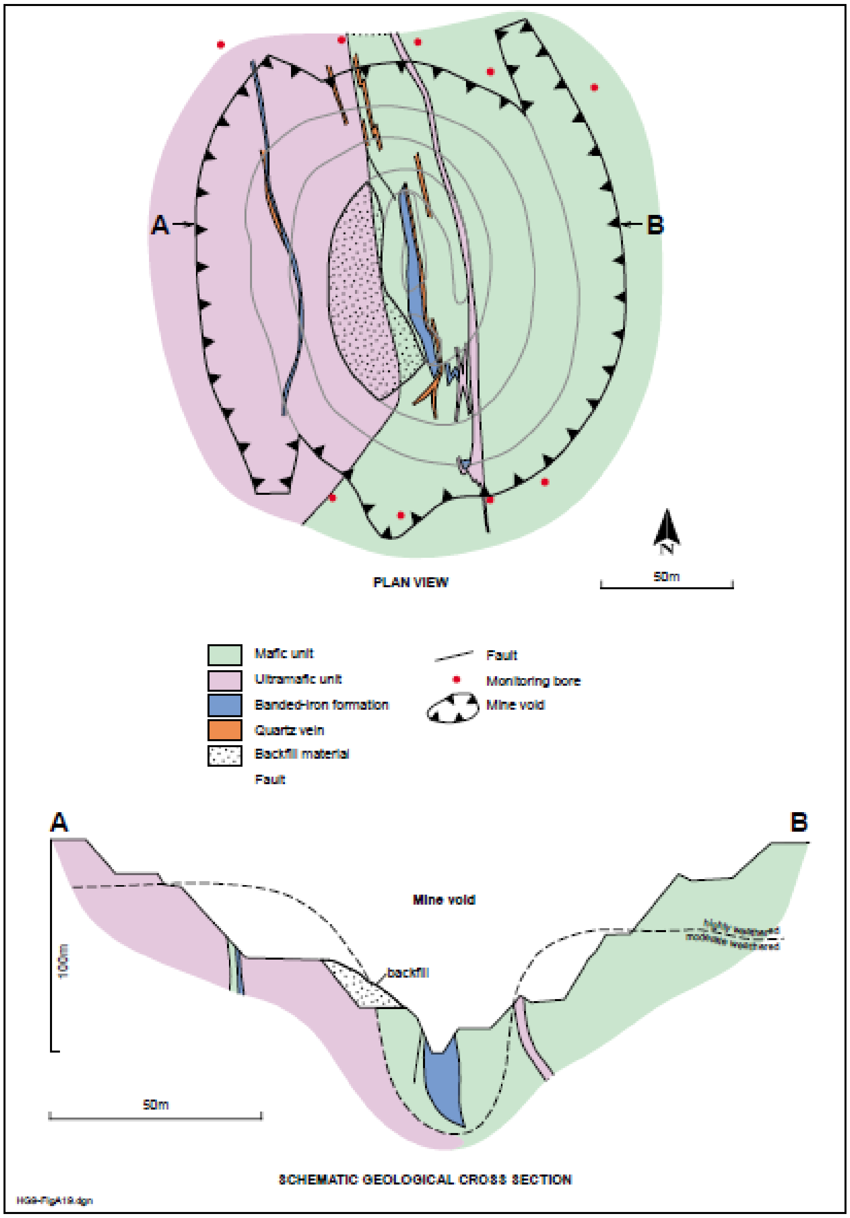
PIT DESCRIPTION

The final dimensions of the Marymia K1SE pit were 170 m by 150 m, by 50 m deep and it was located south west of the K1 pit [1]. The K1 Pit had a surface area of 93 000 m^2 and a capacity of 2.114 Mm^3. Inspection of the pits showed that no cracking (indicative of potential large-scale wall failure) was present around the pit crest, haul road or berms. Tests also confirmed the durability of the wall materials when subjected to cycles of wetting and drying. Some minor wall failure was anticipated during deposition, as tailings and ponding water against the highly weathered clayey material was expected to result in strength loss due to material softening [2]. Neither pit showed signs of stress and the slope of the pit walls were shallow.

Both the K1SE pit and the K1 pit intersected four major geological units during mining: a highly sheared talc chlorite schist; mafic amphibolites; banded iron formation (BIF, hosting the mineralization); and massive hangingwall mafic rocks. Shearing along lithological contacts is common with a pervasive foliation, subparallel to the shearing, within the talc chlorite schist [1]. The hosts at Marymia comprise rheologically and geochemically heterogenous and metamorphosed komattiitic, tholeiitic and sedimentary rocks, with lithological contacts invariably being structural [3][6]. K1SE pit geology had a shearing-foliation dipped 70^o-75^o towards 218^o-232^o magnetic with joint sets and slip fault. The foliation in the K1 Pit generally dipped between 60^o to 85^o to the east [1]. Figure 8-4 shows the geology of the K1SE mine void, including the split between the mafic and ultramafic units.

The Plutonic Greenstone Belt is a north-easterly trending greenstone sequence, has a strike length of about 50 km and comprises sheared foliated ultramafic volcanics, tholeiitic basalt and metasedimentary (talc chloride schist) rocks especially on the eastern portion [1][2]. The Marymia Inlier has been subjected to additional Palaeoproterozoic periods of deformation and metamorphism, making it predominately brittle, resulting in faulting, thrusting and shearing [6].

Figure 8-4 Geology of K1SE Mine Void [2]



Groundwater was only encountered in the BIF towards the base of the pit and the water table is about 50 meters below ground level (no major dewatering was required during mining). Groundwater salinity was generally less than 1500 mg/L total dissolved solids (TDS) [2]. The water level in the base of the pit prior to commencement of the tailings underdrainage construction was estimated to be Reference Level (RL) 593 [1]. Table 8-1 lists the water levels and groundwater levels for both the K1SE and the K1 pit.

Table 8-1 Water Levels in K1SE and K1 Pits [1]

Pit	Water Level in Pit Base	Regional Groundwater Levels	Comments
K1SE	RL596	RL 586.7	~56 m below ground level and well below the K1SE pit base
K1	RL595	RL 586.7	~53 m below ground level and well below the K1 pit base

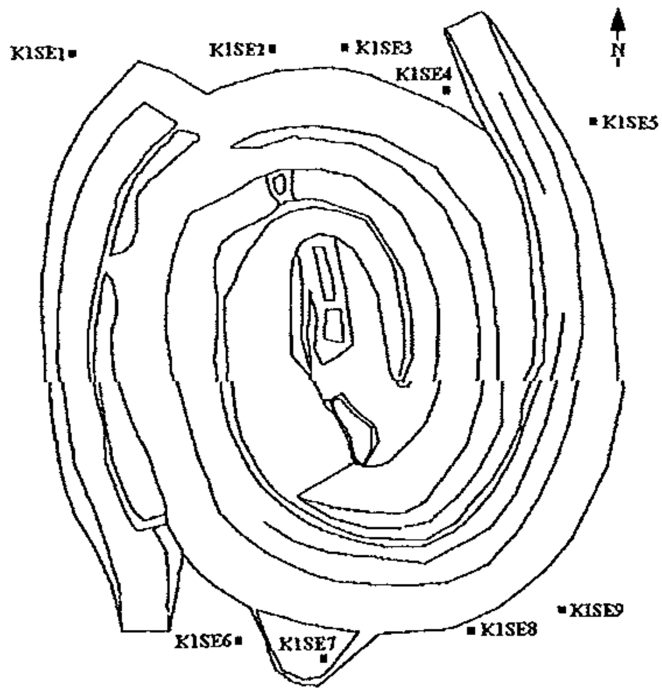
STRATEGIC USE OF PIT

Tailings from processing ore from the Marwest pit were placed into the mined out K1SE pit in 1994 -1995 as was approved by DMEWA [1]. The ore contained fibrous actinolite and tremolite and the resultant tailings dictated containment in a sealed environment [2]. The DMEWA had stipulated that a number of factors be addressed, including the stability of the pit, the impact of the tailings on the area hydrology, water recovery from the tailings, final density of the tailings and rehabilitation after the pit was back filled [1]. Resolute Mines collected data for K1SE pit first, and if successful, the K1 pit would then be assessed for use [1]. The shallow sloped pit walls were examined for kinematic modes of instability; however, geotechnical inspection showed no cracks present around the pit crest, haul road or berms [1][2]. The deposition into K1SE Pit was successful and Resolute Mining decided to follow the same process for K1 pit. Approval was obtained in late 1996. The pit storage operated intermittently from June 1997 – November 1997 then continuously until February 1998 [1].

The final dry density and water recovery characteristics of the tailings were assessed to determine the preferential mode of tailing deposition. The maximum amount of water available for recovery (52%) was available within 24 hours of deposition, provided that tailings velocity was low in order to minimise turbulence. Water release would be rapid allowing for water recovery via a decant system, eliminating the need for an underdrainage system [2].

The high water recovery indicated that only a pontoon-mounted pump for water return was required. Discharge of the tailings was from a single outlet initially from the haul road and later from the pit rim. A set of nine groundwater monitoring bores were constructed to the north and south of the pit within the expected flow path (Fig. A19). The bores were designed for the multiple purposes of monitoring and recovery (if tailing water was affecting groundwater resources) [2]. Figure 8-5 shows the locations of the groundwater monitoring bores for the K1SE Pit. The monitoring bores were designed so that in the event that monitoring indicated the tailings water was having a detrimental effect on the groundwater, the monitoring bores could be used for water recovery with submersible pumps [1].

Figure 8-5 Groundwater Monitoring Bore Locations of K1SE Pit [1]



MATERIALS PLACED IN PIT AND METHODS OF PLACEMENT

1,027,688 tonnes were deposited in the K1SE Pit, and 268,406 tonnes were deposited in the K1 Pit [1]. Table 8-2 lists the tonnes of tailings deposited in the pits for storage.

Table 8-2 Distribution of Tailings into Pits for Storage (December 1995 to January 1998) [1]

Tailings Storage Facility	Date of Operation		Tonnes of Tailings Deposited
K1SE Pit	15 December 1995	4 October 1996	483,589
	16 August 1997	17 August 1997	3,650
	20 August 1997	26 August 1997	11,692
	6 November 1997	14 November 1997	14,913
K1 Pit	6 June 1997	23 June 1997	29,549
	16 July 1997	25 July 1997	14,991
	2 August 1997	3 August 1997	3,544
	6 August 1997	7 August 1997	3,055
	27 August 1997	5 November 1997	116,230
	15 November 1997	14 January 1998	101,037

Waste from the mining of the K1SE Pit was dumped in the southern part of the K1 Pit. Small quantities of waste from the mining of the other small pits have been dumped in the northern part of the K1 Pit along the western wall and in the north east corner of the K1 Pit. However, the quantities of waste in the northern part of the pit were not considered to be significant [1].

The spigot technique, size and positions were evaluated to minimize the potential for failure, to provide buttress support as the tailings and supernatant pond rose, a high beach above the supernatant pond was formed by spigotting at the toe of the waste in the southern section of the pit [1]. Tailings were discharged from an open-end line directed onto the ground above the supernatant water level, which enabled the tailings to enter the pond at low velocity. Spigotting of tailings was carried out from a number of different points around the pit in an attempt to develop a uniform tailings surface [1]. The return-water volumes, as a percentage of water in the tailings slurry, ranged from 5% at the start of in-pit disposal to 74% after four months. Two months after operations ceased, the average dry density of the tailings was 1.4 g/cm^3 confirming that the final dry density prediction of 1.64 g/cm^3 was achieved [2].

No major wall failures were observed in the discharge system during tailings disposal. The west wall of the K1SE Pit had minor failures during operations and therefore spigotting was not used (it was done in the Northwest section with spigot discharge from the haul road at RL 615 and then from the berm at RL 635 eventually covering the haul road) [1]. Minor wall failings from the weathered clayey materials may have occurred during deposition (the wetted front above the supernatant extended for a height of 100 mm-200 mm).

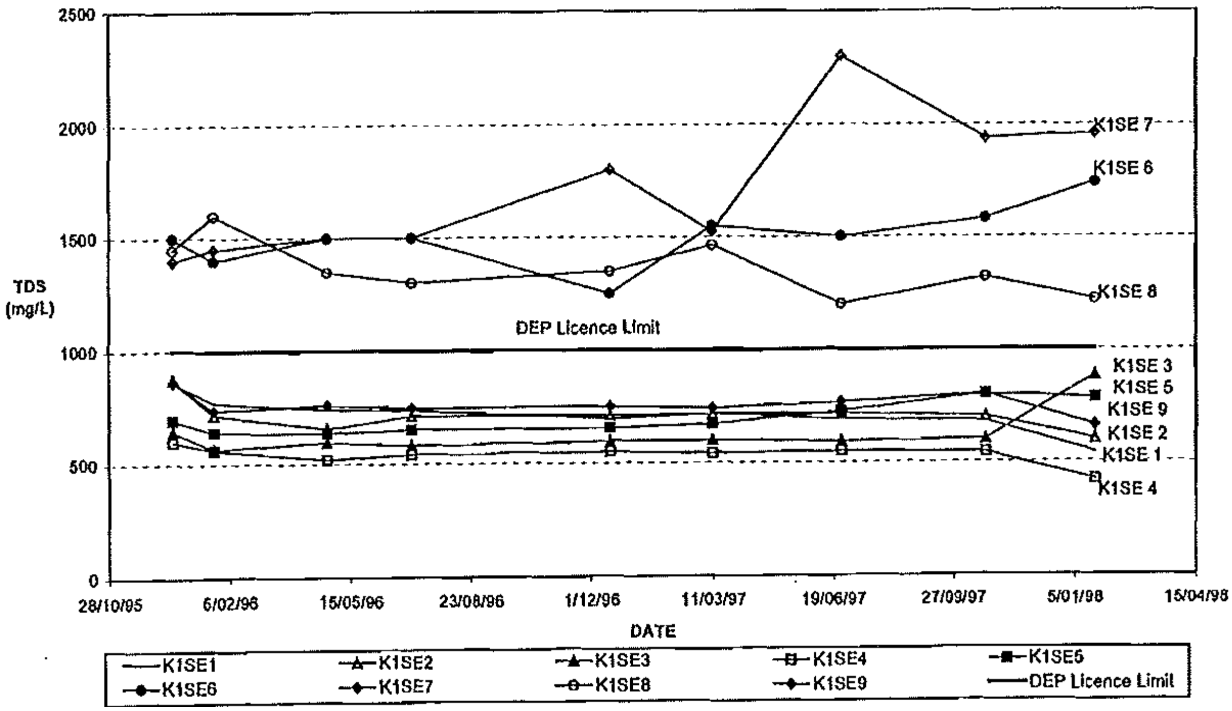
In the first eight months of in pit tailing storage the tailings rose from RL595 to RL638. The wall material exposed to the wet-dry cycle is located between RL638 and RL643 [1].

PERFORMANCE OF IN-PIT DISPOSAL

Inspections of the pit were done on January 12 and June 21 1996; the water recovery system was deemed successful as there was a shallow layer of supernatant water covering 233,345 m^3 of tailings [1]. The retention of water in the pits was 46.0% and 28.8% for the K1SE and K1 Pits respectively. There were 292,834 tonnes deposited into K1SE resulting in an in-situ density of 1.255 t/m^3; therefore, an average dry density of 1.6 t/m^3 from the consolidation testing was achieved [1].

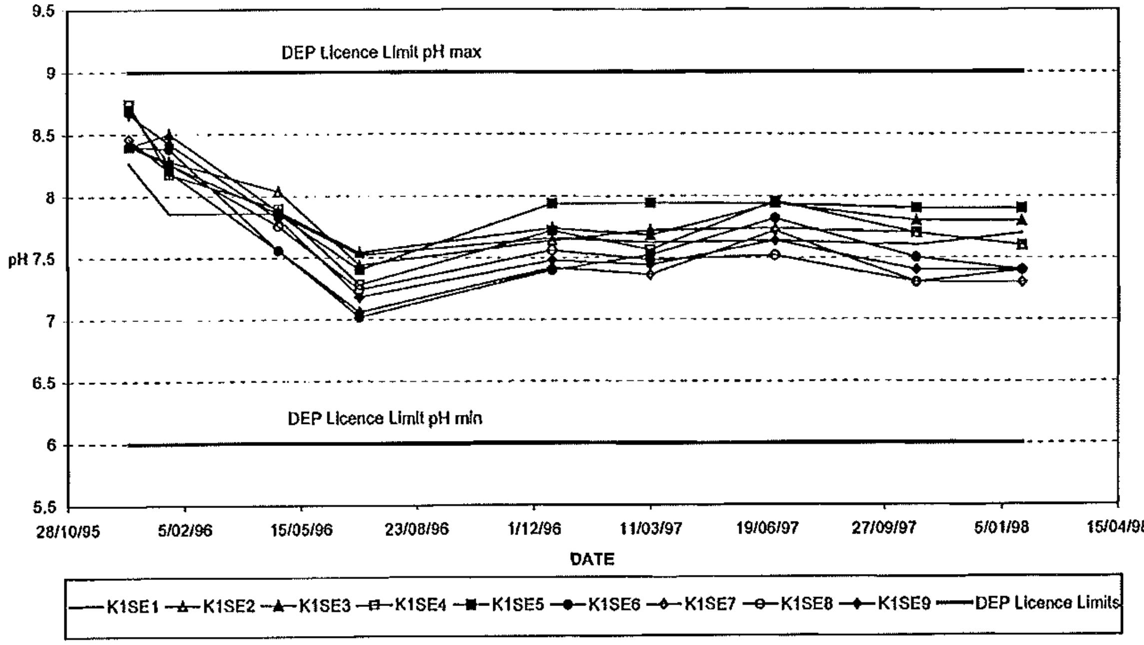
Groundwater quality was tested and determined to be within DMEWA limits except for the TDS from the bore on the east side of the pit (they were higher than the limits set by DMEWA) [1]. Groundwater monitoring showed rising water levels in all bores resulting from tailings deposition, in particular along the shear zone that passes through the pit in a north–south direction. Rising groundwater salinity (about 3000 mg/L TDS) and cyanide (CN) concentrations (about 120 mg/L) were observed. Hazardous asbestos-type material was stored wet beneath a constant cover of water, until finally covered by fresh earth on completion of the tailings operation [2]. Figure 8-6 shows the TDS levels for the groundwater monitoring bores for K1SE and the levels for K1SE 6, 7 and 8 (all along the southern edge of the pit) have levels remaining above the Department of Environmental Protection (DEP) License Limit.

Figure 8-6 Total Dissolved Solids (TDS) from Groundwater Monitoring Bores for K1SE Pit [1]



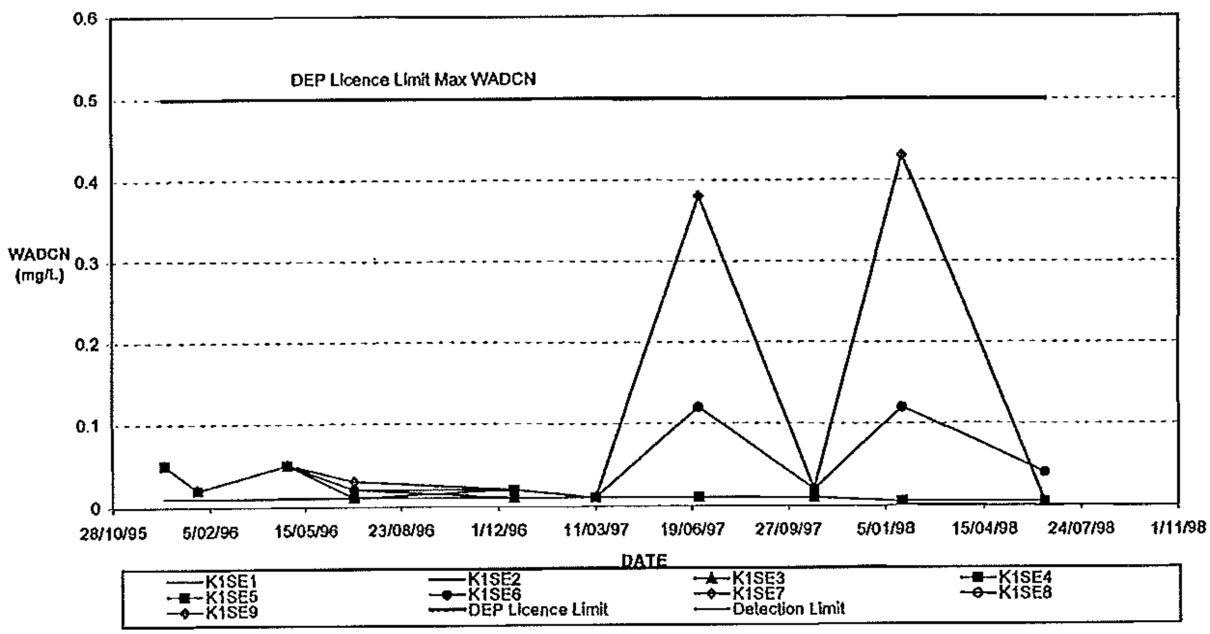
The pH levels in the K1SE pit decreased from November 1995 to June 1996 and have remained consistent. The levels have remained between the DEP Limit maximum and minimum, and since July 1996 have remained between a pH of 7 and 8. Figure 8-7 shows the pH values from the groundwater monitoring bores at the K1SE Pit from October 1995 to April 1998.

Figure 8-7 pH Values of the Groundwater Monitoring Bores at K1SE Pit [1]



The weak acid drainage of CN (WADCN) remained below 0.1 m/L during the initial tailings deposition from November 1995 to November 1997, but spiked in June 1997 and May 1998 at the southern end of the pit (where tailings were being deposited at the time). The levels remained below the DEP License maximum limits. Figure 8-8 shows the WADCN levels of the groundwater monitoring bores for the K1SE Pit.

Figure 8-8 WADCN Levels from the Groundwater Monitoring Bores at the K1SE Pit [1]



REMEDIAL

No remedial actions have been reported to be required to deal with groundwater contamination or pit wall instability.

BENEFITS and LESSONS OF IN-PIT DISPOSAL

The in-pit tailings disposal at Marymia proved to be extremely cost-effective, with considerable savings in capital and operational costs, compared with above-ground tailings storage (TSF) option. The principle benefits for tailings disposal into K1SE pit were environmental including, safe storage of asbestos fibre-containing mine wastes, no necessity for land clearing for a surface tailings facility. Addition benefits included improved site aesthetics and reduction in raw-water requirements [2].

RESIDUAL ISSUES

None.

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CASE STUDY #9 RABBIT LAKE MINE

SYNOPSIS

The Cameco Corporation Rabbit Lake Operation is located approximately 800 km north of Saskatoon, Saskatchewan in the uranium-rich Athabasca Basin, an area characterized by a cool moist climate and a short annual frost free period of just 90 days. Gulf Minerals Ltd. operated the mine from 1975 to 1984, but the mine has since been owned by Cameco Corporation and its predecessor companies. The mine has produced more than 190 million pounds of uranium[22] from five different ore bodies at Rabbit Lake Mine: Rabbit Lake open pit, Collins Bay A, B, & D Zones (open pits), and Eagle Point (underground).

The Rabbit Lake pit was turned into the Rabbit Lake In-Pit Tailings Management Facility (RLITMF) after mining was completed in 1984. The pit has since been a tailings depository for the three other open pits and underground mining at the Rabbit Lake Operation. Tailings from these mines contain elevated levels of radioactivity as well as other metals. Metal and non metal (e.g., arsenic) leaching are attributes of these tailings; these aspects are controlled to a large extent by the precipitation of secondary minerals within the water treatment circuit in advance of in-pit disposal.

The RLITMF has been a pioneering development for in-pit disposal of mine wastes. The Rabbit Lake example is a combination of an engineered pervious surround and tailings characteristics engineered for long term management. The successful isolation of the tailings with their potentially mobile contaminants relies on the consolidation of the tailings into a low hydraulic conductivity mass surrounded by a high hydraulic conductivity envelope. During the operating phase, surrounding groundwater is collected in a raise well immediately adjacent to the pit – this creates a gradient that allows collection of tailings seepage water resulting from consolidation. The pervious surround consists of a gradational increase in grain size of the pit surround from a fine sand filter at the tailings interface, to larger crushed rock at the outer-most zone of the surround. This also allows for draining and collecting of the tailings supernatant and pore water.

More than eight million tonnes of tailings solids have been deposited in the RLITMF since the commencement of backfill in 1984. Initially, an attempt was made to deposit filtered tailings; however, since mid-1986, there has been direct discharge of a thickened tailings slurry material into the pit.

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[22] Uranium production is commonly measured in pounds of uranium oxide – U_3O_8. 190 million lbs of U_3O_8 is 73 million kg U.

