Predicted Continental Pit Lake Water Chemistry Following Mine Closure

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1.0 INTRODUCTION

1.1 Background

The Cobre Mining Company (Cobre) Continental Mine is located in the Central Mining District of New Mexico, which occupies an area near the towns of Fierro, Hanover, Santa Rita, and Bayard in eastern Grant County. Historical mining activities at the Continental Mine have utilized both underground and open pit methods to extract magnetite and copper ore since the early 1900's. Copper production from the Continental Pit began in 1967, and by 1992 mining operations had produced over 14 million tons of ore containing 0.81 percent copper from open pit operations. In 1993, Cobre started mining the Continental Pit situated on the eastern side of Hermosa Mountain. Cobre continued to produce copper ore using both underground and surface operations until large-scale mining was temporarily suspended in the spring of 1999.

Cobre has received regulatory approval to expand mining activities at the Continental Mine, which will include excavation of Hanover Mountain, expansion of the South Waste Rock Disposal Facility, and northwest expansion of the Continental Pit. Currently, the Continental Pit is about 760 m (2,500 ft) long in the north-south direction with a width of about 550 m (1,800 ft). The bottom of the pit is at an elevation of 1914 m (6,280 ft) above mean sea level (amsl), which is about 400 m (1,300 ft) below the summit of Hermosa Mountain on the northwest portion of the pit. At closure, surface water will be diverted away from the Continental Pit, but the pit will continue to capture direct precipitation and pit wall runoff. At the end of mining, the ground water table is expected to rise when dewatering activities are terminated, resulting in the formation of a terminal pit lake. A pit lake is defined as terminal if evaporation from the lake surface maintains a steady state water level that is lower than the local ground water table and there are no ground water or surface water flows out of the pit lake.

1.2 Objectives

Condition No. 85 of Discharge permit DP-1403, approved by the New Mexico Environment Department (NMED) on December 10, 2004, states "*...Cobre shall submit to NMED for approval a work plan, including an implementation schedule, for a study to supplement the existing Pit Lake Formation Model submitted December 1999. The study shall adequately address the comment letter from NMED regarding the submittal of a third party review of hydrogeologic characterization, groundwater flow model and predicted pit lake chemistry of Cobre's Continental Pit dated January 29, 2001. The study shall address the predicted site hydrology for, at a minimum, the scenarios of partial backfilling of the Continental Pit and allowing a pit lake to form after site closure.*"

In December 2005, Cobre submitted the "*Workplan to Predict Continental Pit Lake Water Quality Following Mine Closure*". Conditional approval of the workplan was subsequently granted in a letter from Kurt Vollbrecht (NMED) to Timothy Eastep (Cobre) on July 5, 2006. The Continental Pit Lake water quality predictions are presented herein and were developed by incorporating the comments received by NMED, which are also specifically addressed in Attachment 1 of this report.

1.3 Closure Scenarios and Conceptual Models

At closure the pit may have either of two configurations, the pit as it currently exists, or an expanded pit if additional mining is performed. There are four scenarios for closure of the Continental Pit that were considered and for which conceptual models were developed. The scenarios include leaving the pit open, back-filling it partially, and back-filling it completely. In all scenarios, water is allowed to flow into the pit to a steady-state level. The four scenarios are:

1. The current, non-expanded pit option - there will be no additional mining in the pit and it will be left open in its current configuration

2. The expanded pit option - the northwest wall of the pit, to the top of Hermosa Mountain, will be mined and the subsequent pit will be left open
3. Partial backfill of non-expanded pit with evapotranspiration - the pit will be partially backfilled with waste rock to a level that will allow plants to grow using ground water
4. Complete backfill of non-expanded pit with flow-through - the pit will be backfilled with waste rock to a level that will prevent evaporation and evapotranspiration, so that ground water flows through the pit.

The first three scenarios were simulated with the pit lake chemistry model and the results presented in this report. The fourth scenario was not modeled because results of the third scenario indicated that ground water quality standards for sulfate would be exceeded during initial flushing of the backfill. Thus, similar results would have ensued from modeling the fourth scenario. The following section discusses the assumptions, processes and conditions used in modeling each scenario, as well as a conceptual model of each scenario. Table 1 summarizes the water and solute balance processes that are included in each scenario, and whether they change following submersion due to filling of the pit with water.

1.3.1 Scenarios 1 and 2, Non-expanded and Expanded Open Pit

Scenarios 1 and 2 are handled in the same way. Figure 1 is the conceptual model that shows the boundaries across which the inputs and outputs of water and solute mass are accounted, and the processes that add and subtract water and solute mass. Because the focus of the model is the concentration of solutes in the water, the boundaries follow the air-water interface, and the pit wall-water interface. Unlike most lake models, the air-water interface for a pit lake moves substantially over time as the pit fills with water, and there is no water outflow via surface or ground water.

In these scenarios, sources of water to the pit lake are ground water, runoff from precipitation on the pit walls, and precipitation on the surface of the lake. There are no surface streams into or out of the lake. The only loss of water from the pit is by evaporation directly from the lake surface. Precipitation and evaporation rates are modeled as constant throughout the year in order to simplify the model. This simplification is made because high-frequency or short-period cycles such as precipitation do not generally affect long-term (e.g., 300-year) predictions. The only differences between the Scenarios 1 and 2 are in the physical dimensions of the pits, the ground water inflow rate, and the proportions of wall rock exposed at different elevations.

In Scenarios 1 and 2, sources of solutes to the pit come from the water sources identified in the water balance, and the weathering of geological materials. The geological materials are categorized as intact wall rock, rubble, and floor rock (i.e., essentially wall rock but on nominally horizontal surfaces). The term weathering refers to all geochemical processes including dissolution, oxidation and redox controlled reactions of all rocks and minerals, not just oxidation of pyrite. Solutes from weathering of wall rock above and below the water level are handled differently. Solutes from geological materials above the steady-state water level are added to the pit via pit wall runoff, represented by the measured chemistry of a seep (seep HSN-01, which is located north of Hanover Mountain but emanates from the a mineralized area of the Colorado Formation) on the wall and test results from meteoric water mobility testing of composite sample CF 103 (SMI, 1999). The amount of solute from weathering of wall rock, floor rock and rubble, is calculated based on the amount of each type of wall rock, their duration of exposure for weathering prior to inundation, and results of humidity cell tests on wall rock materials. These soluble weathering products are allowed to accumulate on the wall rock until they are washed into the lake by ground water as the water level rises. Because there is no outflow of water from the pit, the only loss of solutes from the water is precipitation of minerals.

1.3.2 Scenario 3, Partial Backfill with Vegetated Surface

In this scenario, the non-expanded pit is partially backfilled and the surface supports vegetation which maintains the water balance by replacing the process of evaporation with evapotranspiration. Figure 2 is the conceptual model for the partial backfill scenario in which vegetation provides evapotranspiration. The backfill material is assumed to be leach cap rock of Hanover Mountain (as it is the most logical candidate to be considered for backfill material), which was evaluated with a humidity cell test (WRC-09). It is assumed that the backfilled material will have a porosity of 25 percent.

The components of the water balance for water in the backfilled pit are slightly different from the open pit scenarios. In Scenarios 1 and 2 the lake surface is assumed to be 100 percent effective at capturing precipitation, whereas in the backfill scenario, only 15 percent [based on baseline characterization work (SMI,1999)] is allowed to infiltrate while the remainder is lost by evapotranspiration. Further, because there is no open water surface, direct evaporation is not included. Evapotranspiration by plants is included on a seasonal basis when the water table rises to the root zone, which is assumed to be 1.5 meters (5 feet) deep.

Sources of solutes are the same as for the open pit scenarios except that weathering of the backfill material is added, and rubble is ignored because it is inherently included in the backfill. Similar to weathering of wall rock, weathering products of the backfill are allowed to accumulate until the water level rises and dissolves the weathering products. Unlike the open pit, which has oxygenated water, pore water in the backfill is assumed to be anoxic. This assumption is made because ground water from deep aquifers generally has very low dissolved oxygen concentrations, and pore water will not be oxygenated by the air in the unsaturated zone of the backfill because the air will be depleted of oxygen by reaction with sulfides in the backfill. Therefore, unlike the open pit, weathering of the wall rock, as well as the backfill, will stop once it is inundated. While the water level rises from ground water inflow, infiltration of precipitation will carry weathering solutes downward toward the rising water surface. The result is that these solutes are delivered to the rising water surface. To simplify the model it is assumed that infiltration does not transport solutes downward and that the rising water table dissolves all the weathering solutes.