Starting in 2000, the tailings were deposited by deep injection to prevent the development of seasonal ice and frozen layers in the RLITMF. In 2008, the capacity of the pit facility was increased through a Pit Crest Expansion excavating a portion of the adjacent, former Rabbit Lake – lakebed and increasing the final tailings height.

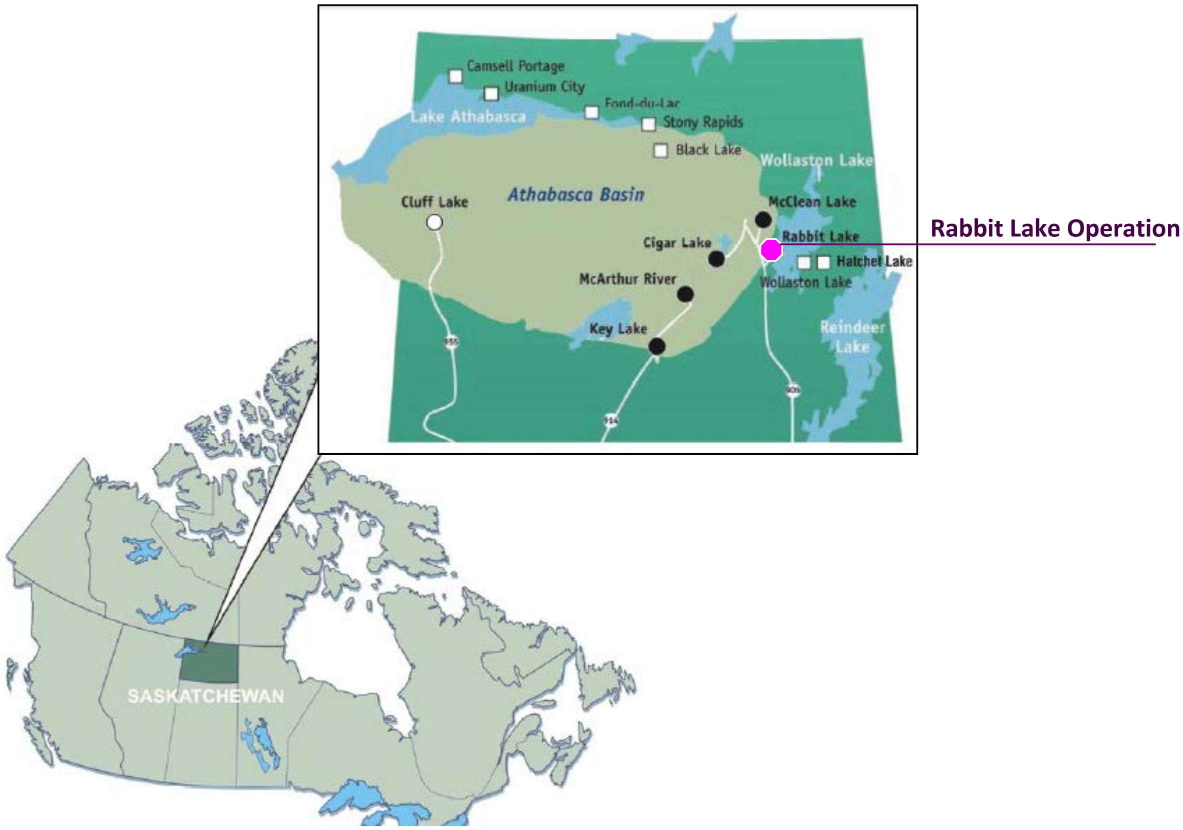
The RLITMF has successfully contained contaminants as evidenced by monitoring piezometers down gradient from the pit that shows the elements of potential concern to be in concentrations below the requirements of the Saskatchewan Ministry of the Environment (SMOE) permit conditions.

LOCATION

The Rabbit Lake Mine is near the west shore of Wollaston Lake, in northern Saskatchewan, approximately 800 km north of Saskatoon. It is on the northeastern edge of Canada's uranium rich Athabasca Basin, where the world's highest grade uranium resources have been discovered [7][1]. The mine site is located on a major continental drainage divide hosting two large river drainage systems to the north and south of the site with local streams draining into Wollaston Lake [4]. Figure 9-1 shows the location of Rabbit Lake Mine.

The summer season in the area of Rabbit Lake Mine is fairly short and host temperatures between 10ºC and 20ºC, and a frost-free period of less than 90 days [4]. The winters are cold with average lows around -28ºC. Average annual precipitation is 556 mm, with just over half of the year's precipitation seen between the months from June through September [21]. Approximately, 60% of the annual precipitation falls as rain [4].

Figure 9-1 Rabbit Lake and the Athabasca Basin, Saskatchewan [4]



MINE TYPE AND HISTORY

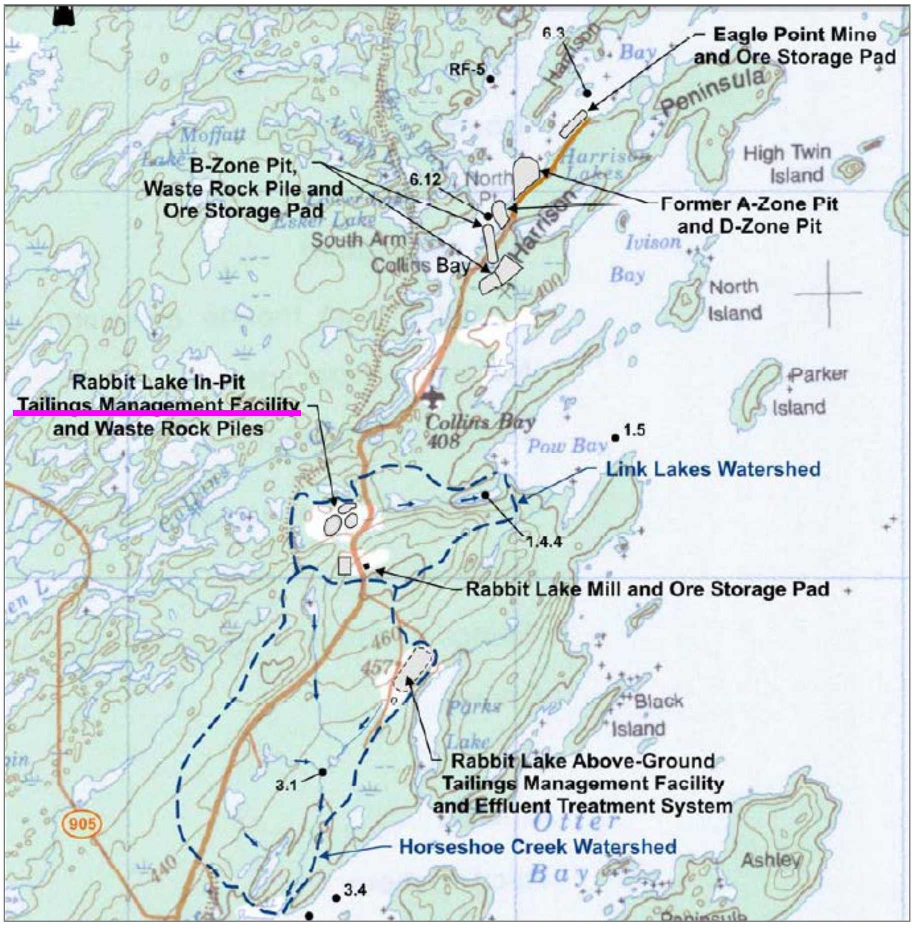
The Rabbit Lake uranium deposits were first discovered in 1968 by Gulf Minerals Ltd. and Uranerz Exploration and Mining Limited. Gulf operated the mine from 1975 to 1981 - in 1981 Eldorado Nuclear acquired the property. In 1988, Eldorado and Saskatchewan Mining Development Corporation amalgamated to form the Cameco Corporation - the current owner of the Rabbit Lake Mine.

The Rabbit Lake Mine is located in the Canadian Shield. Locally, granites are overlain by Aphebian Wollaston Group sediments metamorphosed to the amphibolite facies. Host rock types include biotite-quartz-feldspar gneiss, quartz feldspar gneiss, and quartzite [2].

The Rabbit Lake Mine is the longest operating uranium production facility in Canada, and has a maximum annual production capacity of 4.25 million kg of uranium concentrate. By 2013, 190 million pounds of U_3O_8[23] had been produced from the five ore bodies at Rabbit Lake- Rabbit Lake open pit, Collins Bay A, B, & D Zones, and Eagle Point (Figure 9-2) [26].

[23] Rabbit Lake production is equivalent to the production of all 10 mines in the Elliot Lake Ontario camp 1957 to 1996.

Figure 9-2 Rabbit Lake Mine Site and In-Pit Tailings Management Facility [4]



The Rabbit Lake open pit operated between 1975 and November 1984, and upon pit mine closure was converted into the Rabbit Lake In-pit Tailings Management Facility (RLITMF – the "Facility") [2]. Along Collins Bay, the B Zone ore body was mined from 1984 to 1991, the D Zone was mined 1995 to 1996, the A Zone was mined from 1996 to 1997, and the Eagle Point mine commenced operation in 1994 and continues at present (2014). All ore produced at the Rabbit Lake site has been processed at the Rabbit Lake mill [4].

Tailings produced from the Rabbit Lake open pit were deposited and remain in the Rabbit Lake Above-Ground Tailings Management Facility (AGTMF) (Figure 9-2). Rabbit Lake tailings are a heterogeneous mix of sandy, silty and clayey sizes material (maximum particle size of 0.425 mm [20][25]. The ore processing has included grinding, acid leaching and solvent extraction concentration of uranium [16].

The Rabbit Lake pit has been a tailings depository for tailings produced from three open pits and one underground mine at the Rabbit Lake Operation since 1984. The tailings from these other ore sources differ with variations in the levels of metal sulphides, arsenides and their oxidation products, as well as residual uranium and other radionuclides. Some of the tailings have the potential to leach metals, particularly nickel and arsenic, and all contain radionuclides.

Uranium tailings, particularly from high grade uranium ores in Saskatchewan have been subject to stringent containment requirements driven in large measure by strong regulation by both provincial and federal governments. In Saskatchewan, in-pit disposal is considered the best disposal technology for uranium tailings. This technology has been accepted as best to permanently isolate the tailings and prevent release of leachable components.

At Rabbit Lake, consideration was given to expanding surface disposal, but the Rabbit Lake open pit was selected based on several factors including proximity to the mill, and new technology outlined by Geocon, 1988 that would permanently isolate the tailings [19].

The RLITMF design and tailings deposition consisted of a bottom rock drain, an engineered pervious surround method, originally placement of tailings as a semi-dry filter cake, and closure with water cover over an engineered cover and a soil and sand diffusion barrier. This original plan was modified in 2008 with a saturated drainage layer. The tailings would be prevented from seeping into the pervious surround by a gradational increase in grain size of the pit surround from a sand filter, to larger crushed rock at the outer-most zone of the surround. The collection of groundwater in a raise well would create an inward gradient controlling the tailings seepage water during the operating phase.

Tailings placement techniques evolved over time from mechanical placement of filtered tailings, to thickened subaerial discharge to injection of tailings below the tailings surface. At an early stage of tailings disposal, ice lenses had formed during the winter exposure period and some lenses remained in the tailings thereafter. The principle concern about residual ice was related to physical instability arising from ongoing consolidation due to melting and the release of tailings porewater. Thermal modeling predicted that natural thawing of the ice lenses accumulated in the RLITMF would occur within 100 to 200 years. This natural thawing would release tailings porewater water, which would necessitate a seepage collection and a treatment system being maintained during the extended thawing period. For this reason, deep injection of fresh tailings into the deposited tailings was initiated in 2000 to warm and thaw existing ice lenses, and prevent the creation of new ice lenses.

More than seven million tonnes of tailings solids have been deposited in the RLITMF since the commencement of tailings backfill in 1985. Initially, filtered tailings were placed in the Facility but starting in mid-1986, a thickened slurry waste material was deposited directly into the pit

[26]. The target solids density for the resultant mill slurry was 35% to 40% to prevent particle segregation in the tailings slurry. The weighted average tailings slurry density for 2008 was 39%, which was an increase from the 36.6% in 2007. The average tailings dry density in 2008 was 1.085 t/m^3 and was 1.094 t/m^3 the following year.

Monitoring instruments were installed in the pit surround as it was being backfilled and in wells around the pit for ground water monitoring. The results of environmental monitoring of the RLITMF, after almost three decades of operation, demonstrate complete containment [1][18]. Down gradient monitoring of groundwater has shown contaminant concentrations well below the requirements of the Saskatchewan Ministry of the Environment.

In 1992, the projected tailings height was 401 masl (metres above sea level) and by January 2006 the RLITMF had attained 411.8 masl hosting approximately 5 Mm^3 [19]. In 2009, the capacity of the RLITMF was increased to 9 Mm^3 from approximately 7 Mm^3, by constructing the Pit Crest Expansion [20]. This was accomplished by excavating a portion of the adjacent former Rabbit Lake – lakebed and increase the final tailings height to increase the overall capacity.

Decommissioning of the in-pit facility is anticipated to occur 15 to 20 years after the final tailings are deposited at Rabbit Lake.

Modifications for the potential co-milling of Eagle Point ore and Cigar Lake uranium-rich solution were outlined in the 2008 Rabbit Lake solution processing project environmental assessment and included the pit crest expansion [4].

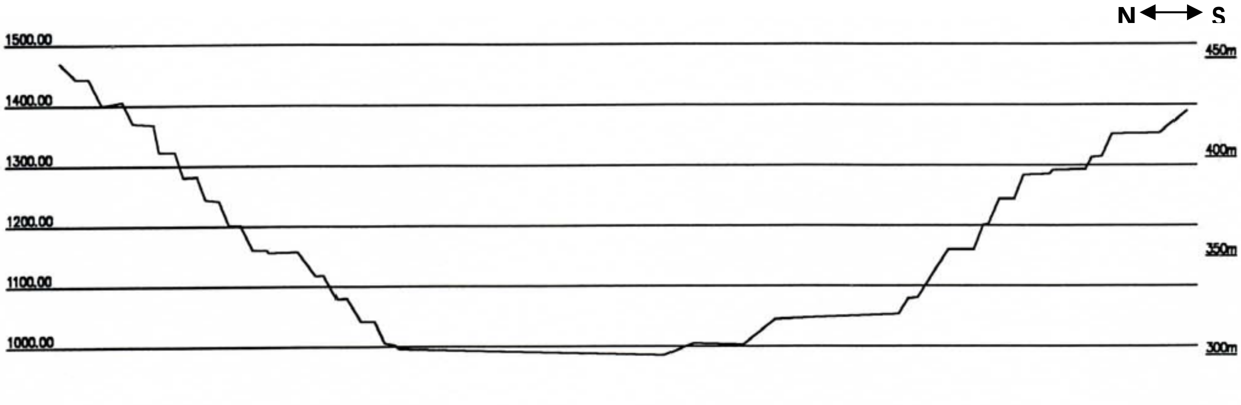
RABBIT LAKE PIT DESCRIPTION

The original Rabbit Lake open pit surface dimensions were approximately 460 m by 365 m with a maximum depth of about 122 m [28]. The pit was mined by conventional methods from 1975 to 1984, and had a low inflow of mine water (about 25 m^3/hr). Figures 9-3 to 9-5 depict pit conditions during and post mining of the Rabbit Lake open pit.

Figure 9-3 Rabbit Lake Pit during Operations in 1979 (Sask. Gov't # 1453-232) [5]

Figure 9-4 Rabbit Lake Pit at the Start of Tailings Deposition in 1985 [27]

Figure 9-5 A North-South Cross-Section of the Mined Rabbit Lake Pit (1999) [21]



The Rabbit Lake Mine site is located in the Canadian Shield. Locally, granites are overlain by 450 m Aphebian Wollaston Group sediments metamorphosed to the amphibolite facies. Host rock types include biotite-quartz-feldspar gneiss, quartz feldspar gneiss, quartzite, calc-silicates and metamorphic pegmatite. Structurally a series of reverse faults and fracture zones hosted the uranium mineralization [2]. The uranium mineralization ore occurred in all the major rock types as a number of high-grade lenses and is not strata-bound or strata-form [2].

STRATEGIC USE AND DESIGN OF THE IN-PIT FACILITY

The original RLITMF design involved the use of a bottom rock drain, an engineered pervious surround method, placement of tailings as a semi-dry filter cake, and closure with a surface lake and a soil/sand diffusion barrier. The closure design was later altered to include an engineered cover instead of the creation of a shallow pit lake.

The design criteria for the pervious surround included: minimizing the potential for ground and surface water contamination, and permanently containing the tailings material and porewater in the Facility [20].

Table 9-1 Input Parameters for the Visual MODFLOW/MT3D99 Modelling of the RLITMF [24]

Parameter	Tailings Body	Filter Sand	Coarse Rock Drain	Regional Aquifer
Total porosity	0.43[a]	-	-	-
Effective porosity, n_e	0.36[a]	0.30[b]	0.30[c]	0.25[c]
Bulk density (g/cm^3)	1.38[d]	1.63[b]	1.80[c]	1.98[c]
Horizontal hydraulic conductivity K_h (m/s)	1.0×10^{-7}[c]	1.0×10^{-4}[c]	1.0×10^{-1}[c]	1.0×10^{-5}[c]
Vertical hydraulic conductivity K_v (m/s)	2.8×10^{-8}[c]	1.0×10^{-4}[c]	1.0×10^{-1}[c]	1.0×10^{-5}[c]
Effective diffusion conductivity D_c (m^2/s)	4.5×10^{-10}	2.0×10^{-10}[b]	n/a	n/a
Adsorption coefficient (arsenic) K_d (cm^3/g)	2[a]	0	0	0

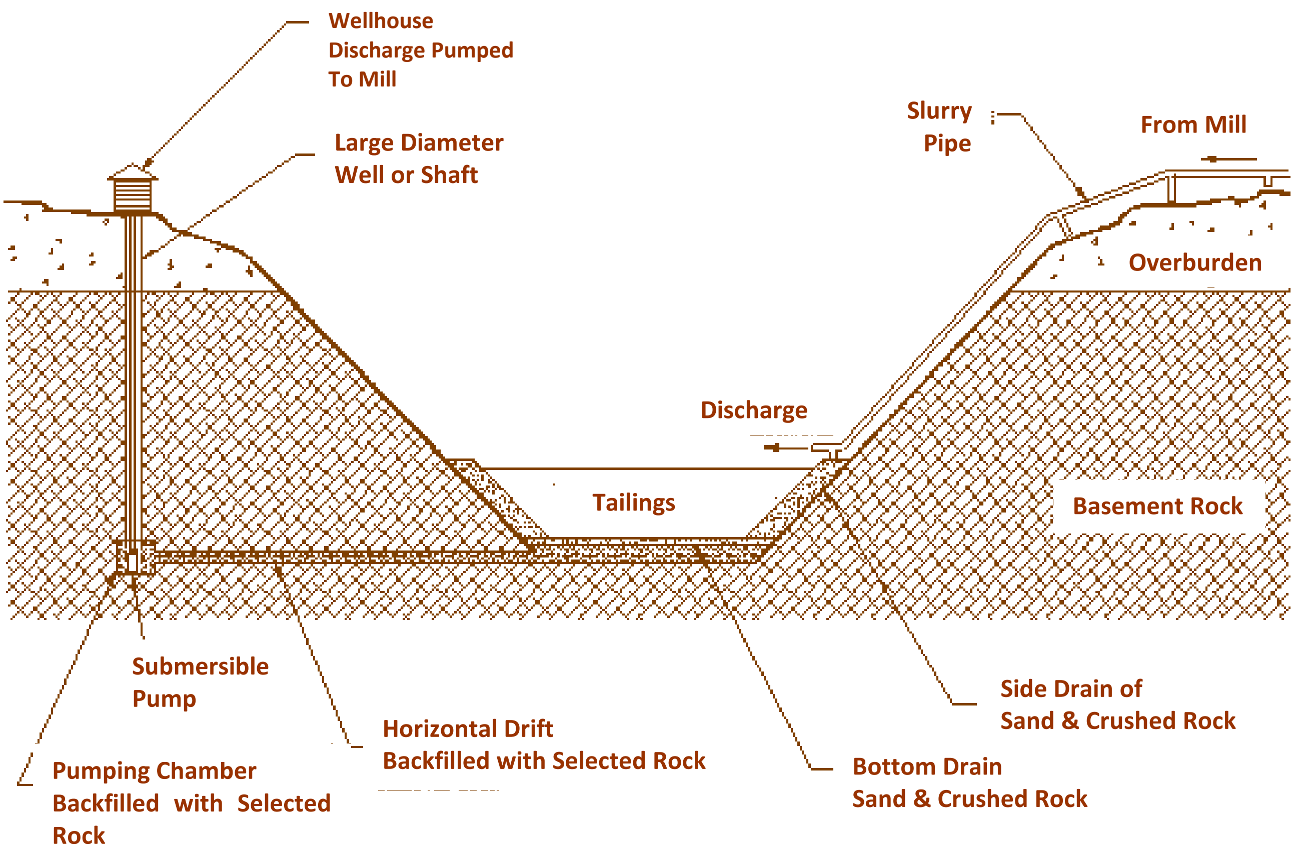
[a] Current Study.
[b] Based on laboratory testing of samples collected from the Rabbit Lake filter sand (Golder Assoc. 1995).
[c] Based on hydrological studies conducted for the RLITMF (Golder Assoc. 2005).
[d] Predicted density of the tailings in the RLITMF (Cameco 1999).

A significant aspect of the pervious surround design was the highly permeable zone of crushed rock and filter sand surrounding the tailings. The solid particles in the tailings would be prevented from migrating into the pervious surround by a gradational change in grain size from the coarsest zone of the surround, to the stored tailings. To achieve this, the grain size was closely controlled during the construction of the surround.

The pervious surround provided a way to drain and collect the tailings supernatant and pore water, allowing for the dispersion of the excess porewater and assisting in tailings consolidation. The RLITMF drainage system consists of a three-layered drain at the bottom of the pit, and a two-layered side-drain along the pit walls. Drains are maintained in a dewatered condition via a lateral drift and a raise pump house with a normal operating water level at an elevation of 308 masl [4, 21].

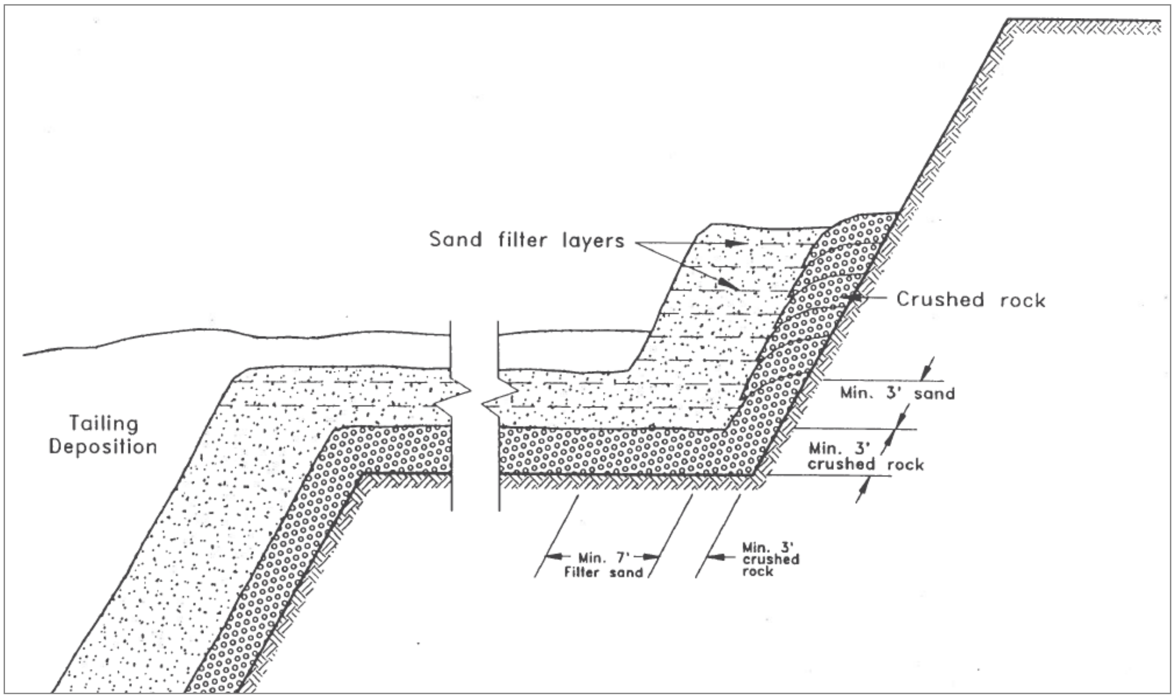
Tailings seepage water is collected in the raise well and either pumped to the mill for reuse as mill process water, or treated in the mine water treatment plant. The design also allows for the surrounding groundwater to be collected which creates an inward gradient that maintains the containment of tailings seepage water. (Figure 9-6) [4].