1.3.3 Scenario 4, Complete Backfill with Ground Water Flow-through

In this scenario, the non-expanded pit is backfilled to an elevation that is above the steady-state water table after it rebounds. Figure 3 is the conceptual model for this scenario. The surface might have vegetation that is supported by precipitation, but unlike Scenario 3 the vegetation does not rely on water from the water table. The backfill material is assumed to be taken from the leach cap rock of Hanover Mountain, which was evaluated with a humidity cell test (WRC-09).

The components of the water balance for water in the completely backfilled pit are slightly different from Scenario 3. It is assumed that the water table is far enough below the surface of the backfill that plant roots cannot reach it and so there in neither evaporation nor evapotranspiration. Evapotranspiration by plants is assumed to prevent infiltration of precipitation that lands on the backfill. Sources of solutes are the same as for Scenario 3.

2.0 MODEL DEVELOPMENT

2.1 The Modeling Procedure

Simulating the chemistry of the water for the closure scenarios requires the use of two types of models, a mass balance model and a chemical equilibrium model. These two components of the water chemistry model are used in succession. First, the mass balance model developed for each scenario was used to calculate the mass of water and solute in the pit at any given time. The modeling framework used for the mass balance calculations was a dynamic systems model (DSM). The DSM outputs the volume of water in the pit and the total amount of solutes (e.g., sulfate, iron, calcium) that entered the pit up to a given time in the future. The volume of water and mass of solutes yields the unequilibrated concentrations (that is not charge balanced and has not been subject to geochemical controls) of solutes. In the second step, the unequilibrated solute concentrations from the DSM are used in a thermodynamic chemical equilibrium model that takes into account the influences of the atmosphere, precipitation/dissolution, and sorption to determine the resulting chemistry of the water. Unlike the mass balance component in which a separate model was developed for each scenario, the geochemical equilibrium component used a single model, PHREEQC (Parkhurst and Appelo, 1999), for all calculations. The functioning of both the DSM and chemical equilibrium model is discussed in more detail in Section 4.0.

The DSM models developed for the Continental pit closure scenarios are mechanistic mass balance models that account for the inputs and outputs of both water and solutes. The mathematical representations used in the mass balance models are based on the conceptual models developed for each scenario, which identify those processes and conditions known to affect the variable of interest. As a mass balance model, the DSM tracks the masses associated with each input and output processes throughout the simulation period, which helps identify the processes that control the system.

2.2 The Physical System

The conceptual models present a qualitative picture of the physical system being modeled. In the mass balance model, values for several physical parameters are needed at each time step in order to calculate the gains and losses across physical boundaries. These gains and losses include, for example, evaporative losses, weathering of wall rock, and water elevation. These physical parameters of the pit were developed from topographic maps of the existing and expanded pit shown in Figures 4 and 5, respectively. These parameters are the water volume as a function of elevation, the length of the shoreline (perimeter) as a function of elevation, and the surface area of the air-water interface as a function of elevation. In addition to the pit, the underground workings are filled with ground water simultaneously, and the volume of the underground workings is included. Figures 6, 7 and 8 show the volumes, shoreline length, and lake surface area for the non-expanded pit. Figures 9, 10 and 11 show the volumes, shoreline length and surface area for the expanded pit.

2.2.1 Stratification and Mixing in the Continental Pit Lake

In addition to the boundaries that exist at visible interfaces (i.e., rock/water, and water/air), intermittent and sometimes permanent water/water boundaries can exist within the water column of a lake due to stratification. An important aspect of the Continental Pit Lake will be how water in the pit stratifies and mixes. Under certain conditions, stratification can affect the distribution of solutes, the oxidation-reduction potential, and the solubility of minerals. There are two types of density stratification that can occur in pit lakes, thermal stratification that occurs on a seasonal basis in nearly all lakes, and TDS/salinity stratification that persists continuously for years in only a small percentage of lakes. Intermittent or permanent stratification is important because it regulates the delivery of dissolved oxygen (DO) from surface waters that are in contact with the atmosphere, to the deeper isolated waters. The presence of DO controls the oxidation-reduction potential of the water, which affects not only the weathering of pyrite, but also the speciation of redox-sensitive elements. In order to appropriately specify the oxidation-reduction potential for thermodynamic chemical calculations and to determine whether submerged pyrite will continue to oxidize, the stratification pattern, as well as oxygen sources and sinks must be determined. These internal physical processes were evaluated to determine whether the model should include them as transient boundaries and processes within the primary system boundaries, depending on whether they lead to different chemical processes in different parts of the lake. This evaluation is presented in Attachment 2. In brief, the evaluation concludes that the pit lake will thermally stratify in the usual dimictic pattern, and will have low concentrations of phosphorus, which will limit algal productivity and will maintain a well-oxygenated water column throughout the year. The effects of temperature and carbon dioxide equilibria are discussed in Section 4.2.

3.0 PROCESS REPRESENTAION AND INPUT VALUES

3.1 Hydrologic Balance

The water mass balance components for the open pit lake include: (1) direct precipitation to the lake surface, (2) runoff from precipitation falling on the pit walls, (3) ground water inflow, and (4) evaporation from the lake surface. Upland runoff will be diverted around the pit and is, therefore, not incorporated into the water balance. The resulting conceptual hydrologic model is shown in Figure 1. For the partial backfill scenarios, there are two changes to the water mass balance, which is shown in Figure 2. The direct precipitation to the lake surface is replaced with direct precipitation on the backfill and partial infiltration, and evaporation from the lake surface is replaced with evapotranspiration only when the water table reaches the root zone.

Currently, the Continental Pit is dry, however pumping of ground water stopped in 1999 and underground mine workings underneath and adjacent to the pit are filling with ground water. Changes in the water depth in the pit lake as the ground water table rebounds are determined using the dimensions of the pit combined with the water balance. For a given time interval during filling, the volume of water in the pit lake is determined from the balance of water flowing into the pit from precipitation, ground water, and runoff, and water leaving through evaporation. It is assumed that the water level in the underground workings is the same as the pit. This calculation is repeated over time until inputs and outputs balance and a steady-state water level has been achieved. Results of this analysis indicate that at steady-state the evaporative loss is great enough that the water level in the pit never rebounds all the way to the level of the pre-mining water table. The result is that ground water continues to flow into the pit indefinitely and it becomes a terminal lake (i.e., there is no outflow of ground or surface water). The steady-state elevations calculated from the water balance were input into the existing ground water model to verify that the pit lake is indeed terminal (Telesto, 2008).

3.1.1 Ground Water

The pit lake ground water inflow component was determined using a numerical ground water model that was developed for the Continental Mine site (Telesto, 2008). The numerical model was calibrated to a steady state condition corresponding to a period characterized by relatively constant hydraulic stresses associated with operational activities. Additional details regarding ground water model calibration and predictive simulations are provided in the supplemental ground water study for the Continental Mine site (Telesto, 2008).

For each scenario (i.e., non-expanded pit, expanded pit, and backfilled pit), estimates of ground water flow rates (as a function of elevation) from the ground water model were input as a lookup table into the pit lake chemistry model. Figure 12 shows the estimates of net ground water inflow rates for each scenario as predicted by the numerical ground water model.