Figure 9-6 The Pervious Surround and Drainage System of the Rabbit Lake In-Pit Tailings Management Facility [18] (early tailings deposition method depicted)



The bottom drain of the surround consists of a 4 m thick select rock fill layer, a 1.5 m thick intermediate rock layer and a 1.5 m thick filter sand layer. The side drain consists of a 1.0 m thick layer of permeable rock along the full perimeter of the pit. The minimum thickness of the filter zone is 1.0 m and 2.0 m for the vertical bench faces. The filter zone width was maintained at 3.5-4.5 m [4].

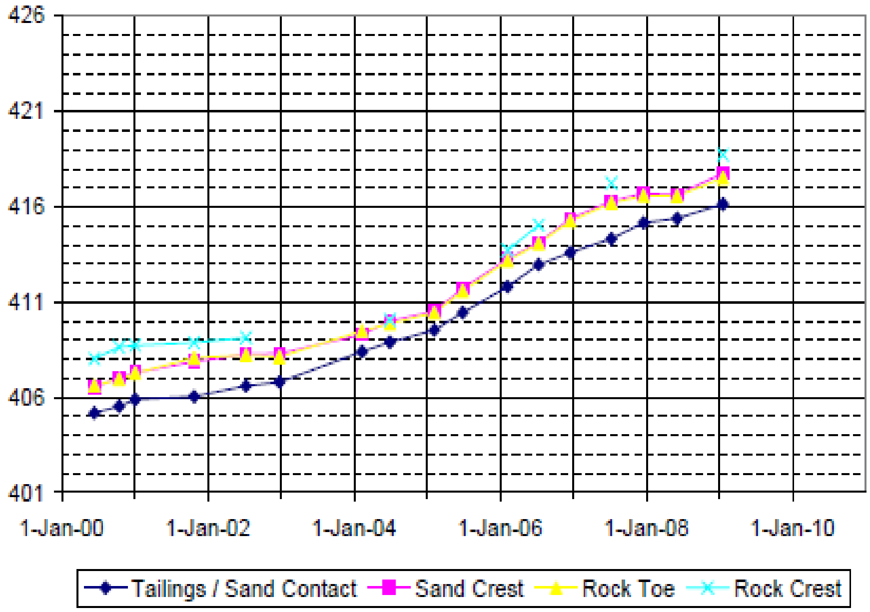
Figure 9-7 General Section Construction Planning Phases- Criteria of the RLITMF Pervious Surround [29]



Tailings placed in the pit consolidate from their own weight [1]. When the tailings consolidation is complete, pumping and treating the drainage from the pervious surround will cease [20]. Once the pit is totally filled, the tailings would be covered with engineered layers of soil, sand and crushed rock. The pervious surround enclosure has horizontal drainage across the crushed rock section. The surround will be covered with till to allow re-vegetation.

The pervious surround was continuously constructed as the pit was filled maintaining an elevation approximately one metre above the tailings (Figure 9-8 [2]). Pit expansion and construction of the horizontal drain was completed in 2008. The total volume of available space increased by approximately 650,000 m^3 by increasing the final tailings height to 426 masl and a lateral excavation along the north and west side. The final tailings height was increased in height, by about 5.5 m [1][2]. The total capacity for tailings storage (within the pervious surround) is approximately 9 Mm^3 [20]. Installation of new tailings distribution lines and other minor construction activities was completed in 2009, with the first tailings placement into the expansion occurring over that same summer [2].

Figure 9-8 Elevation of the RLITMF Pervious Surround and Mine Tailings [2]



MATERIALS PLACED IN-PIT AND METHODS OF PLACEMENT

The tailings were deposited using various methods - subaerial discharge, submerged discharge and deep injection, which began in 1999. The total quantity of tailings solids deposited in the RLITMF by the end of 2010 was approximately 7.5 Mt since first placement in 1985 [4]. In 2006, the depth of the tailings was 106 m.

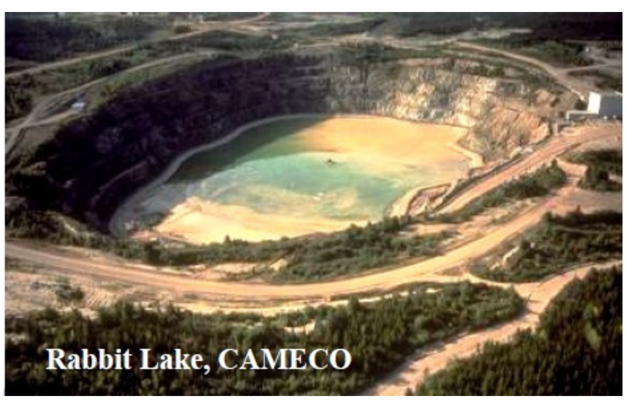
In 2008 the total solid input into the Facility was 206,113 metric tonnes with an average slurry density of 39%. The relatively low % solid is typical of leached uranium tailings that contain hydroxides and gypsum precipitates from the neutralization of leach acid and from solution treatment sludges. Overall, the tailings contain fine to coarse-grained silts and sands, with a maximum particle size of 0.425 mm. It is the precipitate and fines fraction that controls the tailings physical properties, in particular water content, and this component of the tailings contain most of the potential contaminants – radionuclides, metals, arsenic and other minor contaminants.

The in-pit disposal technology used at Rabbit Lake allows for the high water content tailings to consolidate while providing containment during operations and closure. The engineering and control of the tailings properties through to deposition, and the engineered pervious surround are key elements of the technology.

Table 9-2 Tailings Deposition in the Rabbit Lake In-Pit Tailings Management Facility [24][2][21]

Date	Cumulative Quantities (tonnes)	Cumulative Volume (million m^3)	Tailings Surface Elevation (m)	Ore Source
1986	459,715	0.566	343.00	B-Zone
1987	1,024,312	1.105	357.30	B-Zone
1988	1,509,003	1.568	366.10	B-Zone
1989 (June)	1,815,120	1.898	371.60	B-Zone
1991	1,993,668	2.000	373.20	B-Zone
1992	2,424,708	2.470	379.80	B-Zone
1993	2,833,518	2.870	384.70	B-Zone, Eagle Point
1994	3,057,629	3.110	387.50	B-Zone, Eagle Point
1995	3,293,548	3.370	390.00	B-Zone, Eagle Point
1996	3,604,865	3.670	393.30	D-Zone, Eagle Point
1997	4,038,987	4.180	398.46	A, B, D-Zones, Eagle Point
1998	4,530,348	4.670	402.70	A, B-Zones, Eagle Point
1999	4,758,163	4.920	404.88	A, B-Zones, Eagle Point
2000	4,992,428	5.033	405.88	A, B-Zones, Eagle Point
2001 (June)	5,144,087	5.130	406.64	A, B-Zones, Eagle Point
2002 (December)	5,240,427	5.150	406.80	A, B-Zones, Eagle Point
2004 (February)	5,564,069	5.360	408.40	A, B-Zones, Eagle Point
2005 (February)	5,805,541	5.500	409.53	A, B-Zones, Eagle Point
2006 (February)	6,114,930	5.786	411.81	A, B-Zones, Eagle Point
2008	7,000,000	7.1	-	Eagle Point-
2010	7,500,000	-	-	Eagle Point-
2013 (originally projected)	-	8.5	-	Eagle and solution residue from McClean[24]

Figure 9-9 Rabbit Lake During Tailings Filling in 1992 (left) and in 2009 (right) [28][2]



[24] Solution from McClean not included in current plans

The tailings are expected to develop a desiccated surficial crust two years after the last tailings are deposited at which time drilling will be completed to check for residual frozen tailings zones. Any remaining frozen layers within the tailings will be actively thawed. The final cover layer is expected to be 3 m of clean waste rock that will be placed on the facility up to an approximate height of 430 masl.

PERFORMANCE OF IN-PIT DISPOSAL

Cameco has completed extensive core sampling to determine the concentration of contaminants in both the tailing solids and the porewater in the RLITMF as shown in the data below. Monitoring wells were installed around the pit for ground water monitoring, as well as in the pit surround as the pit was being backfilled [2].

The vertical distribution of compounds of Fe^{3+} and As^{5+} was investigated. This correlation identified age-related stratigraphic layers within the tailings management facility [23].

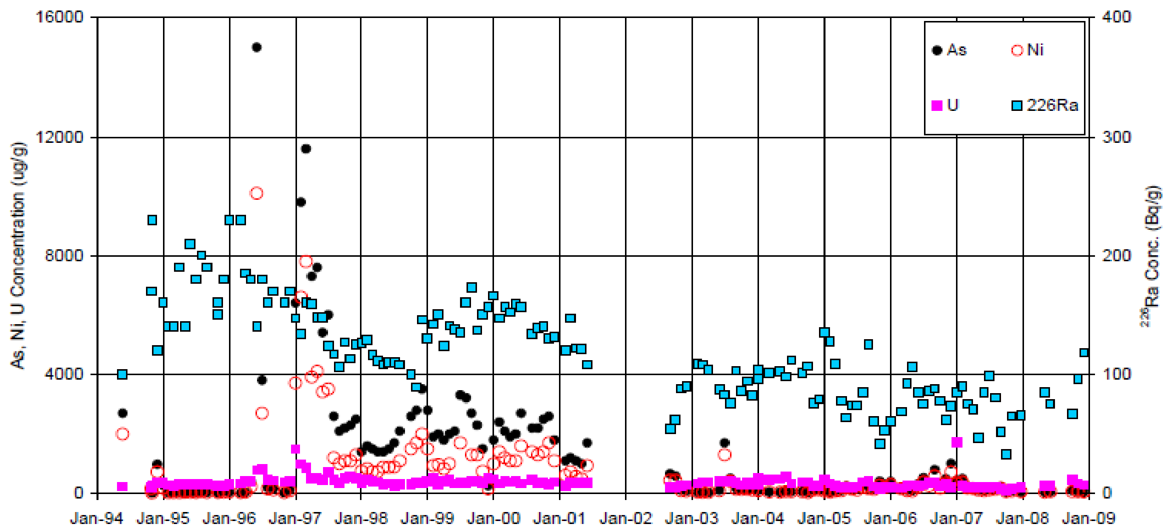
Arsenic is an important component of some of the tailings in the pit. Arsenic mobility is controlled by absorption by ferrihydrite $(Fe^{3+})_2O_3 \bullet 0.5H_2O$ causing it to co-precipitate; ferrihydrite transforms to goethite or hematite thereby making it stable in the RLITMF [23]. This limits the concentration of As dissolved in the regional groundwater system to acceptable levels over the long-term (>10,000 years) [24]. Table 9-3 shows As and Fe analyses of the RLITMF tailings.

Table 9-3 Whole Rock Analysis of Tailings Samples from RLITMF Between 1991-1999 [24]

Parameter	Depth below Surface (m)					
	46.6 m	**28.5 m**	**21.1 m**	**7.3 m**	**3.0 m**	**1.5 m**
As (µg/g)	5,000	5,954	55.8	2,968	5,040	1,614
Fe (µg/g)	30,247	23,454	12,593	25,721	27,091	23,625
Fe/As (M)	8.1	5.3	303	11.6	7.2	19.6
As/kg of ferrihydrite (mg)	104,305	159,618	2,790	72,756	117,205	43,086
Year deposited	1991	1993	1995	1998	1999	1999

Figure 9-10 below shows a slight decrease in some potential contaminant concentrations in the tailings solids between 1994 and 2009. 2009.

Figure 9-10 Concentrations of As, Ni, U and Ra^{226} in Tailings Solids at Time of Discharge into the In-Pit Tailings Facility [2]



Early data from monitoring stations in the Facility's pervious surround show moderate parameter levels as shown in Table 9-4.

Table 9-4 Mean Concentrations of Parameters from Monitoring Stations in the RLITMF Pervious Surround (1990-2002) [2]

Parameter	Station 8.2.1 (mg/l) [2]	Station 8.2.2 (mg/g) [16]	Station 8.2.3 (mg/L) [2]
pH-Lab	8.1	8.2	8.2
Ca	28	29.3	10.4
Cl	8.6	9	8.2
Hardness	153	156	140
HCO_3	151	151	148
K	4	4	3.8
Mg	20.4	21	20.6
Na	26	25.6	25.5
SO_4	74	78	75
TDS	258	257	253.4
As	8.7	8.3	8
Mo	0.021	0.015	0.014
Ni	0.002	0.002	0.002
Ra226 (Bq/g)	0.07	0.107	0.08
U (µg/L)	39	47	104

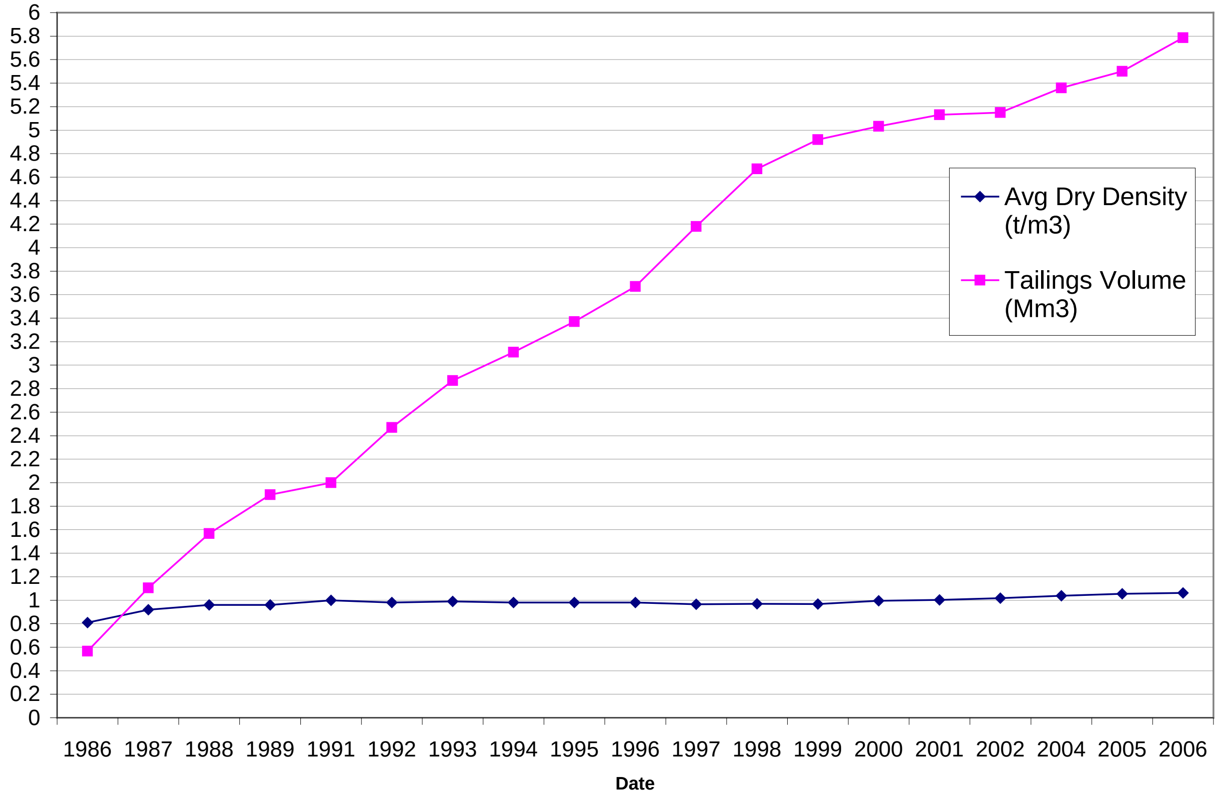
Table 9-5 compares the results of tailings and porewater analyses, from 1994 and 2008. The data shows very low metal concentrations, low radionuclides, and low nitrogenous and non metal compounds. Calcium, iron and nickel concentrations have declined but more importantly so have arsenic, uranium and Ra226 concentrations in both the tailings solids and the porewater. Aluminum and manganese have increased marginally over the same period.

Table 9-5 Comparison of Tailing and Porewater Parameters of the RLITMF [2][5][[16][28]

Parameter	1994 Tailings Solids Mean Conc. (mg/g)	2008 Dec. Tailings Solids (mg/g)	1994 Tailings Pore Water Mean Conc. (mg/L)	2008 Dec. Combined Tailings Porewater (mg/L)
pH-Lab	-	-	8.4	7.62
Alk-T	-	-	-	55
Ca	50	20.7	650	425
Cl	-	0.15	-	106
CO_3	-	-	-	1
Hardness	-	-	-	2980
HCO_3	-	-	-	67
K	10	9.2	17.5	31
Mg	9	12	7.2	468
Na	-	0.54	-	160
SO_4	1300	49.4	1886	2900
TDS	-	-	-	4670
Al	29	48	0.88	0.013
As	7	0.044	55	28
Cu	-	0.14	-	0.029
Fe	16	11.6	0.019	0.004
Hg	-	0.024	-	0.05
Mn	0.136	0.220	0.006	1.64
Mo	-	0.093	-	3.67
Ni	5	0.047	0.08	0.036
Pb	0.43	-	<0.02	0.002
Se	-	0.004	-	0.037
Ti	-	-	-	0.002
V	-	0.147	-	0.013
Zn	-	0.014	-	0.04
Zr	-	0.028	-	0.004
Pb210 Bq/g, Bq/L)	-	190	-	0.2
Po210 (Bq/g)	-	74	-	0.4
Ra226 (Bq/g)	57	118	51	29
Th230 (Bq/g, Bq/L)	-	140	-	0.27
U	500	288	200	1830

The tailings have been deposited into the RLITMF via three methods since operations started: 1) deposited as a "dry" filter cake; 2) thickened and subaerially deposited; and 3) injected below the tailings surface. These methods, plus consolidation have slightly increased the average tailings dry density from 0.967 t/m^3 in 1999 to 1.062 t/m^3 in 2006, as seen in Figure 9-11 below.

Figure 9-11 Average Dry Density and Cumulative Tailings Volume of Tailings in the RLITMF [21]



Submersible pumps in the raise well dewater the pervious surround. The raise well water level is typically maintained at 301.7 masl to 309.4 masl (3.6 to 7.6 m above the base level). The total volume of water pumped based on pumping records from the raise in 2008 was 1,431,854 m^3 (since 1994 the average has been 1,418,195 m^3). The water is a mixture of the contained decanted tailings water, porewater and groundwater. The water is pumped to the mill for use as make-up water and/or directed to the effluent treatment system. Average concentrations of monitored components in raise well water for 2008 are shown in Table 9-6. Uranium was present in significant concentrations in the raise well water, probably due to the present of a uranium-carbonate complex.

Table 9-6 2008 Raise Well Station Data [2]

pH-Lab	7.8	**SO_4** (mg/L)	1057
Flow (L/min)	51	**As** (mg/L)	0.85
Solids TSS (mg/L)	1.3	**Mo** (mg/L)	3.8
Hardness (mg/L)	1149	**Ni** (mg/L)	0.1
HCO_3 (mg/L)	140	**U** (mg/L)	4.45
Ca (mg/L)	260	**Ra^{226}** (Bq/L)	3.7
Mg (mg/L)	123		

REMEDIAL

Adjustments to the methods of tailings deposition have occurred throughout the life of the pit, initially to deal with trafficability of the filter tailings, then to reduce the accumulation of frozen tailings. The closure plan for the Rabbit Lake In-Pit Tailings Management Facility has also been modified to include a saturated drainage layer instead of establishing a pit lake, allowing additional tailings storage capacity. At the end of operations residual frozen tailings will need to be thawed to achieve the desired environmental performance.

BENEFITS AND LESSONS OF IN-PIT DISPOSAL AT RABBIT LAKE

Rabbit Lake has been a pioneering case of in-pit disposal with the use of an engineered pervious surround and groundwater collection system, and engineered tailings properties. Through the design of the pit and updated technologies of the disposed tailings, the in-pit disposal methods have created a permanent containment for uranium mine tailings.

Continuous modeling and testing of the pervious surround concept during and throughout the Facility construction aided in the strength of the outcome. Drilling programs within the pit greatly aided the monitoring and adjustment of the disposal of tailings, and addressing the formation of ice lenses.

RESIDUAL ISSUES

No residual issues are currently apparent at the Rabbit Lake Tailings Facility.

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CASE STUDY #10 MCCLEAN JEB AND SUE C PIT DISPOSAL

SYNOPSIS

Located in Northern Saskatchewan, the McClean Lake uranium mine facilities include the mined out JEB pit which has been successfully used as an engineered tailings disposal facility. The JEB in-pit disposal is one of the most sophisticated examples of the in-pit disposal of mine wastes. Also at McClean Lake, there several other pits including the Sue C Pit which has been used to dispose of sub-ore grade waste rock that contains significant concentrations of leachable metals and radionuclides.

The JEB in-pit tailings disposal is a state-of-the art system employing pre-engineering of tailings characteristics and tailings placement under water in a porous natural rock surround envelope. An important aspect of this in-pit disposal concept is the rendering of the tailings as a relatively impermeable mass compared to the porous rock surround. This is in contrast to the Rabbit Lake in-pit scenario where the pit walls are low permeability, tight rock and where and an “engineered porous surround’ was installed (Case Study #9). In addition, the undesirable condition of ice entrainment (observed at Rabbit Lake) has been overcome by the deposition of thickened tailings under a water cover.

The results of the monitoring water quality surrounding the JEB in-pit disposal facility have demonstrated a high level of success in containing the metal, arsenic and radioactivity content of the tailings. In addition, the methodology allows for decommissioning and closure with a high degree of reliability.

LOCATION

The McClean Lake uranium mining facilities are located 700 kilometres northeast of Saskatoon, Saskatchewan, as shown in Figure 10-1 [7]. The McClean uranium deposits are on the eastern edge of the Athabasca Basin [3] in the Upper Collins Creek Watershed. The minimum and maximum average temperatures for the area are -24^{o}C and 15^{o}C occurring in January and July respectively. The average snow depth and rain fall is 45 cm and 101.7 mm respectively, also peaking in January and July [6]. The McClean mine sites are shown in Figure 10-2.