3.1.2 Climate Data

The Continental pit is located in a semi-arid region of southwest New Mexico. The regional climate is described by data obtained from the Fort Bayard weather station and Chino Mine located six miles southwest and four miles south of the site, respectively. The following statistics have been developed from these databases:

- Mean annual precipitation of 40 cm/yr (15.7 in/yr) (Fort Bayard)
- Mean annual temperature of 12.8° C (55.1° F) (Fort Bayard)
- Mean minimum temperature of -3.7° C (25.3° F) (January, Fort Bayard)
- Mean maximum temperature of 30.4° C (86.8° F) (June, Fort Bayard)
- Mean annual pan evaporation rate of 202 cm/yr (79.7 in/yr) (Chino Mine)

Site-specific weather data are limited and therefore mean conditions and future trends were predicted using the regional climate data and local elevation as inputs in climatic models. To estimate the mean climatic conditions at the Continental Mine, the PRISM model (Daly and others, 1994) was used. Table 2 summarizes the mean monthly values for the resulting data set.

3.1.3 Precipitation

Based on the PRISM model for extrapolating climatic data, the mean annual precipitation at the site is estimated to range from 36 cm/year (14 in/yr) at lower elevations (Cron Ranch) to 61 cm/yr (24 in/yr) at higher elevations (north of Hanover Mountain). The mean annual precipitation used during development of the hydrologic model was 46.5 cm/year (18.3 in/yr). The monthly distribution of precipitation as a percentage of the mean annual precipitation is assumed the same as the distribution measured at Fort Bayard, which shows a distinct wet season during the months of July through September. These data were utilized previously for the baseline evaluations (SMI, 1999). Because long-term precipitation data is unavailable for the Continental Mine, the stochastic weather generator CLIGEN (USDA, 2000) was used to create a synthetic 200-year daily precipitation record. This data set was created by first developing a synthetic 200-year data set for Ft. Bayard, New Mexico, and then scaling the data for the Continental Mine so that the mean annual precipitation was equal to the 46.5 cm (18.3 in) estimated using PRISM. Direct precipitation is that precipitation falling on the pit lake surface. As the pit fills with water, the lake surface area increases and a larger portion of the total inflow will be derived from precipitation that falls directly on the lake surface.

3.1.4 Evaporation

The pan evaporation at the Continental pit was assumed to equal the 202 cm/yr (79.7 in/yr) measured at the Chino Mine weather station. Evaporation from the lake surface, 142 cm/yr (55.8 in/yr) was estimated using a factor of 0.7 times the pan evaporation (Kohler et al., 1955). Although pan evaporation varies seasonally, in order to simplify the model, it was assumed that the evaporation from the lake surface is uniform throughout the year. Because the climate at the Continental Mine is semi-arid, the pan evaporation is greater than precipitation throughout the year, even during the cooler winter months.

3.1.5 Evapotranspiration (ET)

It is assumed that salt tolerant phreatophytes will be planted at the surface of the backfill. For modeling purposes an unsaturated thickness of 1.5 meters (5 feet) above the nominal water table was used. This allows the water table to fluctuate with the seasonal evapotranspiration. The actual thickness of the unsaturated zone would need to be determined by the type of the plant that will be planted at the surface of the backfill.

Optimized evapotranspiration cells yield ET rates that are about 80 percent (Brutsaert, 1982) of the pan evaporation rate (202 cm/yr, 79 in/yr), so an ET rate of 160 cm/yr (64 in/yr) is used in the model. Studies in the southwestern US show a strong seasonal component of ET, with maximum values extending from March through September, and minimum values (about 10% of maximum) for the remainder of the year. Figure 13 shows the assumed seasonal pattern of ET for the site.

3.2 Solute Sources

Solutes sources to the pit include those associated with each of the water sources, as well as weathering of the exposed geological materials, such as wall rock. The sources and procedures for estimating the quantities of solutes that enter the pit are described in the following sections.

3.2.1 Hydrologic Solute Sources

Ground water

As ground water flows into the pit, it carries with it solutes that originated in the regional aquifer and will accumulate in the pit lake water. Water chemistry data have been collected from monitoring wells near the pit (MW-5, MW-5a, MW-7, MW-8, and MW-21).

Monitoring well MW-5 is screened at about 70 to 76 m (230 to 250 ft) below ground surface (approximate ground surface elevation of 2,085 m or 6,842 ft) in the Lake Valley limestone, and although it is located cross-gradient to the pit, it represents the chemistry of water in contact with that formation.

MW-5a is about 800 m (2,625 ft) to the southwest of the pit and is screened in the Colorado Formation and is in very close proximity to the Barringer Fault zone. Water chemistry from this well suggests a very close correlation to MWMP data taken from the Barringer Fault. Ground water in this area flows toward the southeast direction. This well is located cross gradient from the pit and is considered to be too far away to influence pit inflows

MW-7 is a shallow well located about 900 m (2,953 ft) to the north of the pit and is screened at about 16 to 22 m (50-70 ft) below ground surface (approximate ground surface elevation of 2,134 m or 7,002 ft) in a Syenodiorite porphyry. MW-7 is believed to be influenced by infiltration from the Main Tailings Impoundment (MTI) Reclaim Pond. Ground water flow through this location is toward the southeast direction.

MW-8 is located about 800 m (2,625 ft) to the northeast of the pit and is screened in the Colorado Formation. MW-8 is also a shallow well and is screened at about 20 m (ground surface elevation of approximately 2,089 m or 6,854 ft) in the Grape Gulch drainage.

Because of the distance and location in a different drainage, ground water in this area should have no influence on pit inflows

MW-21 is located about 400 m (1,312 ft) southwest of the pit and is screened at about18 to 27 m (ground surface elevation of approximately 2,106m or 6,910 ft) in the Lake Valley Limestone. The well is located side gradient from the pit and ground water moves in a southeast direction.

MW-5 is considered to be most representative of the ground water chemistry that is expected to discharge into the pit from the west-northwest direction. In addition, MW-5 is screened in a limestone formation which makes up a significant portion of the current exposed pit bottom. The chemistry of MW-5 has been monitored on a quarterly basis since 2000 and is used to represent the majority of ground water flowing into the pit. The average chemistry (based on data collected between April 2000 and April 2007) of MW-5, which is used in the model, is summarized in Table 3.

Because the ground water originates from the deep aquifer and has been in contact with the regional geologic materials for a long time, it is assumed that the ground water is in chemical equilibrium with wall rock. It is also assumed that because ground water is from the deep aquifer, the dissolved oxygen concentration is essentially zero and that it does not cause additional weathering of the wall rock. However, as the water level in the pit rises and inundates wall rock, the inflowing ground water also dissolves the products of wall rock weathering, and introduces those solutes to the lake. This latter source is discussed in Section 3.2.2.

Precipitation

Solutes contained in precipitation will also contribute to the loading of chemical mass to the water in the Continental pit. The chemical composition of precipitation for the site is assumed to be equal to the average precipitation chemistry from 1985 to 2006 (Table 4) from the National Atmospheric Deposition Program site located at Gila Cliffs National Monument, New Mexico (Station NM01).

Runoff and Seeps

Precipitation that falls on the wall rock and dissolves weathering products also introduces solutes to the pit lake. These soluble weathering products are handled in two ways in the model to avoid double counting their masses. The mass of solutes introduced by runoff and seeps is calculated for only the area of pit wall that is above the steady-state water level. This runoff source of solutes is continued at a fixed rate through the whole duration of the simulation. For the solutes produced at elevations below the steady state water level, these are linked to the duration of weathering exposure, and are introduced to the pit when the rising water level inundates the rock and dissolves them. This latter source is discussed in Section 3.2.2.

For runoff and seeps, it is assumed that 15 percent of the precipitation that falls on the wall rock infiltrates into the pit wall, travels as interflow and discharges as seeps on the pit wall. The chemical composition of the runoff and seeps depends on the formation through which the water travels. A seep (seep HSN-01) from the Colorado Formation has been sampled and analyzed and the chemical composition is summarized in Table 5. Meteoric water mobility test data (shown in Table 5) for composite sample CF103 (SMI, 1999) has been used as a surrogate to represent runoff chemistry from Paleozoic carbonates.