Figure 10-1 Location of McClean Lake Mine in Saskatchewan, Canada [19]

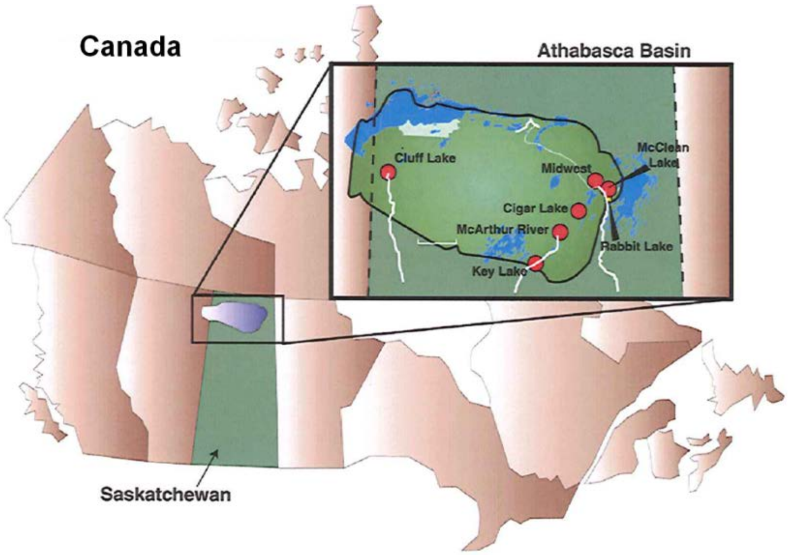
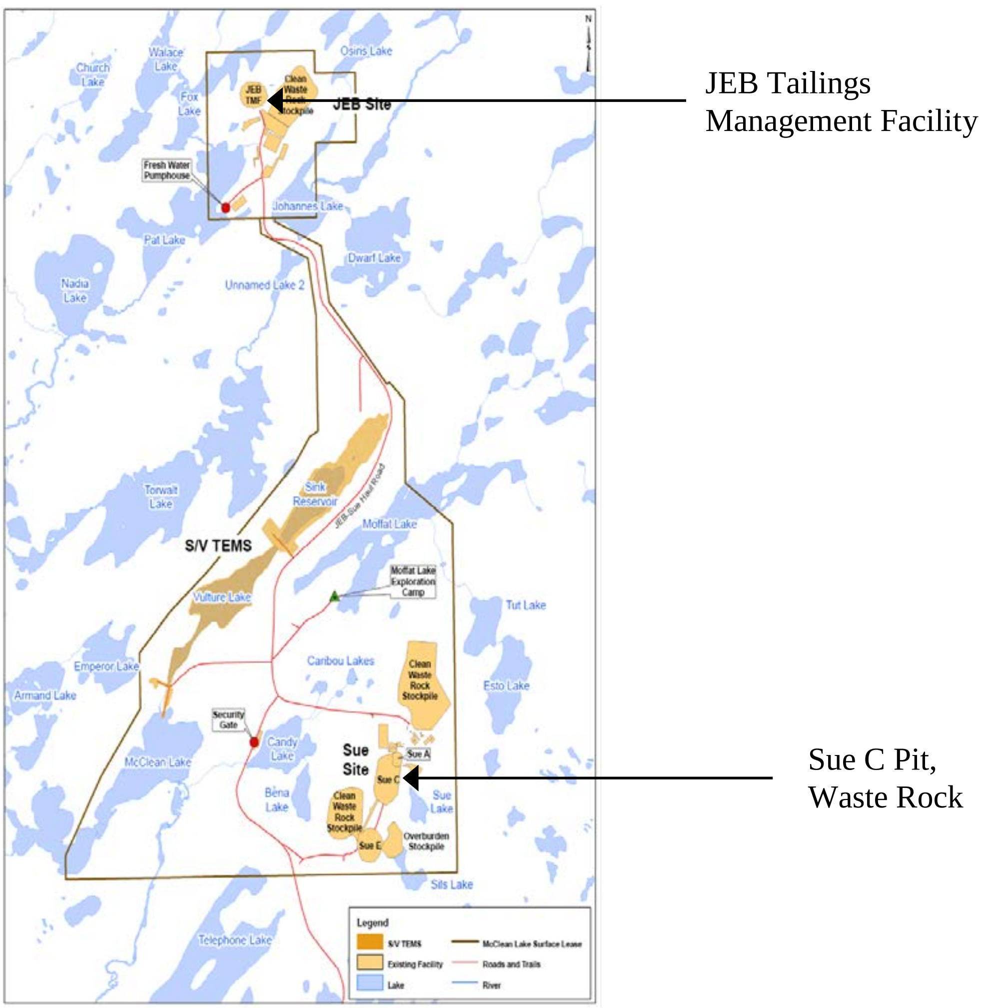


Figure 10-2 McClean Lake Operations, General Site Layout [7]



MINE TYPE and HISTORY

The JEB uranium ore body was discovered in 1982 by the Canadian Oxy – INCO Joint Venture [18][3]. In late 1994 surface stripping of the JEB pit began and in 1997 the deposit was completely mined out using open pit techniques. The JEB open pit mine was converted to the JEB Tailings Management Facility in 1997 (Figure 10-2 and Figure 10-5 show the layout of the new facility). Operated and managed by AREVA, the McClean Lake mill facility began operating in 1999 [17] and is one of the newest and most technologically advanced uranium mining facilities in the world [5]. Ownership is shared between AREVA Resources Canada Inc. (70%), Denison Mines Inc. (22.5%) and OURD Canada Co. Limited (7.5%) [16]. Currently, the milling facility has started to receive high grade ore from the Cigar Lake uranium project.

The Tailings Management Facility (TF) was designed with a 25-year capacity that would allow it to accommodate the tailings from the processing of ore from the 5 McClean uranium mines, as well as from Cigar Lake and other area mines [3]. In November 1999, subaqueous disposal of tailings into the JEB pit was readied and tested. In early 2003 the processing of stockpiled McClean Lake ores and tailings deposition into the JEB pit began [18].

The processing of stockpiled McClean ores was completed in July of 2010 and the McClean facility was put into a care and maintenance mode. However, operations were restarted in the JEB Mill in 2012 to test modifications that would accommodate the high grade Cigar Lake Mine ore anticipated to be received in 2013 [5] . The Cigar Lake Mine has a milling arrangement with McClean Lake operation for processing 100% of the ore – this processing actually began in October 2014 [2, 25].

The mill initially had a capacity of 6 million lbs of U_3O_8. In 2001, the annual licensed capacity was increased to 8 million lbs (3,077 tonnes uranium) until 2009 [17]. As of 2008 the mine had only 2.7 million lbs of uranium reserves in ore stockpiles with an average grade of 0.62% U_3O_8 [25]. The process uses sulphuric acid leaching with hydrogen peroxide oxidation and a solvent extraction recovery process to extract and recover uranium product from the ore [3]. Table 10-1 summarizes the ore milled, the grade and production of the McClean Lake facility from 2003 to 2007. Uranium recovery was very high – typically >98%.

Table 10-1 Operation of the McClean Lake Uranium facilities from 2003 to 2007 [1]

	2003	**2004**	**2005**	**2006**	**2007**
Ore Milled (thousand tonnes)	132	152	177	131	170
Average Grade (% U_3O_8)	2.07	1.86	1.45	0.68	0.53
Production (thousand pounds U_3O_8)	6,028	6,005	5,490	1,795	1,905

The ore is composed chiefly of quartz, illite, kaolinite and chlorite, and precipitated secondary minerals include gypsum, scorodite, annabergite, hydrobasaluminte, kaolinite, pyrite, pitchblende, chlorite, theophrastite and ferrihydrite [20].

The deposit was a fault-controlled uranium mineralized pod which was approximately 50 m by 122 m by 7 to 17 m thick. The majority of the deposit, which included the higher grade uranium ore (up to 40% U_3O_8), occurred as pitchblende nodules within the sheared, strongly altered, clay-rich basement graphitic pelitic gneisses [18].

As well as uranium and other radionuclide contents, the McClean ores contained significant concentrations of sulphides and arsenides, as well as heavy metals such as nickel and cobalt [18]. The tailings retain all the heavy metals, arsenic and uranium progeny radionuclides (e.g. Th230, Ra226). In addition to the immobilization of heavy metals with lime, iron was added to immobilize arsenic and precipitate radium as barium-radium sulphate.

The current milling plans for the next 25 years at McClean Lake's JEB Mill anticipate the generation of about 5 million cubic metres of tailings, primarily from the Cigar Lake high grade ores. AREVA proposes to expand the JEB TMF capacity by 4.5 million cubic meters (Mm^3) with construction starting in 2017. The expansion plans, submitted in August 2011, outline the details of an increased elevation of placed tailings to 465.5 masl. The embankment around the pit perimeter to an elevation of 468 masl 18 metres higher than the current low side of the JEB Facility as depicted in Figures 10-3a and b. A minimum 10 m set back would be maintained between the toe of the embankment and the adjacent Fox Lake high water mark. The inside slope the embankment would be constructed with an impervious till/bentonite liner to ensure containment of the pond above the deposited tailings. [1]

Figure 10-3(a) Concept images of the JEB TMF in its current state (left image) and the proposed pit expansion (right image) [1]

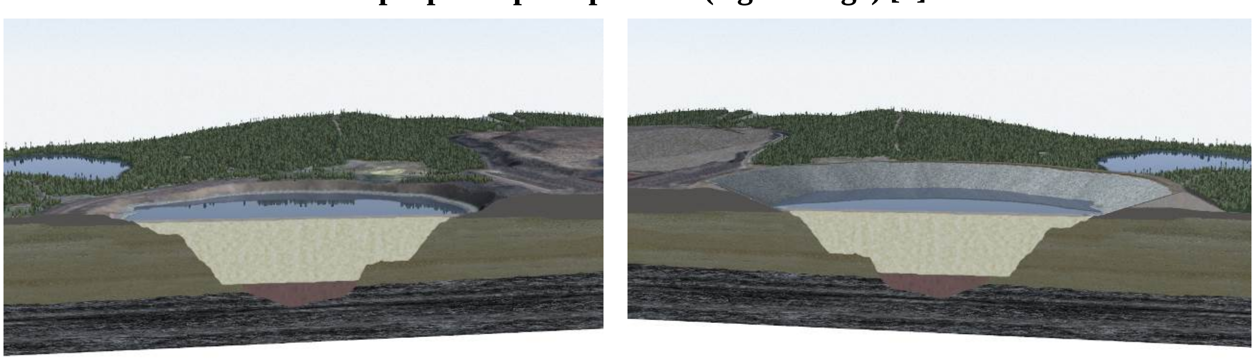
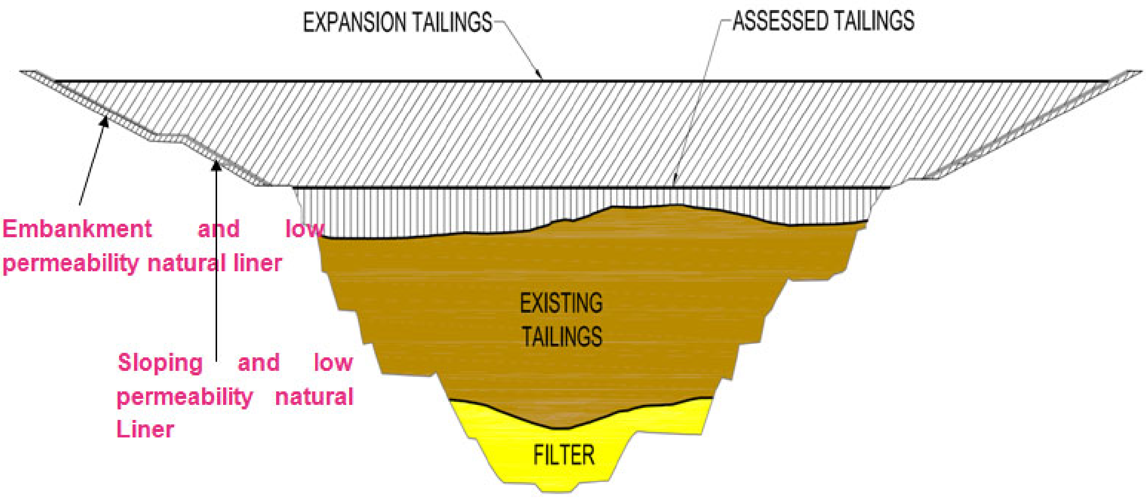


Figure 10-3(b) JEB TMF – Existing and Proposed Expansion [1]



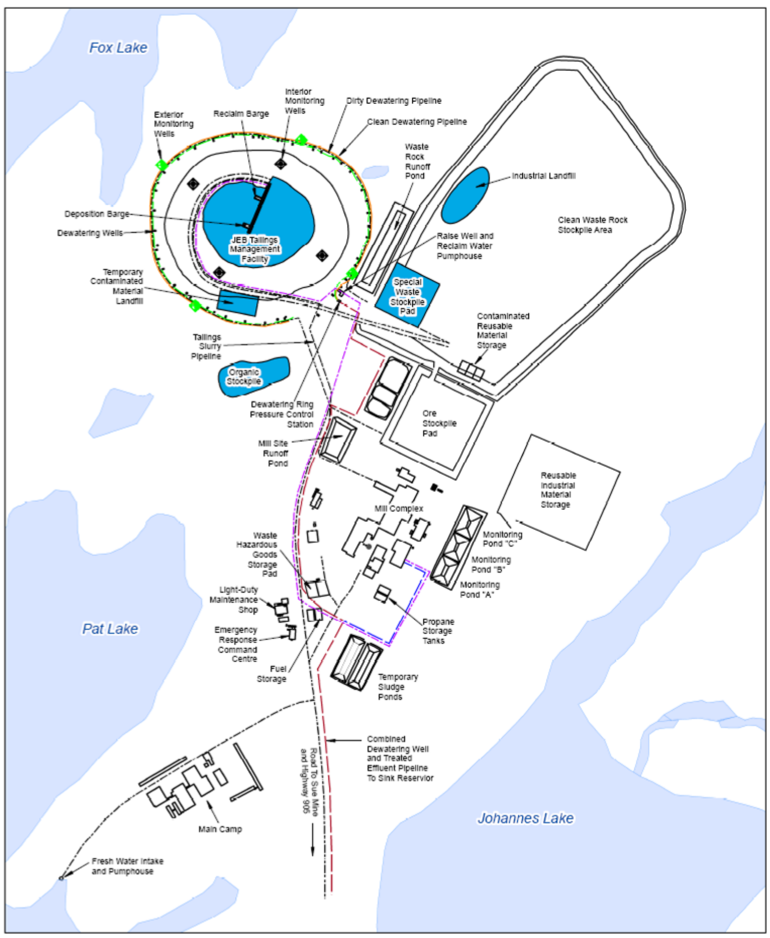
JEB PIT DESCRIPTION

The mined out JEB pit had a volume of 4.8 million m^3, with a maximum diameter of 420 metres and a depth of 118 meters. Figure 10-4 shows the JEB pit when mining was completed, before the pit was backfilled. Figure 10-5 shows a later illustration of the site with a partially filled pit, (Figure 10-6 shows the site plan). Three important aspects of the facilities are shown – (1) mill is in close proximity to the JEB pit, (2) the footprint of the facilities is relatively small and (3) the facilities are surrounded by lakes.

Figure 10-4 Aerial view of JEB Pit before Filling [22]

Figure 10-5 View of JEB open pit tailings (from Google)

Figure 10-6 JEB Site at McClean Lake Operation [7]



STRATEGIC USE OF THE JEB PIT

The Saskatchewan provincial regulatory authorities have a preference for uranium mines, that no new separate above ground tailings storage facilities would be considered, making in-pit disposal of uranium mine tailings the preferred strategy. This preference is reinforced by the technical advantages in isolating both reactive tailings and waste rock in mined-out pits. One of the challenges at McClean was to design in-pit tailings and waste rock disposal systems that would isolate the contained heavy metals, arsenic and radioactivity present in tailings and mineralized waste rock. McClean pits including the JEB pit are in close proximity to local lakes (see Figure 10-4) and the sandstone pit walls are hydraulically conductive.

Many designs were considered for the pit-based TMF including an engineered pervious surround, a partial engineered pervious surround or a natural surround. A natural porous surround was chosen based on containment efficiency and feasibility. The impacts of various configurations were modeled and it was concluded that the natural porous surround would provide the best environmental protection (Table 10-2) [21]. An engineered liner was determined to be difficult to construct in a flooded pit and a partial liner was shown to not have a significant effect on reducing diffusive flux to the environment [4].

Table 10-2 JEB Tailings Management Facility Design Optimization [4]

Alternatives Considered	Initial TMF Design	Optimized TMF Design	Justification
Above ground versus in-pit disposal	In-pit	In-pit	In-pit disposal is expected to minimize environmental effects and provides best long-term physical isolation and security
Pit Choice	JEB pit	JEB pit	JEB pit chosen over SUE E or purpose-built pit due to capacity requirements and no decrease in environmental effect if sited elsewhere; cost of purpose-built pit prohibitive
Subaerial vs. subaqueous placement	Subaerial	Subaqueous	Subaqueous placement reduces worker exposure to dust, gamma radiation and radon gas; provides for less tailings segregation and lower hydraulic conductivity in consolidated tailings; no frost lens formation in tailings mass
Full pit liner vs. partial pit liner vs. no liner	No liner	No liner	Full pit liner may be impossible to construct in a flooded pit. Partial pit liner does not have a significant effect on reducing diffusive flux to environment
Tailings neutralization to strong alkaline pH with CaO. Tailings neutralization with CaO and $Fe_2(SO_4)_3$ to pH 8.5	Tailings neutralization with CaO only to strong alkaline pH	Engineered tailings chemistry with $BaCl_2$ and $Fe_2(SO_4)_3$ addition and final neutralization to pH 8 with lime	Minimize Ra source through addition of $BaCl_2$; minimize As and Ni source terms in tailings mass through ferric sulphate addition and pH control; lower diffusive from tailings on closure
Pervious surround vs. natural surround	Pervious surround	Natural surround (subaqueous)	Natural surround minimizes in-pit construction, protects worker safety; effective containment of

Alternatives Considered	Initial TMF Design	Optimized TMF Design	Justification
	(subaerial)		tailings pore water; decreased containment flux receptors
Unthickened tails, thickened tails, paste tails	Unthickened tails (11% solids)	Thickened tails (35% solids)	Thickened tailings minimizes water to be treated
Final cover – water or solid materials	Final cover – till cap and pond	Final cover – complete backfill with waste rock and till cap	Waste rock cover allows enhanced tailings consolidation and lower tailings hydraulic conductivity; better isolation from surface environment on closure; no flux from tailing to overload surface pond
Dewatering sump, dewatering wells, dewatering drift	Dewatering drift and raise and pit sump	Dewatering drift and raise complemented by dewatering wells	Enhanced hydraulic containment; minimizes amount of water requiring treatment and minimizes loading of contaminants to the environment during operations

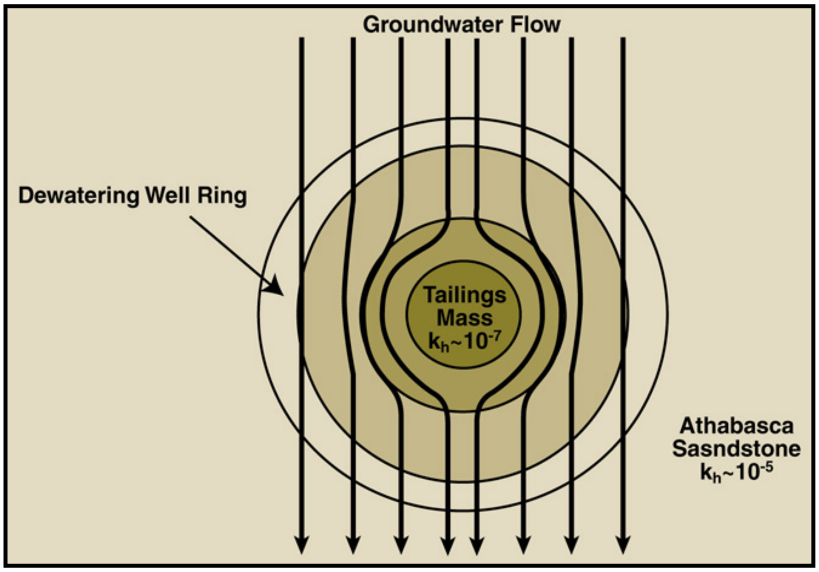
As water tends to flow along the path of least resistance, a crucial aspect of the design was maintaining a much lower hydraulic conductivity in the tailings compared to the surround, thereby allow groundwater to flow around the tailings in the pit facility instead of through them. This significantly reduced the flux of potential contaminants form the tailings into the groundwater.

The use of the JEB pit also required the flexibility for disposal of tailings produced from the milling of the ore from several sources (McClean, Sue A, Sue B, Sue C, Midwest and Cigar Lake) to be deposited in a single pit, thereby decreasing the number of sites requiring long-term monitoring.

During mining operations, waste rock was segregated for metal leaching, ARD potential and uranium content using on XRF testing of samples and radiometric scanning of trucks (briefly discussed below) [19].

The JEB TMF was specifically designed to minimize the migration of soluble constituents from the tailings facility to the receiving environment both during the operating and post-decommissioning periods. During operations and on closure, hydrological isolation has to be maintained and confirmed in measurements. The design of the decommissioned facility would rely on control of source concentrations within the tailings pore water and a hydraulic conductivity contrast between the tailings mass and the surrounding host rock such that groundwater preferentially flows around the tailings. This results in the relatively slow process of molecular diffusion being the dominant mechanism for contaminant release from the tailings, which is minimized by relatively low source concentrations. Figure 10-7 depicts the groundwater flow around the tailing mass due to the lower hydraulic conductivity of the tailings mass.

Figure 10-7 Groundwater Flow at JEB Pit; water flows around the tailing mass due to the lower hydraulic conductivity within the pit [26]



The ability to use subaqueous deposition is important for tailings produced from high grade uranium ore because it provides shielding from gamma radiation, blocks radon emanation and prevents frost entrainment which has been observed during the surficial discharge period at the Rabbit Lake in-pit uranium tailings facility. In addition, dust dispersion, especially from freeze drying in winter, is eliminated. It was decided that the tailings would be deposited using a tremie pipe under a cover of water that would be deep enough to absorb most of the radiation emanating from the material. The pipe would be suspended from a floating barge shown in Figure 10-9 below.

ENGINEERING OF TAILINGS AND PIT PLACEMENT

The JEB open pit mine was modified to suit the requirements of the tailings management facility (TMF). This included the installation of 29 dewatering wells as shown in Figures 10-8 and 10-9 [23]. The facility had been excavated through approximately 10 m of glacial till overburden, 75 m of porous sandstone, and into the granitic basement rock to a total depth of approximately 118 m below grade [24].

Figure 10-8 Design and Operational Features of the JEB Tailings Management Facility and Disposal System [24]

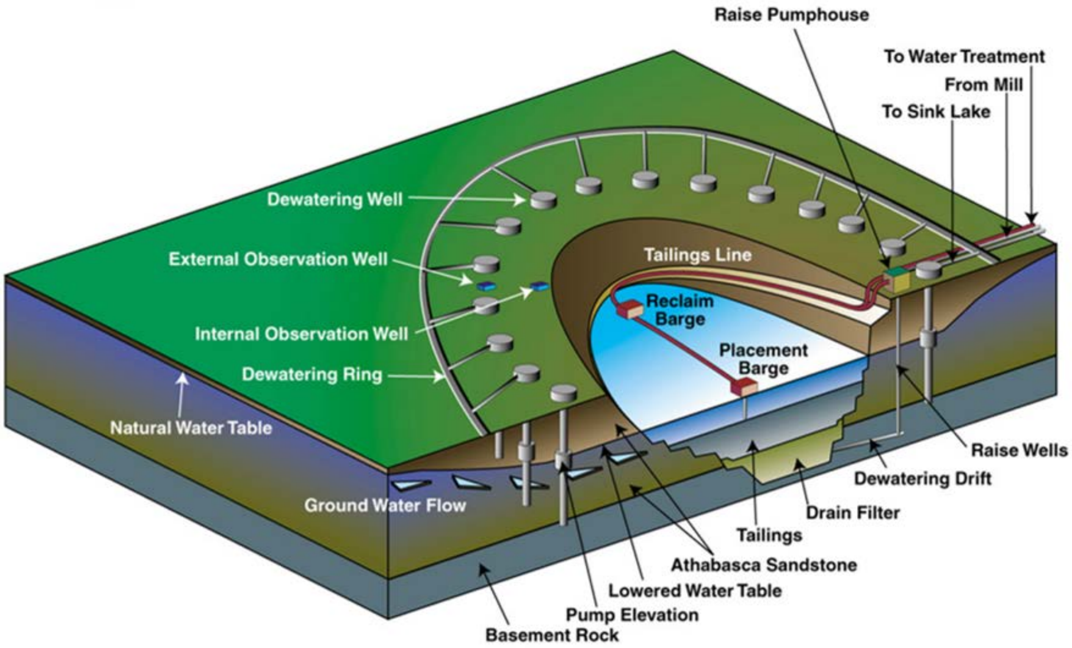


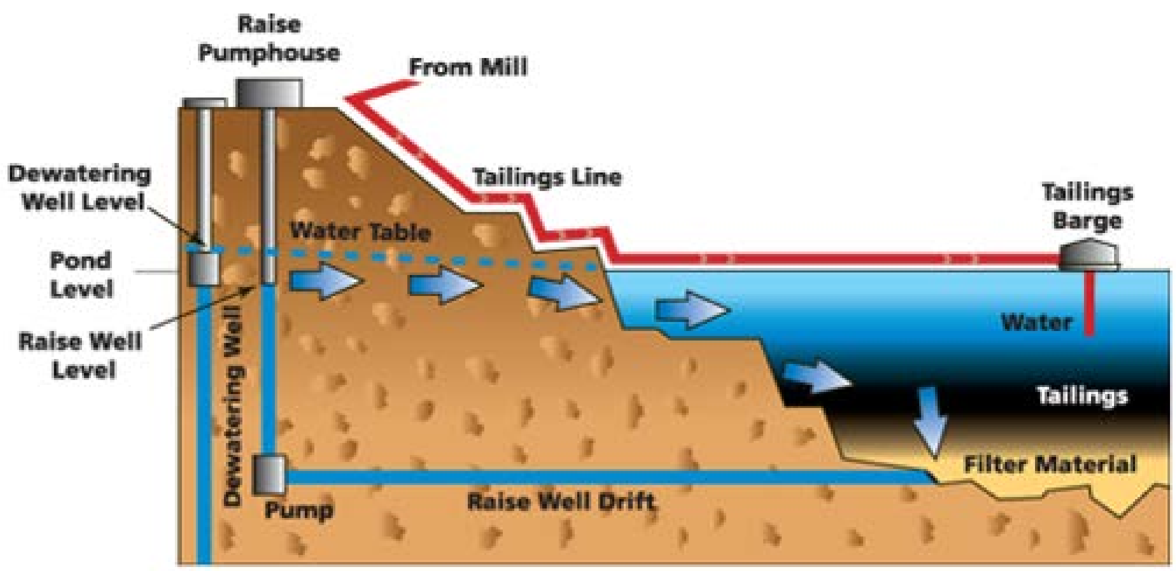
Figure 10-9 JEB Tailings Management Facility [19] [7]

The milling process wastes (tailings and precipitates) are mixed and undergo a lime neutralization process with ferric iron added to precipitate arsenic and barium (chloride) to remove radium. Slaked lime is used to adjust the pH to maximize the efficiency of arsenic precipitation while controlling nickel and other trace metals [4]. Flocculent is added to the neutralized slurry to promote settling. The tailings are thickened to 30% to 35% solids before being discharged to the TMF [24]. This unusually low percent solid (for tailings) is due to the presence of significant percentages of gypsum and metal precipitates common in uranium tailings from acid leach processes.

Solids in the neutralized tailings slurry are typically comprised of 50-70% leach residue minerals with the remainder composed of precipitated solids. The tailings slurry is pumped into the JEB Tailings Management Facility for final disposal using a subaqueous emplacement process. Underdrains allow for the removal of the excess pore water expelled by settling and compaction of the tailings, which is pumped to a water treatment facility [20].

The JEB tailings pit required that the tailings be produced and deposited in such a way that they consolidate in place to produce a material that has a hydraulic conductivity at least ten times less than that of the surrounding host pit rock walls. To maintain this, tailings were placed in the central zone of the TMF to reduce potential segregation by isolating more permeable tailings zones from active groundwater within the aquifer. In this fashion if some segregation does occur, the fine-grained, lower hydraulic conductivity material will isolate the coarse-grained material from the aquifer, thereby minimizing the increase in groundwater flow and therefore the resulting contaminant loading to receptors [4]. Figure 10-10 shows the groundwater flow around the tailings mass in the JEB pit.

Figure 10-10 JEB Tailings Management Facility [4]



To ensure hydraulic containment of tailings pore water during the operating period a ring of de-watering wells was installed around the edge of the pit. The submersible pumps in these wells hold the ground water level at an elevation slightly above the operating desired pond level. These wells act as the primary control on the TMF pond water level at lower regions of the pit and intercept clean ground water before it enters the TMF. As tailings is placed into the upper region of the pit, pumping from the raise well drift is the means of providing hydraulic containment. To monitor ground water levels four observation wells (external) are installed within the ring. In addition, four internal monitoring wells are installed between the de-watering well ring and the pit. A base drain and graded filter package constructed of sand and crushed rock at the base of the TMF allows collection of tailings pore water, while containing the tailings solids above the filter, and enhances tailings consolidation by promoting dissipation of excess pore water pressure within the tailings mass. Water is removed from the base drain and pumped to surface for treatment through a dewatering drift and raise system. Hydrodynamic containment of TMF waters is ensured by maintaining the following water level height hierarchy: exterior well > interior well > pond level > base drain level [24].

The tailings lines from the mill run down the TMF ramp and onto a floating walkway leading to the placement barge (Figures 10-8 to 10-10). The discharge pipe is suspended below the barge and the tailings are placed within previously placed tailings using a shallow injection tremie method. The reclaim water barge is also used to precisely control the pond level by returning excess water (from groundwater not collected by the dewatering wells and that displaced by tailings additions) back to the mill [24].

PERFORMANCE OF IN-PIT TAILINGS DISPOSAL

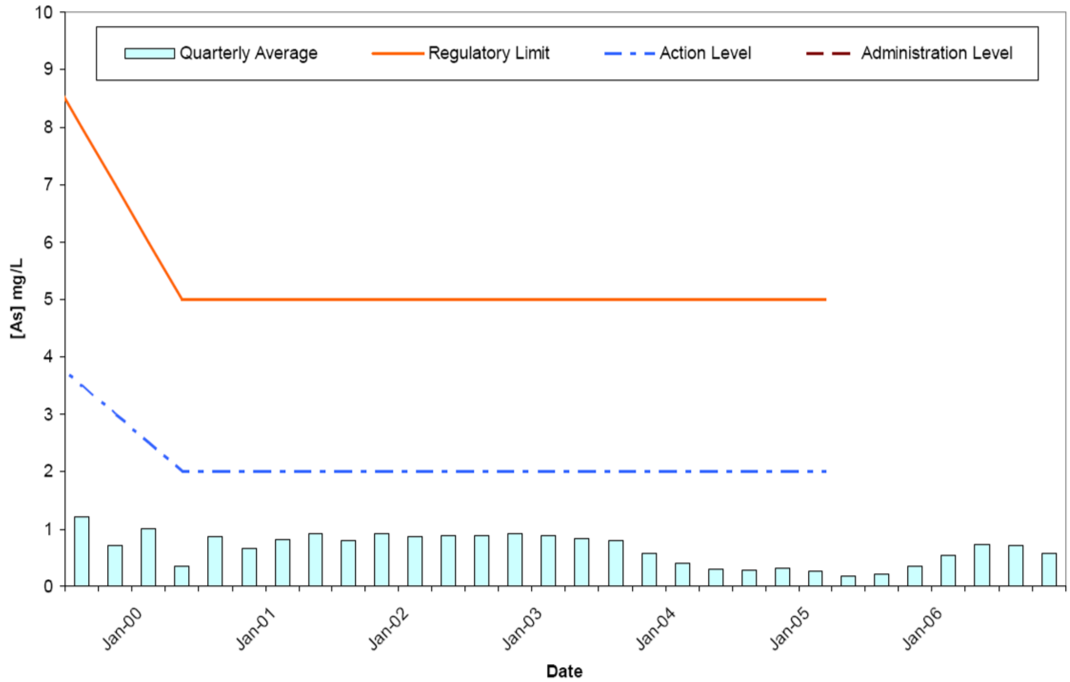
Initially, concerns were expressed by agencies and individuals about the performance of the proposed tailings management facility. Among these concerns was the close proximity of the JEB pit to Fox Lake (about 125 m). Some early evaluations considered that arsenic levels could eventually exceed both Saskatchewan Surface Water Quality Objectives (SSWQO) and Canadian Water Quality Guidelines (CWQG) in Fox Lake [21].

Part of the uncertainty related to the predicted concentrations of arsenic in the actual porewater of aged tailings. Estimates for the arsenic source term ranged from as low as 1 mg/L or less, to as high as 100 mg/L or more. An engineering firm reviewed published data and experimental results from other, existing tailings facilities, and they found that arsenic concentrations in the porewater, instead of decreasing with time, would likely increase at a rate of about 500 per cent per year [21]. To address this potential increase in arsenic with time, which would have likely meant significant impacts to Fox Lake, AREVA experts proposed ways of reducing porewater concentrations by creating conditions in the tailings that would result in the formation of stable secondary minerals. This methodology was confirmed by extensive laboratory tests and solid phase identification.

Several years of operation has consistently produced results that are better than applicable regulatory targets. The success of the in-pit tailings disposal at McClean is due in large part to the control of arsenic and nickel concentrations in the tailings pore water. Arsenic concentrations in TMF porewater decreased from a high of about 57 mg/L to 1-2 mg/L after two to three years of burial. Studies and TMF observations indicate that over the long term, the solubility product of scorodite (iron arsenate) influences the maximum arsenic concentrations in tailings porewaters

[20]. The pore water arsenic concentration in the tailings has remained well below the regulatory limit of 5 mg/L and the action limit of 2 mg/L during production. Figure 10-11 shows the quarterly average arsenic levels for the tailings underflow. Concentrations were well below regulatory and action levels.

Figure 10-11 Quarterly Average Tailings Thickener Underflow Pore Water Arsenic Concentrations [4]



As shown in Figure 10-12, TMF reclaim water is a major component in the mill site water balance. Both the JEB TMF and Sue C site have active treatment plants Lake Mine effluent data collected between 2004 and 2011 show contaminant levels far below Canadian Metal Mine Mining Effluent Regulations as seen in Figure 10-13(a,b,c).

Figure 10-12 Water balance summary within the McClean Processing Facility [19]

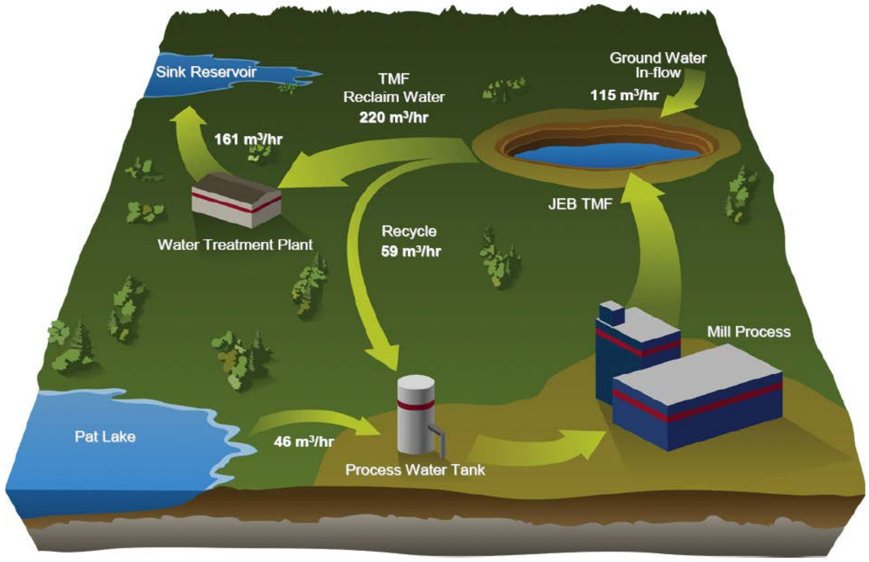


Figure 10-13(a) McClean Mine Effluent Data from the JEB and SUE Water Treatment Plant from January 2004 to December 2011 for As (regulation limit of 0.5 mg/L), Ni (regulation limit of 0.5 mg/L), Ra-226 (regulation limit of 0.37 Bq/L) [9]-[16]

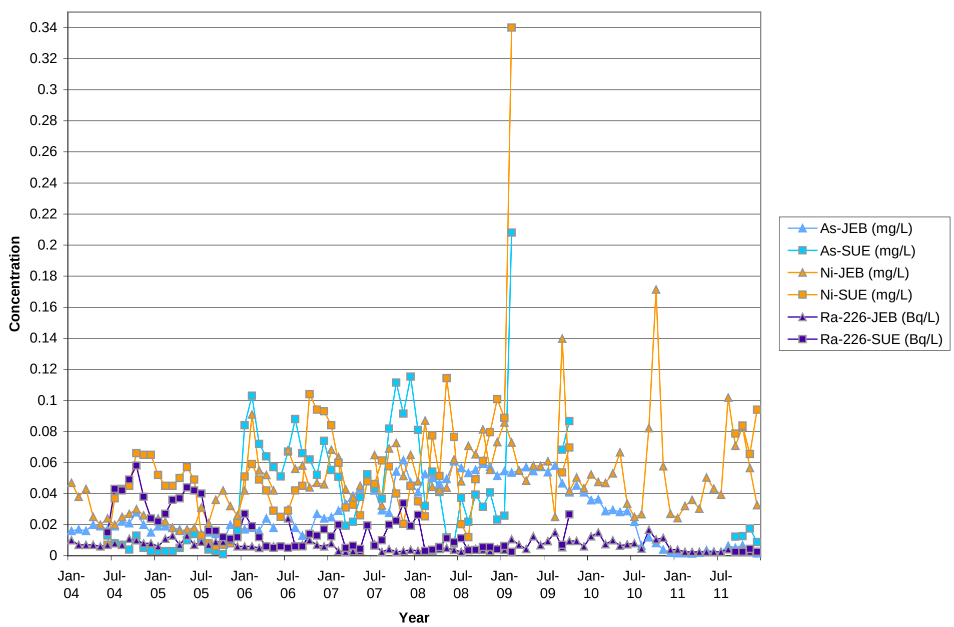


Figure 10-13(b) McClean Mine Effluent Data from the JEB and SUE Water Treatment Plant from January 2004 to December 2011 for Cu (regulation limit of 0.3 mg/L), Pb (regulation limit of 0.2 mg/L), Zn (regulation limit of 0.5 mg/L) [9]-[16]

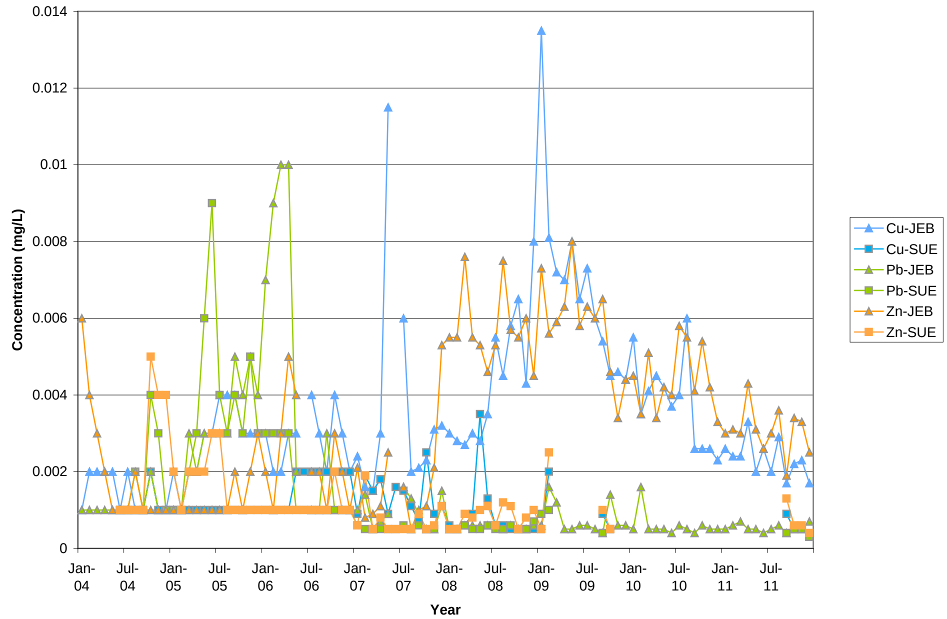
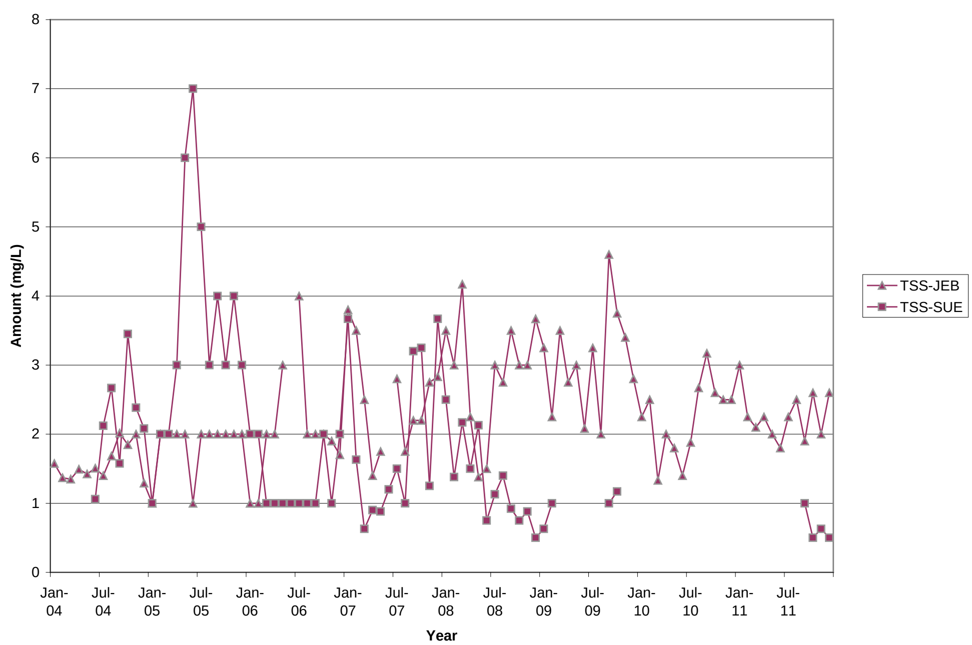


Figure 10-13(c) McClean Mine Effluent Data from the JEB and SUE Water Treatment Plant from January 2004 to December 2011 for Total Suspended Solids Concentration (regulation limit of 15 mg/L) [9]-[16]



Both Fox Lake and Pat Lake both can be affected by migration of soluble contaminants in tailings porewater in the pit [4]. Protection of lake water quality is an important consideration. Both lakes have much lower concentrations of observed arsenic metals than guidelines require. Figure 10-14 and Figure 10-15 show the current and predicted levels of metals for Fox Lake North Basin and Pat Lake respectively. In Fox Lake, all elements maintain a concentration below 10 µg/L and remain below the guideline levels.

The arsenic levels in Fox Lake and Pat Lake are currently below 1 µg/L, and predictions increase the concentrations marginally above 1 µg/L, still remaining significantly below guideline levels (Figure 10-14 and Figure 10-15).

Figure 10-14 Current and Predicted Effects of Tailings Disposal for Fox Lake North Basin [22]

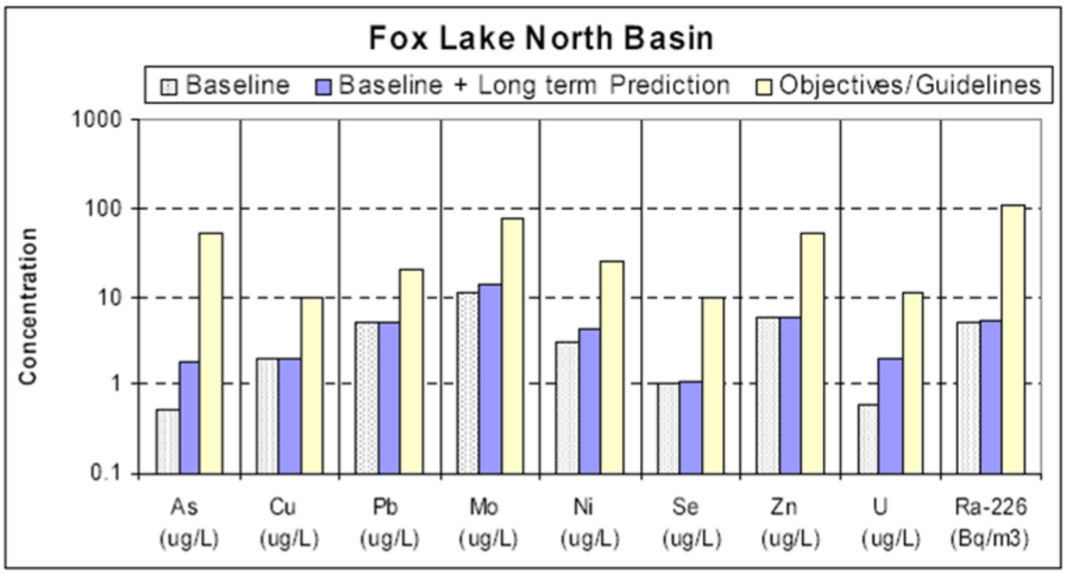
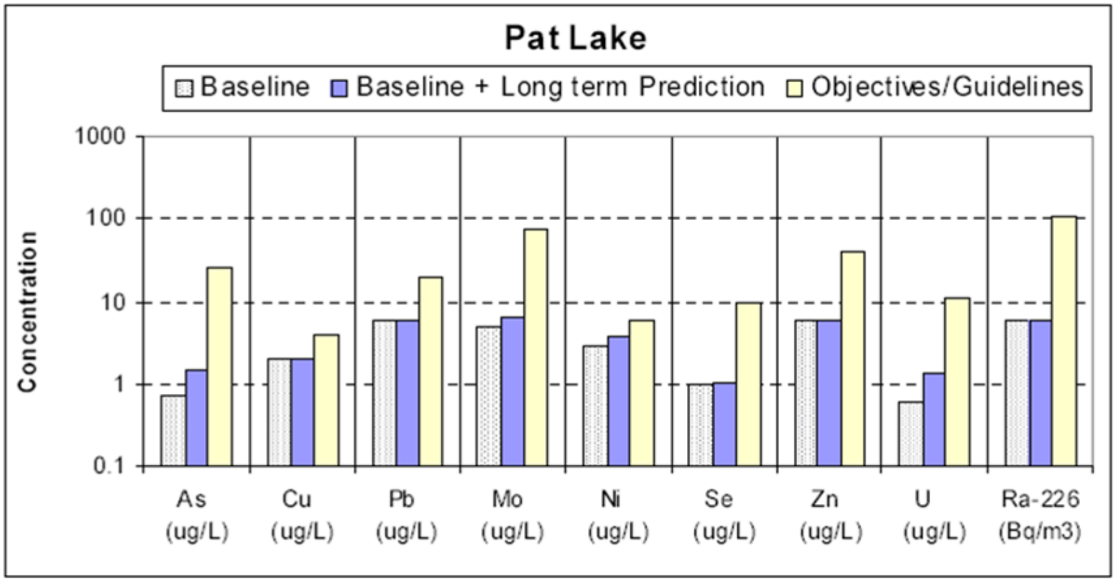


Figure 10-15 Current and Predicted effects of tailings disposal for Pat lake [22]

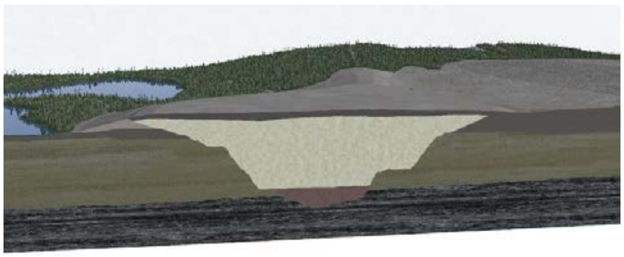


REMEDIAL AND PLANNING FOR THE FUTURE

The JEB TMF is proposed to be expanded [1]. The in-pit disposal is proposed to be extended above grade and contained with impermeable embankments. With the expansion, the JEB tailings facility will be what might be considered a hybrid facility, but with the engineering of tailings geochemical and geotechnical properties and disposal of thickened tailings under a water cover continuing to be important features to ensure contaminant containment.

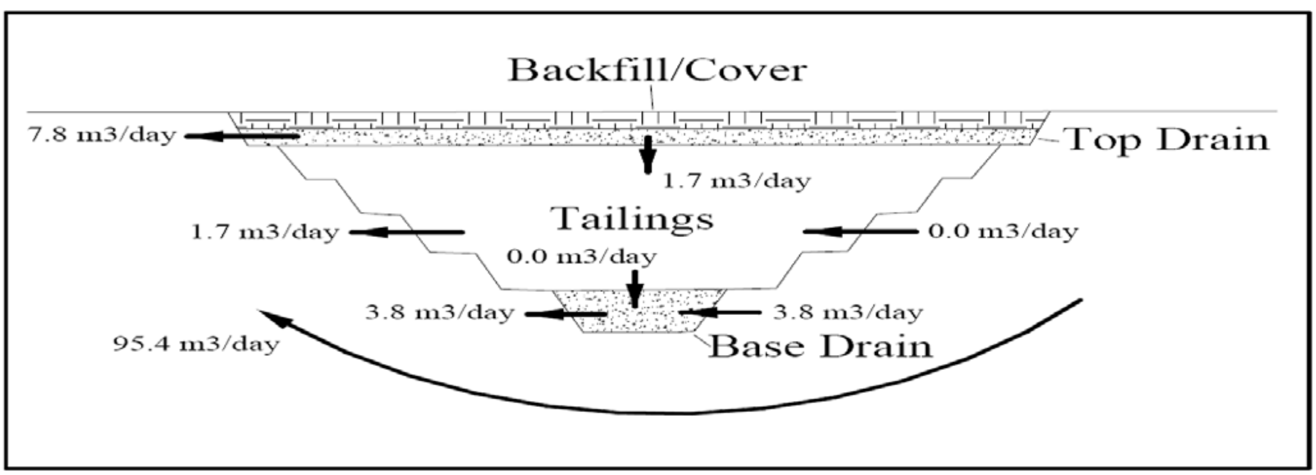
The closure of the facility will be undertaken when the facility is full after at least 25 years of uranium ore processing. Currently AREVA plans to decommission the JEB TMF by covering the tailings with a cap consisting of about over sand, waste rock and till. The till will be amended with bentonite to reduce permeability. The purposes of the cap would be to prevent plants, animals, birds and people from accidentally coming in direct contact with tailings, and to drain surface water away from the pit [21]. The cover will be constructed in such a way to blend into the surrounding environment [1] as shown in Figure 10-16.

Figure 10-16 Proposed Expanded JEB TMF with cover placed during decommissioning, blending into surrounding landforms [1]



Once the tailing deposition reaches a predetermined level, the tailings will be covered by sand, till and waste rock. The water cover would be removed and treated and a leachate collection system would be installed to dewater tailings as the material consolidates under the surcharge of waste rock. Temperature and pressure transducers would be installed in the tailings. It is anticipated that the weight of this mound would cause approximately 4 m of settling due to consolidation of the tailings. This consolidation process would expel porewater which would be recovered by the leachate collection system and sent to the water treatment plant [21]. Figure 10-17 shows a potential post decommissioning flow and water balance for the JEB TMF. The simulated flows vary from 0.2 m^3/day to 15.0 m^3/day depending on the tailings mass (using 1E-9 m/s to 1E-6 m/s). [4].

Figure 10-17 JEB TMF - simulated Post Decommissioning Flows (tailings hydraulic conductivity = 1E-8 m/s) [24]



During the consolidation stage, occasional recontouring of the mound will be necessary to maintain a well-drained surface. When consolidation is complete, a cover of bentonite amended till would be placed to reduce surface permeability and the surface would be contoured with a layer of till. The resulting land form would be graded and revegetated.

BENEFITS and LESSONS OF IN-PIT DISPOSAL AT McCLEAN LAKE MINE

The 1997 report by the Joint Federal-Provincial Environmental Assessment Panel on Uranium Mining Developments in Northern Saskatchewan had described the JEB pit as a risky proposal [26]. The Panel had identified concerns with arsenic and other contaminants leaching out of the tailings into the ground water. The proof of stabilization was presented in detailed laboratory work and has been confirmed through site monitoring. The McClean in-pit disposal of tailings and waste rock has been verified as a preferred option for the disposal of high grade uranium tailings. The key to the success of the in-pit disposal is the detailed understanding of pit and waste characteristics and the precise engineering of both the pit containment and the wastes placed in the pit. Very few changes had to be made to the initial design of the JEB Tailings Management Facility. Some are listed in Table 10-2.

RESIDUAL ISSUES

No significant residual issues remain; there is a high degree of public and regulatory acceptance of this in-pit disposal technology.

WASTE ROCK DISPOSAL IN SUE C PIT

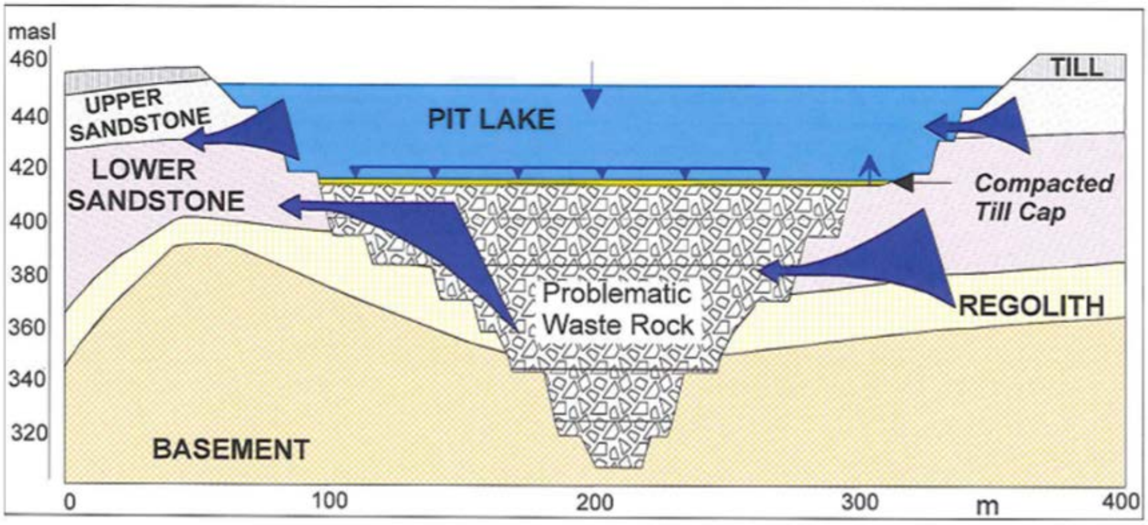
Waste rock was segregated to minimize the volume of problematic ARD and metal leaching waste rock on surface. Testing to characterize the waste rock, which was to go into a disposal facility, included sequential leach tests and column testing. Rock was leached using a water to solid ratio of 20:1 (50 g/ 1L) with agitation. The leachable concentration of arsenic was found to be independent of age and degree of oxidation, and for uranium and nickel it was highest in aged samples [19]. The saturated column tests used a minimum volume of water (0.15 L), sampled bi-weekly and submitted for chemical analysis. The correlation between pore water concentrations versus solid content was found to be independent of the age of the rock samples [19]. Waste rock was segregated based on sampling blast hole cuttings using a traditional approach of radiometric scanning for uranium and XRF analyses of As and metals [19].

The Sue C pit has been used for disposal of mineralized or ARD waste rock (Figure 10-18 and Figure 10-19) [22]. Mining of the Sue C orebody was completed in February 2002 and all of the ore was stockpiled on the surface. Low grade special waste from the mining of JEB, Sue C, Sue A, Sue E and Sue B deposits were disposed of in the mined-out Sue C pit. The Cigar Lake joint venture also has an agreement to dispose in the Sue C pit special (mineralized) waste from the underground mining development and future operations (1.3 Mm^3 has been placed in the pit from Cigar Lake as of September 2010 [27]) [7]. For the short term the waste rock is being maintained under water to reduce metal leaching and ARD, but will have a compacted till diffusion barrier between the water cover and wastes for the long term. (Figure 10-20).

Figure 10-18 Waste Rock Placement in the Sue C Pit [22]

Figure 10-19 Water-covered Waste Rock in the Sue C Pit [22]

Figure 10-20 Closure Scenario - Sue C Pit [22]



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CASE STUDY #11 LICHTENBERG OPEN CAST[25] MINE

SYNOPSIS

Located in the Ronneburg mine field in eastern Germany, the Lichtenberg "open cast" (open pit) mine was in production from 1958 to 1976[26]. The pit was used as a disposal location for various mine wastes from 1976 until 1990 - the year of Germany's reunification and ending of uranium production in eastern Germany. The Lichtenberg mine produced 13,000 tonnes of metallic uranium and 160 Mm^3 of waste rock.

The Lichtenburg pit was backfilled with acid generating and metal leaching waste rock from open pit and underground operations of the Ronneburg mines. This backfilling was a key part of the most comprehensive and costly multi-mine site and tailings rehabilitation to date anywhere in the world.

The Lichtenburg pit had a surface expression of 160 ha with a maximum depth of 240 m, which allowed for the accommodation of a large volume of material. Between 1991 and 2007, the pit was backfilled with waste rock totalling more than 125 Mm^3. A small part of the former pit area still receives mine waste from final remediation and clean-up work of the Ronneburg site.

The in-pit disposal was planned and executed in a way to best minimize the risk of contaminating groundwater. The waste rock surrounding the mine was classified into three categories of acid generating potential. The pit was also theoretically sectioned vertically into three zones from the bottom of the pit to the top, based on oxygen penetration. Zone A was at the bottom of the pit below the expected final water table and would be an anoxic area when filled. Waste rock with the highest acid content and potential to generate acid was placed in Zone A. Zone B was the second zone from the bottom but still below the oxygen penetration depth, and it was filled with waste rock containing a mixture of neutral and potentially acid leaching rock. The upper-most zone, Zone C, was within the oxygen penetration depth and was filled with neutral waste rock. The upper layer of Zone C contained high proportions of soil, and was topped with a final cover layer of soil so as to support vegetation. The top of the backfilled pit was at a higher elevation then the original pit height. The Lichtenberg pit and surrounding

[25] "Open Cast" – European term for Open Pit.

[26] The dates refer only to the open pit. There was a "Lichtenberg" underground mine, which was opened in 1950 as an independent UG mine, was reunified with the Reust mine in 1962, and became in 1988 part of the Schmirchau UG mine. The underground workings intercepted the pit walls.

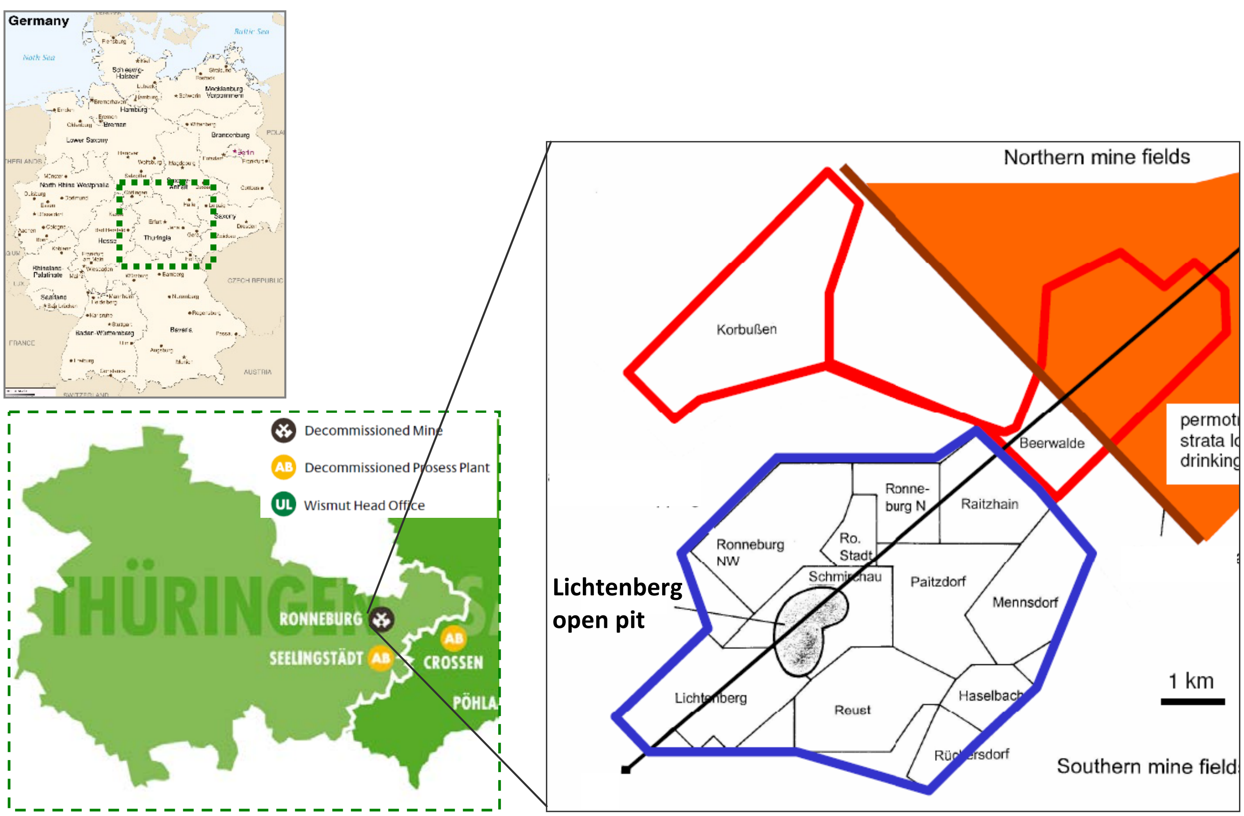
area was mainly reforested with areas of grassland, and is planned to be open to the public once surface remediation is complete.

In-pit disposal at the Lichtenberg site has been successful based on monitoring information to date. The main characteristics of the current ground water is a pH neutral water, high in sulphate levels, but with contaminants of concern generally immobilized.

LOCATION

The Lichtenberg Mine site is in the Ronneburg mine field and is immediately adjacent to the town of Ronneburg in the state of Thuringia, in eastern Germany (Figure 11-1) [23][3]. One of the most important drainage systems of the region is the catchment area of the Gessenbach Creek located in the western part of the district between the cities of Ronneburg and Gera. The Gessenbach Creek and its tributary Badergraben, follow the valley in a western direction [4]. The upper part of the catchment area near the city of Ronneburg is influenced by the mining activities. The Ronneburg mine site is part of the Weiβe Elster watershed. The Lichtenberg Mine is at an altitude of 300 – 350 mNN and receives an average of 650 mm of precipitation a year, mostly as rain [20].

Figure 11-1 Location of the Lichtenberg Open Pit and Mine Site in the Ronneburg Mine Field, in the state of Thuringia, Germany [5][24]



MINE TYPE AND HISTORY

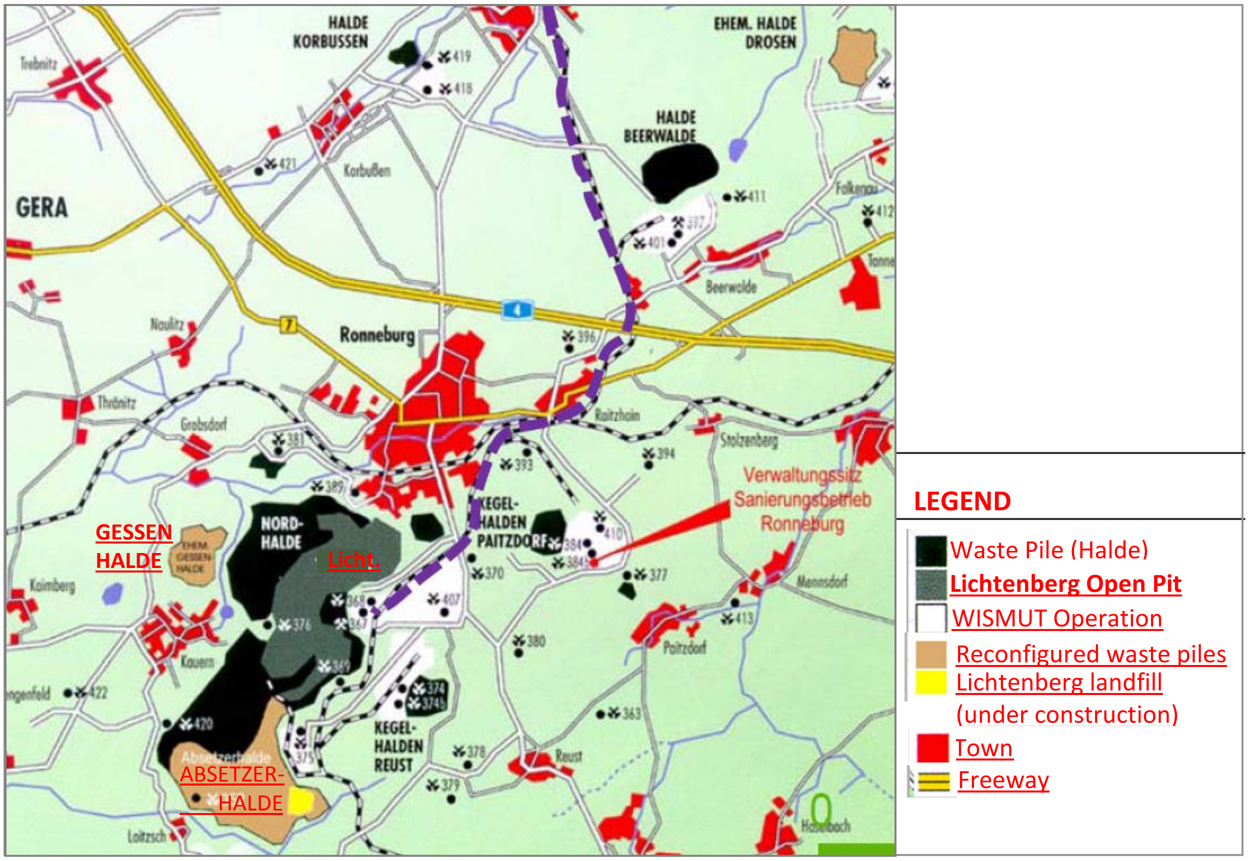
Mining at the Lichtenberg open pit mine began in 1958, as a part of the Ronneburg uranium mine complex in East Germany, and was mined until 1976 [24]. Operated by the bi-national Soviet-German SDAG Wismut, the Lichtenberg Mine started out as an underground mine with operations starting in 1950, although spontaneous fires temporarily halted mining and production. The fires were caused by the presence of very reactive iron sulphides in a carbon-rich shale. In 1962, the independent mine was reunified with the Reust Mine and became part of the Schmirchau Mine in 1988.

The total rock excavated from the Lichtenberg open pit was 160 Mm^3 from which 13,000 t of metallic uranium was produced [24][30]. The mine waste rock was piled around the pit and two of the larger waste dumps were the Absetzerhalde and Nordhalde dumps (Figure 11-2).

Open pit mining ceased in 1976 when the pit reached the then town boundary of Ronneburg[27]. Mining was also halted because the expanding open pit "over-mined" the former underground drifts of the Schmirchau mine and created a difficult situation for proper underground ventilation. At this point, ventilation was also becoming limited for the large, deep Lichtenberg pit. The final solution for sufficient pit ventilation required the use of jet engines from the Soviet military.

[27] A large part of the original town of Ronneburg had been moved to accommodate open pit mining in the late 1950's.

Figure 11-2 Lichtenberg Open Pit, and Waste Piles of the Mine Site and Piles of the Ronneburg Mining District [16]



The Ronneburg mining district consisted of 16 waste rock dumps (including the three from Lichtenberg: the Gessenhalde, Absetzerhalde and Nordhalde waste rock piles) with approximately 200 Mm^3 of mostly acid generating waste rock and approximately 1000 ha of contaminated areas. The mines within the Ronneburg complex totalled more than 40 major shafts and adits with a combined length of 1,043 km, and open mine workings of 26.7 Mm^3, excluding the Lichtenberg open pit [5].

Large amounts of groundwater were pumped out of the mines to keep them dry during mining operations. Water pumped from the Lichtenburg pit (which was interconnected to underground mine workings) was released into local streams, negatively affecting water quality. At the end of operations, high levels of radium and uranium of about 3000 Bq/kg[28] were detected in the sediments of rivers in the Ronneburg area. These levels were a 100-fold increase over natural background levels. The southern main part of the mining area is located in the Variscan basement of the Ronneburg Horst. This area is drained by several small streams, some of which

[28] Bq/g: Bq - Becquerel is a unit or radioactivity; 1 Bq/g = 1 disintegration per second/gram.

dried up due to the mining activity but may again be water-bearing after flooding of the underground mines. Flooding of all but one Ronneburg mine was close to complete in 2011 [24].

Following the completion of open pit mining in 1976, the Lichtenberg pit was used as a waste disposal area until December 1990 when all mining ended at the site [30]. The closure of mining operations also followed the reunification of Germany in 1990 [5]. About 76 Mm^3 of waste rock had been placed in the worked out part of the Lichtenberg pit as an internal storage dump before the larger remediation plans were initiated [24]. In 1991, the former SDAG Wismut was legally transformed into a remediation company with the Federal Republic of Germany as the only shareholder. Wismut GmbH currently manages the decommissioned former uranium mining and milling facilities and mining-disturbed lands [24].

Most of the waste piles from the Ronneburg mining district produced drainage waters contaminated by heavy metals, acidity and radionuclides. The combination of reactive sulphides and carbon-containing shale led to continuous fires, especially in portions of the Absetzerhalde waste pile. Excavated contaminated soils from the remediation of the Ronneburg area that occurred between 1997 and 2001, also contributed to the waste piles [5]. Some of the contaminated soil was biologically treated to remove hydrocarbons [5].

Remediation of the Ronneburg mine site involved the relocation of all sixteen waste rock piles (with the exception of pile #381 and the Beerwalde dump) into the mined-out Lichtenberg pit [30][24].

Backfilling of the pit took place between 1991 and 2007 with waste rock originating from surface and underground mining that had accumulated around the open pit, including the Lichtenberg waste piles [20][21]. The pit was filled with a total of more than 125 Mm^3 of waste rock [31]. Lime was added to the acid-generating waste rock during excavation and loading, to neutralize acidity and potential acid generation (PAG).

Backfilling of the Lichtenberg pit was performed in a strategic manner according to acid content and acid-generating potential. The bottom of the pit, Zone A was located below groundwater level after mine flooding, and was filled with the most contaminated waste rock that had the highest potential to generate acid. Zone A became an anoxic layer after backfilling. Zone B, located above Zone A, was filled with waste rock that possessed an uncertain potential to generate acid. Zone C comprised the top layer of waste rock with a very low potential to generate acid.

The backfilled pit covered a surface area of 210 ha. The top layers consisted of neutral waste rock and soils, later landscaped as part of a public agricultural exhibit, as seen in Figure 11-3. Waste relocation was basically completed in 2008. Placement of the cover materials and re-vegetation were planned for completion in 2010 [28].

Figure 11-3 The Remediated Lichtenberg Pit as Part of the Federal Horticulture Exhibition, 2007 [12]

Due to the radioactivity content of much of the Wismut waste, the *Atomic Act* and subsidiary regulations, such as the Radiation Protection Ordinance, were applied to the site's remediation [12]. The specific remediation goals followed the legal requirements under applicable national and state radiation protection and/or water resources laws and regulations that included the *Mining Act*, *Water Resources Act* and *Soil Contamination Act* [17]. All remedial operations at the Ronneburg site are anticipated to be finalized in 2015 [5].

PIT DESCRIPTION

At its maximum size, the Lichtenberg pit measured 0.6 km wide, 1.6 km long, a depth of 240 m, with an volume of approximately 84 Mm^3 [30][2][7]. In 1990, after being partly refilled, the depth of the pit had decreased to 160 meters (Figure 11-4) [24]. Many of the pit walls were unstable, and a fault zone running though the pit had caused considerable slope failure at the beginning of remediation activities [3].

Figure 11-4 The Lichtenberg Open Pit in 1990 (partially backfilled) [26]

Engineered cells within the pit body accommodated contaminated soils from the Ronneburg remediation. These cells were also used to immobilize water treatment residues from the Ronneburg water treatment plant [5]. Figures 11-5 and 11-6 below show the Lichtenberg pit during operations.

Figure 11-5 Lichtenberg Open Pit Mine During Production, circa 1960's [30]

Figure 11-6 Lichtenberg Pit in 1962 [20]

PIT MINERALOGY, GEOLOGY AND HYDROLOGY

The north-east boundary of the Ronneburg geological fault body, or horst, was formed by an extensive NW-SE-striking fault zone (the Crimmitschauer fault) [2]. The uranium deposits of Ronneburg were strata-controlled, structure bound deposits consisting of intensively folded and faulted incompetent and competent rocks. This deposit featured uranium concentrations in high density small-scale brittle structures (fissures, joints, faults) which formed stockworks within or immediately adjacent to carbonaceous, pyritic black shales. The Paleozoic host rocks mainly consist of argillaceous and siliceous black shales with intercalations of dolomitic and phosphorite nodule beds (Silurian “Graptolithenschiefer”). The main black-shale horizon lies below Ordovician carbonaceous sandy shales and overlies Silurian carbonate rocks. Some Devonian metabasaltic dikes and sills cut the metasedimentary rocks. The waste rock from the pit mainly consisted of black shales, metabasaltic rocks and carbonate of the Ordovician to Devonian age.

The pit rock contained up to 7 wt% sulfides, 5-9 wt% organic carbon, 40-60 ppm uranium and a series of trace elements (Ni, Mo, Zn, Co, Cu, REE) [4]. The uranium ore at Ronneburg is thought to have formed by a combination of hydrothermal and supergene processes. This ore consists mainly of pitchblende and coffinite and is accompanied by sulfide mineralization of mainly pyrite, and minor cobalt and nickel arsenides [24].

Water pumped from the mines during operation was released into nearby rivers and creeks. In the sediments of rivers in the Ronneburg area, concentrations of radium and uranium around 3000 Bq/kg were found, indicating up to 100-fold increases over natural background levels. North of the Crimmitschauer fault is the Variscan basement hosting the ore deposit overlain by platform sediments of the Permian and Triassic ages. These platform sediments contain several important groundwater aquifers; two upper platforms are used for local drinking water supply [24][1].

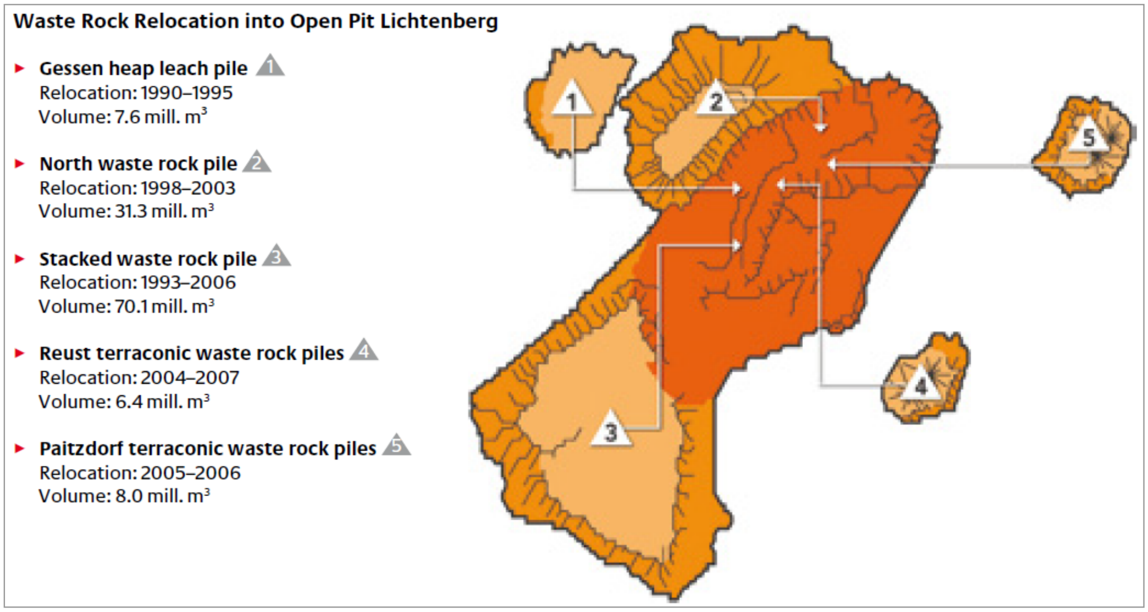
MATERIALS PLACED IN THE PIT

Remediation of the Ronneburg mine site involved the relocation of all waste rock piles generated within the Ronneburg mining district south of the A4 motorway (with the exception of pile #381), into the mined-out Lichtenberg pit [30][24]. The remediation plan also involved the creation of a new pile of waste material at the site of the open pit. An important advantage gained through the relocation of waste rock was the containment of acid generating waste rock piles into one area, reducing the area and number of contaminant seepage problems. Backfilling of the mine pit also provided the benefit of stabilizing the pit walls without special geotechnical measures [12].

After waste rock relocation, the waste dump land footprint was reduced from 460 ha to approximately 250 ha, with the reclaimed land then available for other uses. To date (2013), the Drosen and Korbuβen mining units, the areas of the original waste rock piles #370 and #377, and the Gessenhalde have been fully reclaimed [5].

Initially 7.5 Mm^3 of waste was dumped in the Lichtenberg open pit when Wismut moved the Gessenhalde pile into the pit. Gessenhalde had large volumes of acidity and potential ARD [24]. This activity occurred between 1993 and 1995 and included lime treatment of the waste rock. A rapid relocation of the Gessenhalde pile was necessary because approximately 6 Mm^3 of low grade ore in this former heap leach pile had to be contained urgently (parts of this pile were burning when exposed) [12]. After the planned flooding of the pit, this material would be located below the water table [24]. Figure 11-7 shows the relocation of the waste rock piles into the Lichtenberg open pit during remediation.

Figure 11-7 Relocation of Waste Rock into the Open Pit Mine of Lichtenberg at Ronneburg [5]



As outlined in Table 11-1, the relocation of the two largest Wismut piles (Absetzerhalde and Nordhalde) occurred between 1993 and 2006 [3]. In 1998, approximately 27 Mm^3 of waste rock from the Nordhalde dump was relocated into the pit. The deposit of waste rock consisted partly of acid-generating waste rocks. Before relocation of these waste rocks, highly mineralized and acidic seepage waters were detected to be draining diffusely into the adjacent creeks and surface Quaternary sediments [4]. During operations before 1990, the Lichtenberg pit was filled with approximately 76 Mm^3 of rock. This rock was from the Innenkippe dump which was largely from the extension of the open pit between 1972 and 1977, and the Schmirchau balcony which was waste rock from the neighbouring Schmirchau underground mine [22].

Figure 11-8 Partly Backfilled (with Innenkippe pile) Lichtenberg Pit and Ronneburg Mine Site Before Remediation [20]. (Note waste rock piles in the background).

The Absetzerhalde pile contained 65 Mm^3 (approximately 120 Mt) of black shale, limestone and diabase, ranging in age from the Ordovician to Devonian periods. As in other waste rock piles in the Ronneburg district, the acid-generating mineral is pyrite, present at concentrations of up to 7%. Dolomite and calcite are the principal neutralizing minerals, with total carbonate mineral contents ranging from zero up to 35% in some zones. Uranium, other radionuclides, and several heavy metals were the contaminants of concern. Seepage from the Absetzerhalde waste rock pile demonstrated a pH ranging from 2.5 to 3.1, sulphate concentrations from 7,000 to 25,000 mg/L, and uranium concentrations from l to 8 mg/L [12]. Liquid hazardous wastes that had been disposed in the Absetzerhalde pile needed to be removed and disposed separately, before this pile could be reclaimed. Wismut built a special toxic waste facility for this material on its Ronneburg premises in 1995 [3].

The Schutzdamm Ronneburg mine dump was the last pile to be relocated to the Lichtenberg open pit. This occurred in early 2008. Some of the Ronneburg Mine waste rock piles were remediated instead of relocated into the open pit, and are listed in Table 11-2.

Table 11-1 Relocation of Waste Rock Piles into the Lichtenberg Pit [2][13][12][5][22]

Waste Rock Pile	Volume ($10^6 m^3$)	Base area (ha)	Pile surface area (ha)	Relocation Period	Characteristics
Relocation into Lichtenberg open pit					
Innenkippe (placed in-pit during mine operations)	64	70	-	1964-1989	Acid/neutral
Gessenhalde (acid leach heap)	7.6	28.7	-	1990-1995	Acidic and pyrophoric
Absetzerhalde	70.1	224.7	239	1993-2006	Acid generating
Nordhalde	31.3	83.9	91	1998-2003	Acid generating
#370	1.4	6 (8.1 after reshaping)	7 (8.3 after reshaping)	2003	Neutral
#377	0.42	2.9	3.2	2004	Neutral/slightly acidic
Halde Reust	6.4	20.5	24	2004-2007	Neutral
Halde Paitzdorf	7.96	24.9	28	2006	Neutral (very high sulphate discharge, evidence of fires in 1990)
#4	0.9	7.8	9.6	2006-2007	-
Schutzdamm Ronneburg	0.15	2 .0 (4 after reshaping)	2.3 (4.6 after reshaping)	2007-2008	-
Schurf 12/13	0.016	0.3	0.4	1997	-
Schmirchauer Balcony (placed in-pit during mine operations)	12	44.5	-	-	-
Diabashalde	0.16	1.2	-	2002	-
Total	**190.4**	**456.1**	**684**	**1990-2008**	-

Table 11-2 Relocation of Remediated Waste Rock Piles at Ronneburg (N/A- Not applicable) [2][13][12][5][22]

Waste Rock Pile	Volume ($10^6 m^3$)	Base area (ha)	Pile surface area (ha)	Relocation or Remediation Period
Relocation to Beerwalde waste rock pile				
Korbuβen	0.4	4.1 (6.5 after reshaping)	4.3 (7.5 after reshaping)	2000-2001
Drosen[1]	3.5	22.8	24	1997-1999
Total	**4.5**	**29.5**	**31.5**	**1997-2001**
Remediation in situ				
Beerwalde	4.5 (8 incl. Drosen)	23.9 (~35 incl. Drosen)	25	1997-2003
#381	0.4	6.2	6.5	2004
Total	**5.54**	**31.2**	**31.5**	**1997-2003**

1-Neutral acidity but very high sulphate discharges, indicating high sulphate oxidation rates

Several hundred thousand tons of debris within the waste rock (including scrap metal from industrial areas, soils, sludges from settling ponds, and landfill [18]) mixed with radioactive rock and debris contaminated with hydrocarbons, presented a complex range of challenges for the Lichtenberg site [12]. Due to the high pyrite content of portions of the rock, most of the waste rock was acid generating and some of it was burning. The acid mine drainage (typically pH = 2.5 to 3) carried high uranium, sulphates and heavy metal (such as Ni) concentrations, and demonstrated a very high water hardness. This was due to the rapid oxidation of sulphide minerals, enriched especially in black shales containing high amounts of carbon [24]. Radon emanation was also a radiological concern [12].

Neutral seepage was detected from some of the waste rock piles that contained elevated uranium and nickel concentrations, and very high sulphate levels. This occurred in heaps where the acid generated by the sulphide oxidation process was sufficiently buffered by magnesium-bearing carbonates (e.g., dolomite).

METHOD OF PLACEMENT IN-PIT

From 1990 until 2008 a total volume of 131 Mm^3 of waste rock and related mine waste had been placed into the pit [24]. To handle such volumes of waste rock efficiently, the largest fleet of Caterpillar equipment in Europe was put to work at Ronneburg (Figure 11-9).

Figure 11-9 A Truck Fleet Used During the Waste Rock Pile Relocation into the Lichtenberg Open Pit (1998) [5]

The annual volume relocated from 1996 to 2006 was approximately 10 Mm^3 at a rate of 40,000 m^3 of waste rock per day. The waste rock relocation was carried out by two fleets of dozers (CAT Dl IN), front-end loaders (CAT 994B) and heavy haul trucks (CAT 785 and 773) (Figures 11-10 and 11-11). The scale of the equipment and the target productivities made the relocation very similar to medium scale open pit mining [12].

The acid-generating waste rock was blended with lime in a ratio required to neutralize pore waters in the flooded waste rock (rates ranging from 1:150 to 1:600). The waste rock was placed in thin layers (60 or 120 cm lifts) and compacted by heavy trucks to reduce the hydraulic conductivity (field conductivities of 10^{-7} to 10^{-8} m/s were achieved) and the oxygen transport in the unsaturated material as well as to avoid future cover failures due to settling (expected to achieve a cover failure area of approximately 15%) [12][24]. Loaders were equipped with GPS, allowing the position of each truckload placement to be recorded [12].

Figure 11-10 Early Stage of Backfilling the Lichtenberg Pit (1990, G. Feasby)

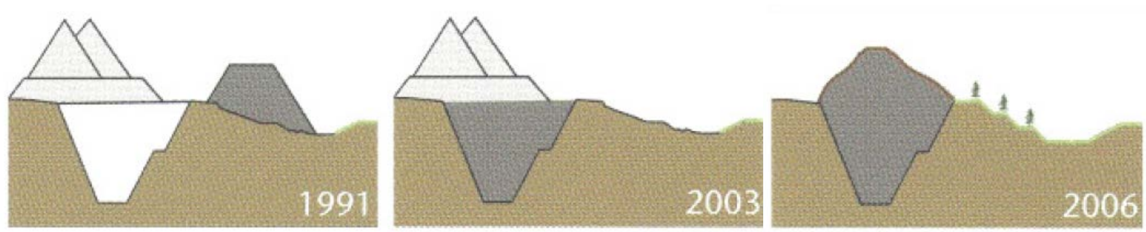
Figure 11-11 Advanced Stage of Pit Backfilling (2002, M. Paul) [16]

SELECTION OF IN-PIT DISPOSAL STRATEGY

To accommodate the placement of more than 130 Mm^3 of waste rock and clean-up material into the Lichtenberg pit, which only had a free volume approximating 84 Mm^3 during remediation, the pit was constructed as a "high rise pit"[29]. This configuration allowed for containment of the full volume as depicted in Figure 11-12 below.

[29] Term noted by SENES Consultants (team member R. G. Dakers) engaged by Government of Germany.

Figure 11-12 The Changing Contour of the Lichtenberg Pit During Remediation and Backfilling[30]



To optimize pit backfilling of waste rock destined for Zones A, B and C within the pit, the relocation was planned and controlled with methods similar to those used for grade control in open pit mines. Long-term and medium-term planning used waste pile drill hole data and "kriging" estimates. Short-term planning was carried out one to three months ahead of actual backfilling, and included a detailed delineation of material types in terms of their acid generating potential. To provide the information needed for the more detailed delineation, test pits were excavated on a 25 m grid and samples were characterized using the paste pH and NAP (net acid potential) pH methods. Where quicklime (CaO) addition was required, a paste conductivity correlation was used to estimate lime addition rates. In general, the relocation was carried out according to the short-term plan. Quality control samples were also taken from the excavation face to confirm material classifications and to allow correction of the short term plan where necessary [12].

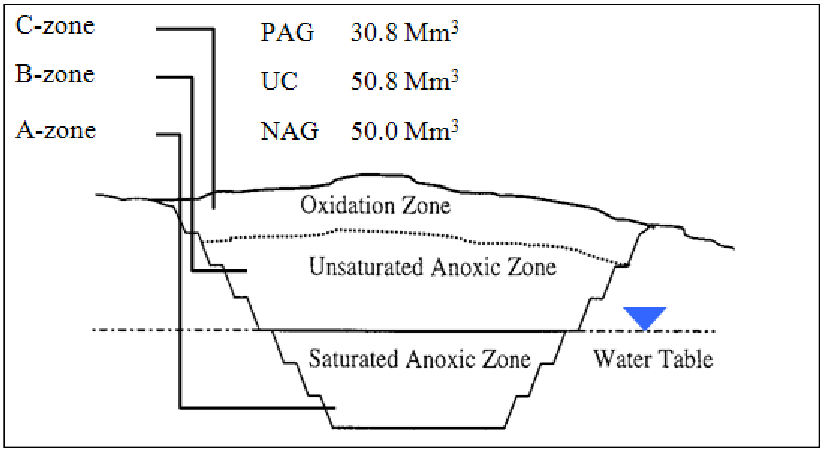
Figure 11-13 Test Pits on one of the Waste Piles in a 25 m by 25 m Grid [18]

[30] Jenk, U., S. Mann and M. Paul. *Recent Status of the Wismut Remediation Project.* 2008.

The sequence of the waste rock placement into the open pit depended on the free acidity and degree of acid generating potential of the waste materials. Waste rock having the highest acid-generating potential was placed below the anticipated post-flooding groundwater level [7]. Targeted placement in the pit allowed the minimization of contaminant release into the groundwater. The waste rock was placed in three Zones as shown in Figure 11-14:

- **Zone A** was the deepest zone in the pit. This zone held *Class A* material that was already acidic or had a potential to generate acidic drainage. The neutralization potential and acid-generating potential ratio of *Class A* material was NP:AP <1 [9]. The paste pH was <4.5, and the NAP pH was <4 [28].
    - After pit backfill and groundwater recovery, this zone would be situated below the water table and the anoxic conditions became the long term control of the potential acid generation through lack of oxygen penetration to this depth. To prevent the short term release of acidity already present in the rock, quicklime was added during relocation. Lime was added to the Class A-material at a proportion of about 1-3 kg of CaO per tonne [22].
- **Zone B** was located directly above Zone A in elevation, and held *Class B* material which had an uncertain potential to generate acid drainage. The net acid potential conductivity of this Class was NAP-conductivity ≥5 [28]. The NP:AP =1-3; ≥4 NAP-pH <4 [9][28].
    - Zone B was located above the water table but still below the depth of oxygen penetration from the ground surface.
- **Zone C** comprised the top layer of the pit and represents an oxidation zone. This zone held Class C material that had a very low potential to generate acid. The *Class C* NP:AP =1-3; NAP-pH ≥5, and NAP-conductivity <5 [9][28].
    - Oxidation of sulphide minerals would occur in Zone C, but the oxidation products would be neutralized by reaction with the abundant carbonates present.

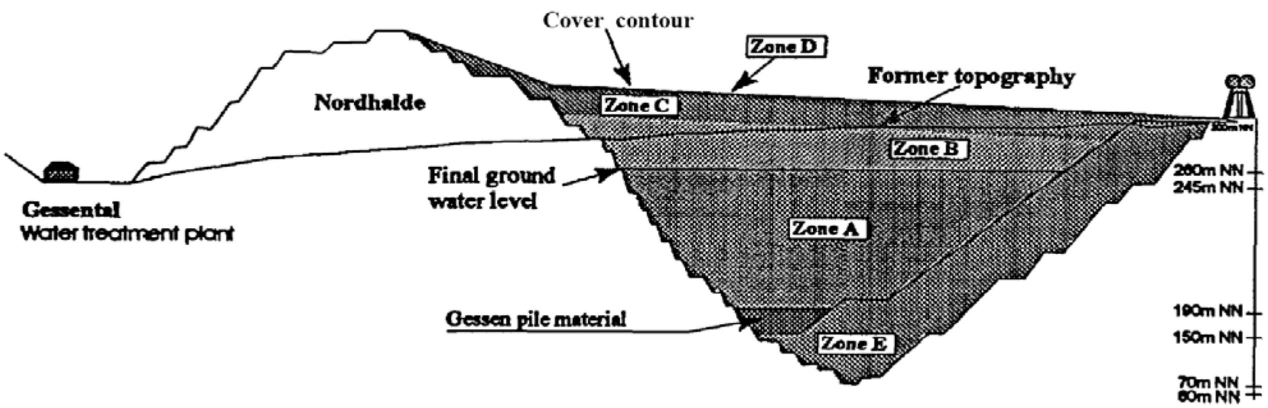
Figure 11-14 Waste Rock Volumes Placed into Each Backfilled Zone of the Lichtenberg Open Pit Mine; PAG-Potentially Acid Generating Material, UC- Material of Uncertain Acid Potential, NAG- Non-Acid Generating Material [12]



According to the relocation plan, waste piles containing greater than 70% of Class A samples were located within Zone A, piles containing 80% of Class C samples and less than 10% of Class A samples were placed in Zone C, and piles containing samples outside of these criteria were placed in Zone B [28].

The Nordhalde waste pile was originally planned to be stabilized in-situ (Figure 11-15), but it was rescheduled to be placed into the Lichtenberg pit in 1998 to permanently isolate acidic wastes. Because of this change in plan after remediation had already begun, 8 Mm^3 of Class A material within the Nordhalde pile was relocated to Zone B instead of Zone A [28]. To contain this PAG material, the acidic waste had been underlain by an in-pit geochemical barrier, consisting of mixed power plant ashes with a sufficient neutralization potential [28]. Given that the site of the Nordhalde dump was to form part of the 2007 Federal Horticultural Show, relocation of the Nordhalde materials were prioritized for completion by 2003.

Figure 11-15 Schematic of the Original Lichtenberg Pit and Nordhalde Pile Remediation, Prior to 1997 [12]



Following completion of the backfill to a level up to 60 m above the initial ground elevation ("a high rise pit"), a dry cover was placed on top of the backfilled mine wastes. The plans for a final pit cover included the use of a multilayer system (infiltration barrier, storage, and topsoil areas). This system was designed to reduce infiltration and radon emanation, provide a substrate for vegetation of the area, and to allow collection of the overflowing contaminated flood water for treatment at an on-site facility [3].

The pit cover area comprised 220 ha, a footprint which is close to 40% more than the original pit opening of approximately 160 ha [2]. As a result of comprehensive studies and investigations, the final cover design was a 1.6 m combined cover of cohesive material from on-site excavation of old cover material of the Absetzerhalde and Nordhalde waste dumps, overlain by a 0.4 m thick layer of natural soil to provide a growing medium for re-vegetation of the site [21]. Approximately, 70% of the re-vegetation was accomplished through planting and seeding of the area [18]. The pit cover was completed in 2012. The resulting design included establishment of a 17 km trail network and more than 20 km of ditches to catch and control surface water run-off [31][30].

A key aspect of water management around the Lichtenberg pit involved collection and treatment of runoff water, as shown in Figure 11-16. The closure strategy for the Ronneburg mine field included the construction of hydraulic barriers of different types in order to minimize the mixing of waters with different water quality and to prevent water flow through the backfilled open pit to downstream catchments and ground water reservoirs. The water management strategy included selective plugging of preferred flow paths within exfiltration areas where mine water discharge was expected upon rebounding of the groundwater. In order to separate mine fields of differing water quality altogether, 117 hydraulic barriers were erected throughout the Ronneburg site [24][1].

Figure 11-16 Water Management and Flood Protection around the Lichtenberg pit [18]

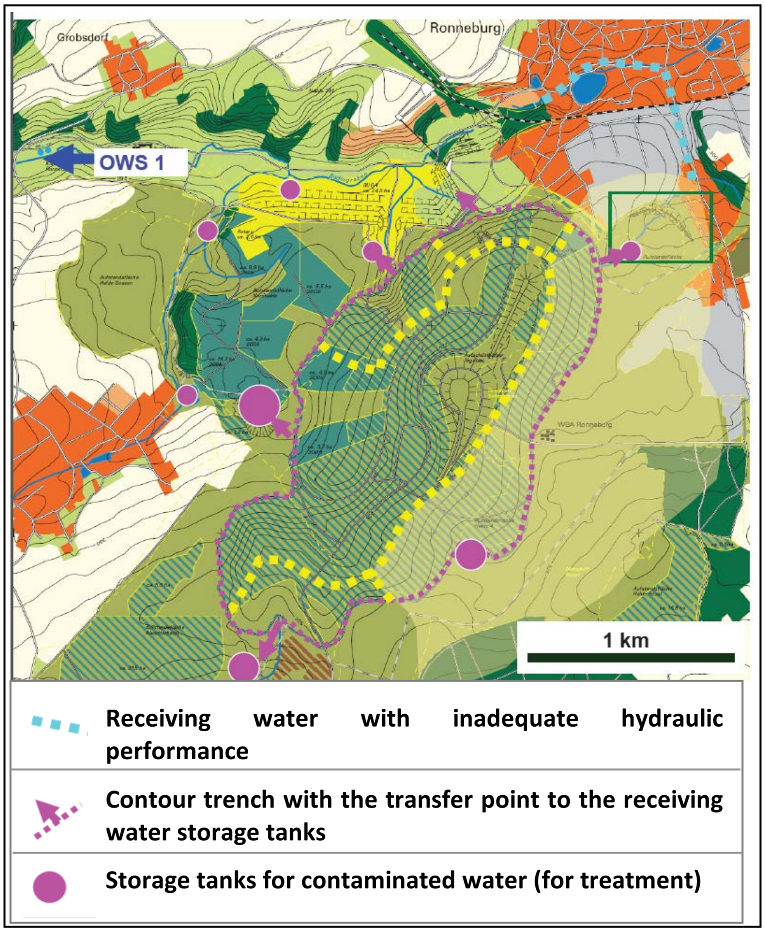


Figure 11-17 Lichtenberg Pit During Remediation - (1) in 2002 [7]; (2) & (3) During Back Filling [30]; (4) in 2005 [14]; (5) in 2006 [19] and; (6) in 2010 [1]

PERFORMANCE OF IN-PIT DISPOSAL

Pre-mining baseline measurements were not available for any of the Wismut sites, so the monitoring program for the Ronneburg area was developed on the basis of the initial 1990 characterization surveys and the predictive analysis of exposure pathways. The baseline monitoring measured the concentration and release rates for contaminants in surface and ground water, seepage from waste rock, air (particulates and radon), gamma radiation levels, soil and, in some instances, the human food chain pathway. The monitoring of liquid discharges well demonstrated the effectiveness of the first remedial measures at all Wismut sites: beginning in 1990 the discharges carried a uranium load of approximately 28 t (approximately 31 Mm^3), which decreased to a level of 4 tonnes of uranium per year in 1998. Since 1998, the decrease has been continuous and the uranium load amounted to approximately 3 tonnes per year in 2005 [6][12]. Based on mine area discharges to local watersheds, the remediation has contributed to a decrease in contaminants release, including uranium, as seen in Table 11-3 and Figure 11-18.

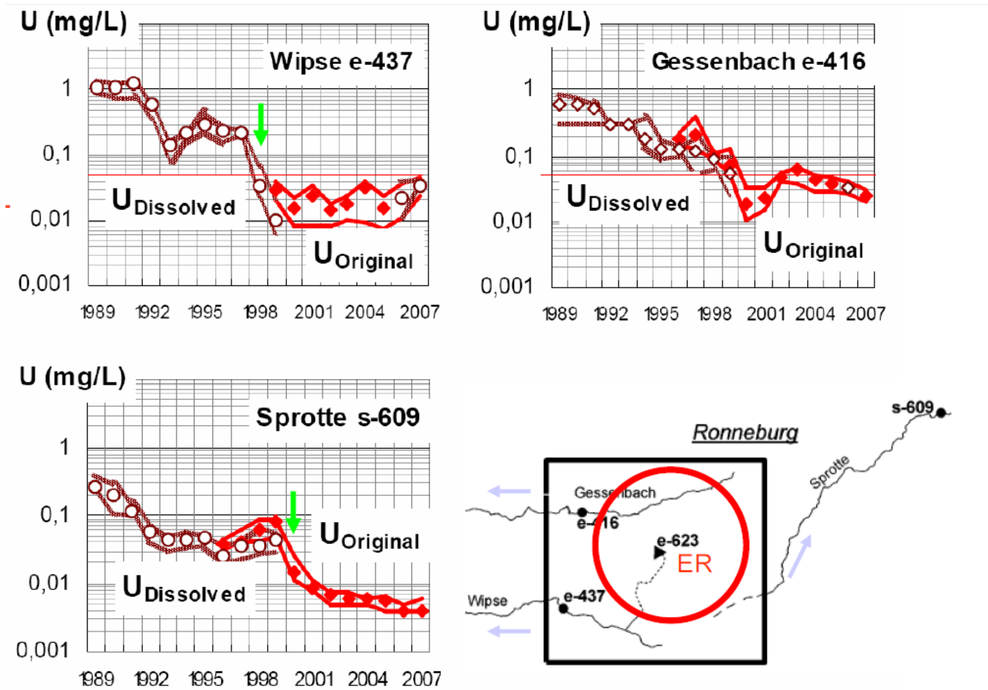
Table 11-3(a) Total Discharge for all WISMUT Sites [19]

	Q (million m^3)	**U (tones)**	**Ra-226 (GBq)**
1989	32.5	27.5	23.4
2006	16.5	2.4	0.3

Table 11-3(b) Uranium Discharge Per Watershed (tonnes per annum). The Lichtenberg pit resides in the Weiβe Elster watershed [19]

	Weiβe Elster w/ Pleiβe	**Zwickauer Mulde**	**Elbe**
1989	15.6	10.0	1.9
2006	0.2	2.1	0.05

Figure 11-18 Uranium Levels in Local Surface Waters of the Ronneburg Site. Note: Authorized levels of Uranium <50 µg/L. Median values, 25 and 75 percentile (ER-Environmental Remediation; Green arrow indicates the end of mining and commencement of remediation) [17]



REMEDIAL

Critics have commented that the final cover design allows for twice the infiltration rates from rainfall than would occur with a more impermeable cover design. This design was selected, however, on the basis of being able to sustain trees and achieve a 200-year durability criterion [3]. From the perspective of long-term performance, experience at other sites has demonstrated that highly engineered barrier covers can be susceptible to disruptions arising from differential settlement, cracking, root penetration, burrowing animals and human actions [12].

BENEFITS AND LESSONS OF IN-PIT DISPOSAL

The remedial investigations and feasibility studies (RI/FS) worked well to justify the selected remediation methods for the majority of area waste rock dumps. The remediation methods were chosen to manage acid mine drainage (Ronneburg), radon emanation, increased content of non-radioactive contaminants such as arsenic or insufficient geomechanical stability of the dumps (Schlema and other, smaller sites). The individual risk-based justification of remediation of the waste rock dumps (based on RI/FS) proved to be a relatively cost-effective approach without compromising the strict health and safety standards.

At the Ronneburg site, the most problematic waste rock was relocated into the close-by Lichtenberg open pit mine in order to avoid having a large number of acid seepage problems spread over an extensive area. Simultaneously, the backfilling of the open pit resolved the problem of the unstable pit walls.

REMEDIATION PROJECT COSTS

Wismut GmbH is a German federal company which, by 2010, had spent more than €5 billion (euros) for its environmental remediation work; this amount was part of the €7.1 billion the government had earmarked for the program [5]. Remediation in the state of Thuringia, containing the Ronneburg area and Lichtenberg Mine, incurred a cost of €2.9 billion by 2010. The remediation cost per kilogram of uranium produced has amounted to approximately $38 US. The costs were high due to a number of factors such as the low grade (low uranium content) and extensive acid rock drainage from the wastes, potential for radiological impact upon a densely populated area, and late development of the closure strategy [19].

RESIDUAL ISSUES

Wismut's monitoring system for the next 30 years includes the surveillance of environmental impact at all the sites including the Lichtenburg Pit. The goal is to measure overall environmental impact and to prove successful implementation of the remediation [20]. The long-term and post-remedial activities include water collection and treatment, maintenance of the restored land and ancillary mine workings, and long-term environmental monitoring [14].

The post-remedial phase (5 years) is planned to monitor and document the efficiency and success of the remediation measures, such as documenting and accounting for physical modifications that pit and waste covers will undergo [26]. The long-term activities over a 30 year period include water treatment (which will likely last more than 30 years), environmental monitoring, and care and maintenance of the restored lands.

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CASE STUDY #12 OMAI WENOT PIT

SYNOPSIS

The Omai Gold Mine (Omai) was a large scale 20,000 tpd gold mine operation located in Guyana, South America. At Omai two pits were developed, the Fennel and Wenot pits. In-pit disposal at the Omai gold mine was a 3-fold activity:

1. Tailings slurry and pond water were diverted into the active Fennel pit immediately following an emergency caused by a tailings embankment failure in 1995;
2. Tailings were deposited into the mined out Wenot pit during 2002-05; and
3. In 2004-05 pit tailings water from Wenot was discharged into the Fennel pit to temporarily maintain hydrological isolation of cyanide-containing water in Wenot.

All three activities were successful in meeting the objective of containing cyanide-contaminated waters. Acid drainage and metal leaching were not key concerns at Omai. Sulphides zones in the ore were pyrrhotite rich and they reacted in the leaching process to form thiocyanates.

For the last 10 years of operation, waste and water management at Omai was driven by the need to maintain extremely low total cyanide concentrations (<0.005 mg/L total cyanide or TCN) in any effluent reaching the local environment. This restriction had a basis in the notorious 1995 spill of tailings water into the local Essequibo River.

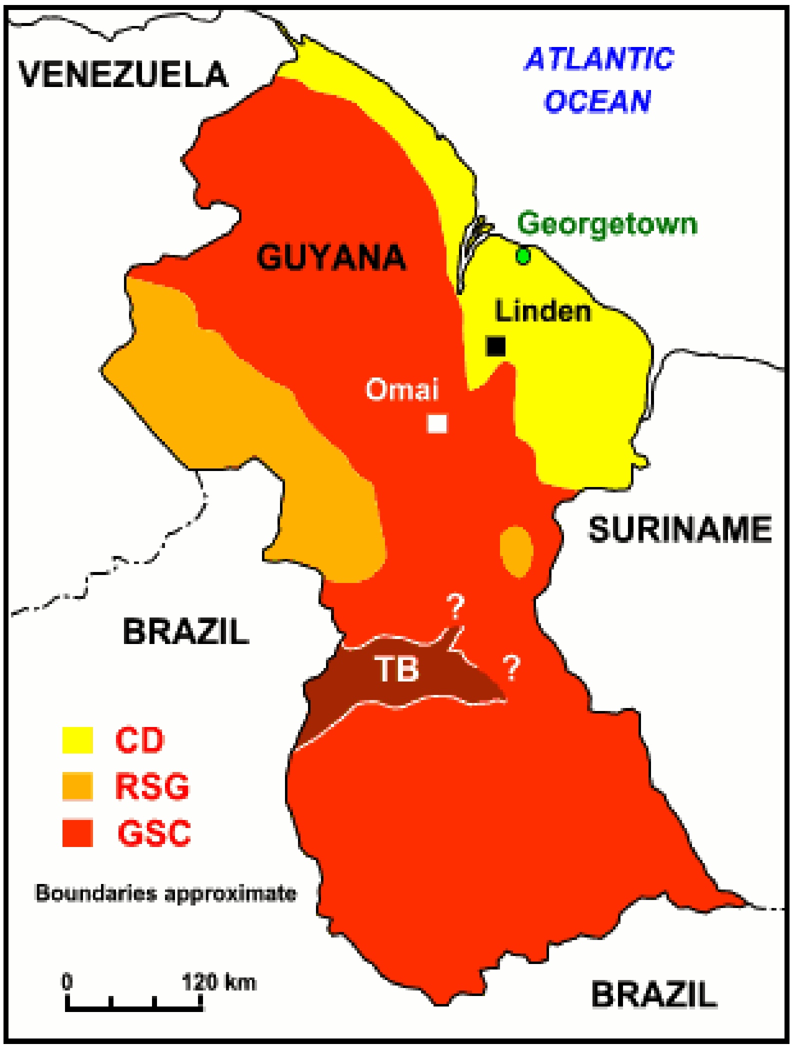
Pit disposal of tailings was practiced by discharging at the edge of the Wenot pit as 50% solids slurry. The tailings solids were allowed to settle without chemical aids and the decant water was recycled to the mill or sent for disposal in the Fennel pit. During operations, and following the end of operations in 1995, the pit water was monitored for ammonia, total suspended solids (TSS) and TCN. TCN concentrations fell below discharge criteria within 6 months following cessation of tailings discharge and the Wenot pit was allowed to fill naturally and overflow to the environment.

Cyanide degradation in a tropical environment is significantly enhanced by exposure to the intense sunlight at equatorial latitudes. This degradation was observed to be inhibited by high TSS in tailings waters, which is frequently encountered in the tropics when processing near surface ore zones that are hosted in saprolitic soils.

LOCATION

The Omai Gold Mine operated in a dense tropical rain forest. The Omai site is located 6^{o} north of the equator in central Guyana, which is an English-speaking country on the north coast of South America. The closed out site is located in a dense tropical forest where daily temperatures range between 20 and 40^{o}C, annual rainfall exceeds 2 metres per year and annual evaporation is about 1.4 metres. The mine location and area general geology are shown in Figure 12-1.

Figure 12-1 Geological Map of Guyana Omai Gold Mines Location [4]



Legend	
Coastal Deposits (CD) (Cretaceous - Recent)	Rocks of the Takutu Basin (TB) (Jurassic - Cretaceous)
Minor Basic Dykes – not on map (Triassic - Jurassic)	Rocks of the Guiana Shield Complex (GSC)
Includes the Roraima Supergroup (RSG) (Precambrian)	Linden - Bauxite Mine Omai - Omai Gold Mine

MINE TYPE and HISTORY

The Omai gold mine operated from 1992 to 2005, and produced over 3.5 million oz of gold from 70 million tonnes of hard rock ores mined from 2 large pits. The deposits had been exploited in the early 1900's by a German consortium and later by local placer miners, but it took until the late 1900's and the application of large scale mining and processing technology to produce an economic operation.

In August 1995, a tailings embankment failed, spilling over 2 million cubic metres of cyanide containing water into a small local river, the Omai River (Figure 12-2), which after a short distance drains into the large Essequibo River (Figure 12-3).

Figure 12-2 Omai River [1]

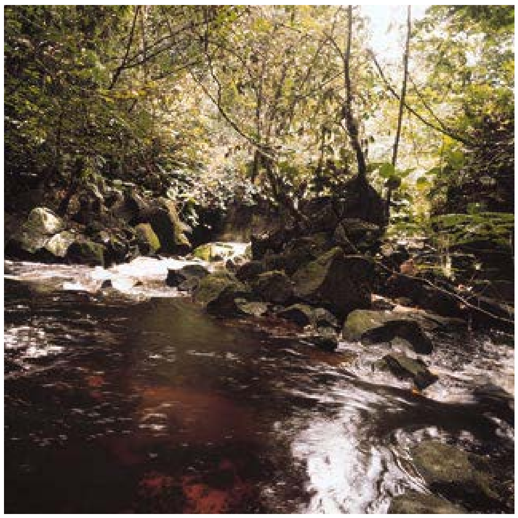


Figure 12-3 Essequibo River, Guyana [1]

The tailings water contained cyanide concentrations (>13 mg/L)[31] that were lethal to fish in a 1.5 km section of the small Omai River, but the dilution in the Essequibo River was so high (>10,000:1) that cyanide concentrations were non-lethal and barely detectable. However, the erosion of the tailings embankments was extensive and the red saprolite[32] soil plume was visible for over 100 km downstream. This led to considerable public concern and the government withdrawing permission to operate the mine for seven months until investigations and a new highly engineered tailings facility were completed.

Shortly after the spill began, a coffer dam was constructed and the spill was diverted into Fennel Pit as shown in Figure 12-4, where the reddish saprolite soil and tailings water ("soup") is evident.

Figure 12-4 Tailings Spillage Diverted to Open Pit August 1995.

The spill event resulted in the tightening of operating conditions at Omai. One of these conditions was the requirement that mine effluent meet very strict effluent standards for cyanide of <0.005 mg/L TCN.

This extremely strict effluent requirement was never exceeded in the remaining years of operation and in-pit disposal was instrumental in achieving this objective. Even though a state-of-the-art peroxide treatment plant had been installed, peroxide treatment was never required to meet the TCN standard. It was initially determined during the 7-month shut down that removal of suspended solids by natural sedimentation or enhanced methods would allow the effluent to meet the strict effluent standards through natural evolution and degradation of the cyanide.

[31] The high cyanide in the tailings water was principally due to high TSS in the tailings water resulting from the processing of saprolitic ore zones.

[32] Saprolite is the product of extreme weathering of granitic rocks in a tropical environment. Saprolite is typically red, has low hydraulic conductivity and contains principally iron oxides and silica.

PIT DESCRIPTION

The Wenot Pit, the smaller of the two Omai pits, is 1.5 km long, 550 m wide at the widest location and 215 m deep (Figure 12-5). The larger Fennel pit is approximately 1.1 km in diameter and 260 m deep (Figure 12-6).

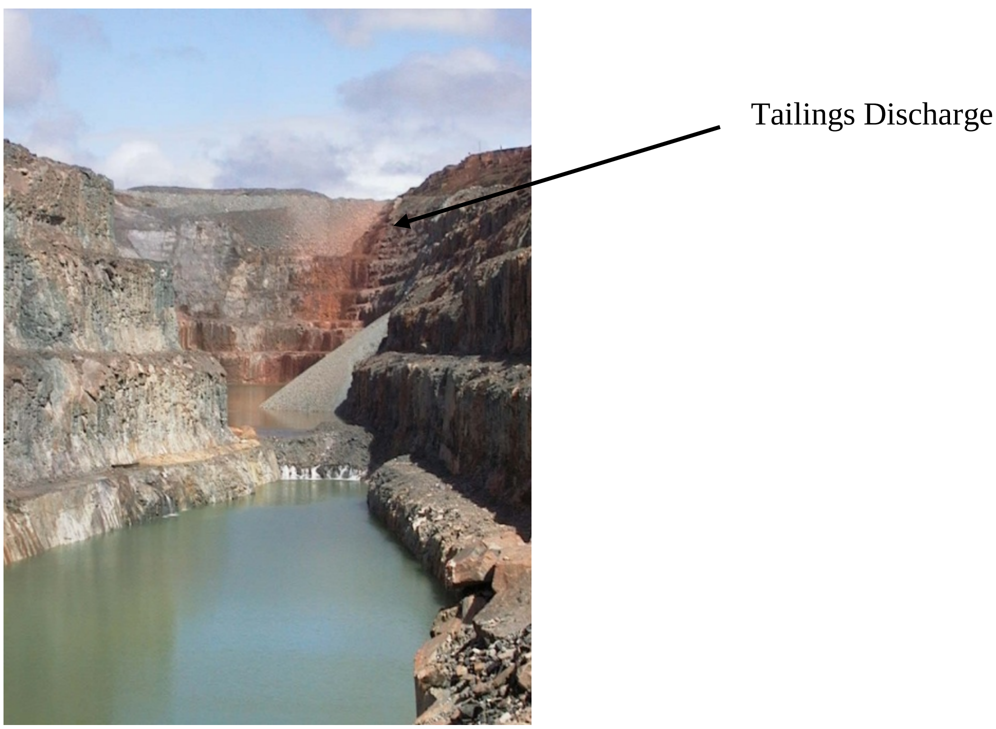
The general geology of the region includes greenstone belts similar to the Archean greenstone belts of the Canadian Shield. The Guyanese gold deposits, like Omai, are located near the margins of granitic to dioritic stocks that intrude the greenstone belts. Both of the Omai pits were relatively dry with most of the water pumped from the pits originating from rainfall.

STRATEGIC USE OF PITS

The first use of in-pit disposal at Omai was the emergency diversion of tailings water (as discussed above).

The second and permanent use of in-pit disposal was the deposition of tailings into the Wenot Pit following completion of mining in this pit in June 2002. The initiation of tailings disposal is illustrated in Figure 12-5.

Figure 12-5 Initiation of In-pit Tailings Disposal at Omai Gold Mines



Hydrological studies had determined that the pit water would be hyrologically isolated from the surrounding groundwater, and in particular the Essequibo River, provided the pit water level was maintained below a selected level (about 10 metres below overflow).

The third application of in-pit disposal at Omai was the transfer water from the Wenot in-pit tailings disposal facility while a low grade ore stockpile was being processed. This allowed Omai to maintain pit waters, which had significant concentrations of suspended solids, ammonia and cyanide complexes (cyanide and cyanate), below the predetermined level. The water transfer is shown in Figure 12-6.

Figure 12-6 Transfer of In-Pit Tailings Decant Water into the Mined-out Fennel Pit

MATERIALS PLACED IN-PIT AND METHODS OF PLACEMENT

The following table summarizes the materials and in-pit placement method at Omai.

Table 12-1 Summary of Materials and In-Pit Placement Method at Omai Wenot Pit

In-pit Disposal Activity	Description	Dates	Material	Placement Method	Follow-up
1	Emergency storage of tailing water	August 1995 to February 1996	Approx. 1 million m^3 mixture of tailings water and soil	Discharged at edge of pit	Pumped to new tailings facility in February 1996
2	In-pit disposal of tailings	July 2002-September 2005	21 million tonnes of fine gold tailings @ 50% solids[33]	Discharge at a single point at pit edge[34]	Monitoring of water elevation and quality during operation and following closure
3	Temporary storage of in-pit tailings decant water	January 2005 to	Wenot pit tailings decant water[35]	Pipe discharge half way down pit wall	Subsequently pumped out for advanced exploration and discharged to the environment CN not detected (<0.002 mg/L)

PERFORMANCE OF IN-PIT DISPOSAL

Activity 1, the diversion of a tailings water spill to a pit, met the needs of the emergency situation. Also, the elevated cyanide levels in the dilute slurry stored in the pit were essentially removed by natural degradation by the time the pit was to be pumped out seven months later.

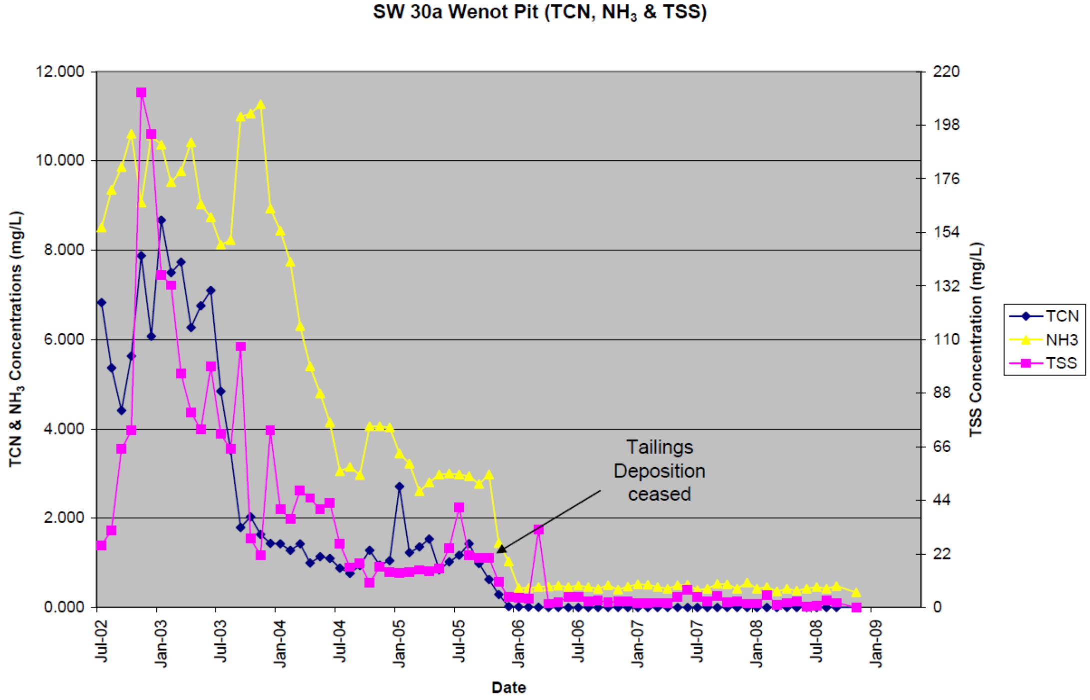
Activity 2, the in-pit disposal of tailings in the Wenot Pit, supplemented by relocation of decant waters to the Fennel Pit, met conservative long term tailings disposal objectives and provided for natural degradation of cyanide to meet very stringent discharge water quality objectives. The completed in-pit disposal is show in Figure 12-7.

[33] Omai tailings were 80% minus 75 μ, often contained clay sized saprolite material, cyanide ranged from 15 – 50 mg/l TCN and ammonia 5-15 mg/l.
[34] Spray from discharge was at times a health concern at the discharge point.
[35] Contained trace amounts of TCN (about discharge objective of 0.005 mg/l TCN.

Figure 12-7 Wenot In-Pit Tailings Disposal after Mill Shutdown

The Wenot pit performance is summarized in Figure 12-8. As shown, the pit water met discharge quality objectives for TCN (<0.005 mg/L), ammonia (<15 mg/L) and TSS (<15 mg/L) very quickly (about 4 months) after tailings deposition ended.

Figure 12-8 In-pit Tailings Disposal Lake Water Monitoring at Omai



The results of the pit lake monitoring indicated that:

- No active treatment action was required;
- The rate of cyanide degradation exceeded the cyanide release into the water column from tailings consolidation and diffusion from the tailings porewater; and
- Small amounts of ammonia continued to be generated from cyanide and cyanate degradation.

The ammonia promoted algal activity in the pit lake as indicated in Figure 12-9. This phenomenon is observed to be unusual in tropical waters that are typically nutrient poor. Biological activity included birds, amphibians, and cayman (species similar to alligator) and capybaras (the world's largest rodent) were observed drinking the pit lake water.

Figure 12-9 Nitrogen-enhanced Algal Activity in Closed Out In-Pit Tailings Facility

REMEDIAL

As commonly experienced with many pits when a pit lake forms, pit walls that are stable during mining can be destabilized as a result of increased phreatic pressures in the pit walls as water levels rise. This was the case on the south side of the Wenot facility, as shown in Figure 12-10, where hydraulically conductive sands were present. On the north side, which was composed of saprolite/laterite earth, the slopes remained stable, with vegetation self-established, as shown Figure 12-11.

Figure 12-10 Slumping South Pit Wall – Wenot Pit

Figure 12-11 Stable North Pit Wall at Omai

An external review by Golder Associates in "*Geotechnical Inspections Mine Closure Omai Gold Mine Guyana, South America*" dated July 3, 2007 concluded the following:

> "*Regular sampling of the pond water at previously established monitoring locations indicate that the Wenot Pit lake water meets direct discharge criteria"; "it is considered that the Wenot Pit lake water level will likely stabilize in the Berbice Sands above the 488 metre elevation"; "at the time of our recent inspection, the laterite/saprolite benched along the north wall of the Wenot Pit were performing satisfactorily... with relatively minor sloughing observed. The Berbice sands in the south wall have generally sloughed and/are slumped to or near a natural angle of repose - additional sloughing or slumping can be*

> *anticipated at a reduced frequency within time*"; and "*Safety berms should be constructed around the perimeter of the pit*".

The only action identified (safety berms) was completed in December and the success at the Wenot Pit was recognized by Guyana Environmental Protection Agency and Guyana Geological Commission.

BENEFITS and LESSONS OF IN-PIT DISPOSAL AT OMAI GOLD MINES

A mine pit was successfully used in a spill emergency two years after mining began. Later a mined out pit was successfully used to dispose of tailings. This significantly reduced the need for surface disposal of tailings. At end of active mining, a second pit was used as a reservoir to permit hydrological isolation of pit lake water while tailings were being deposited in the first pit. Both pits efficiently operated as self-treatment systems to enable Omai to meet the most stringent cyanide and ammonia containment and extremely strict discharge water quality objectives. Tropical conditions permitted the natural degradation of cyanide in pit waters. The use of a mined out pit to dispose tailings eliminated the need to expand surface tailings disposal facilities and provided permanent disposal which eliminates need for long term care and maintenance.

In-pit disposal of tailings at Omai followed examples elsewhere in the world where a mine facility includes more than one open pit, residual resources are not sterilized and the hydrogeological conditions provides isolation of mobile contaminants.

RESIDUAL ISSUES

No chemical issues remain at Omai related to in-pit disposal. Physical barriers (large rocks and berms) were required installed to provide public safety around the pit edge.

The Omai site has been returned to Government of Guyana [1].

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