The average chemical composition of the runoff/seeps is assumed to be proportional to the percent of the wall rock area comprised by each formation. Because seeps do not exist in all the formations exposed on the pit wall, these formations are grouped with either the Colorado Formation or the Paleozoic carbonates. The Colorado Formation is chemically distinct from the carbonates, and only the Beartooth Formation is grouped with the Colorado Formation. All other formations are grouped with the Paleozoic carbonates. For the non-expanded pit approximately 23 percent of the wall rock area above an elevation of 2,009.4 m (6591 ft) is treated as Colorado Formation and 77 percent as a Paleozoic carbonate. For the backfill scenario approximately 20 percent of the wall rock area above an elevation of 1998.3 m (6524 ft) is treated as Colorado Formation and 80 percent as a Paleozoic carbonate. For the expanded pit approximately 27 percent of the wall rock area is treated as Colorado Formation and 73 percent as a Paleozoic carbonate. The loading rate of solutes by runoff and seeps is calculated as the product of the infiltration, the pit wall area, and the area-weighted average solute concentrations.

3.2.2 Geologic Material Solute Sources

The rate at which solutes are released from geologic materials to the pit depends on the minerals present and the site-specific weathering environment, and there are both theoretical and empirical approaches to quantifying weathering. The Condition 85 work plan (Telesto, 2005) proposed that the Davis-Ritchie model (Davis and Ritchie, 1986) would be employed for estimating weathering and solute release from the wall rock. However, there are serious shortfalls in applying the Davis-Ritchie model to wall rock, and this approach has been dropped. The rationale for the change in approach is detailed in Attachment 3.

Figure 14 shows a stratigraphic section of the rock formations in the area of the Continental pit. There are ten formations exposed on the pit wall (Lake Valley limestone/marble, Percha shale, Marble alteration, Oswaldo limestone, Fusselman & Montoya, Hanover-Fierro, Prestock dikes, El Paso, Syrena limestone) that will weather and release solutes to the pit. Figure 15 shows a plan view of the wall rock types for the existing pit. Figure 16 shows a plan view of the wall rock types for the expanded pit. The expansion plan for the Continental pit will result in the exposure of a greater percentage of the Colorado Formation in the ultimate pit wall surface than in the existing pit, although the majority of the pit wall will still consist of calcareous rocks of the Syrena, Oswaldo, and Lake Valley Formations.

The exposed area and elevation of these rock types may influence the progression of pit water chemistry depending on their chemical composition, weathering rates, and duration of exposure prior to inundation. For example, rocks at higher elevations will have more time to weather before being inundated, and will release more solutes than if they were at a lower elevation. Using AUTOCAD, the amount of each rock type that is exposed on the pit wall was quantified at three-meter (ten-foot) elevation intervals for the existing and expanded pits. The results of this analysis is shown in Figure 17 for the non-expanded pit and in Figure 18 for the expanded pit. The amount of each rock type is quantified as the length along the perimeter (i.e., the shoreline) as a function of elevation.

Geologic materials are categorized in the model as intact wall rock, rubble and backfill, according to the physical characteristics of the material and the scenario being modeled. Intact wall rock is material that has not moved during or since the pit was created. Rubble is material that has been dislodged from the wall and settled on relatively level surfaces, and tends to be broken into smaller pieces. The mineralogy and weathering characteristics of rubble are assumed to be the same as adjacent wall rock. Backfill material is assumed to have the composition of the Hanover Mountain oxidized leach cap.

Geochemical Testing and Interpretation

Various testing procedures have been developed to characterize the chemistry of water that contacts geologic materials under different environmental conditions (e.g., ABA, MWMP, HCT, SPLP), however, there are currently no standard tests designed to evaluate the long-term release of solutes from intact wall rock as a pit lake forms. Two types of tests were performed on geologic material from the Continental pit, acid-base accounting (ABA) and humidity cell tests (HCT) (SMI, 1999). The results of these tests reflect two important components of the rock (sulfide and limestone) that influence the water chemistry through oxidation of sulfide minerals, pH buffering with carbonates, precipitation of secondary minerals, and dissolution.

ABA was performed on material from all formations exposed in the Continental pit (Abo-Syrena [n=4], Beartooth [n=3], Colorado [n=8], Continental breccia [n=1], El Paso [n=11], Fusselman & Montoya [n=4], Hanover-Fierro [n=3], Lake Valley [n=4], Oswaldo [n=2], and Percha [n=2]). The results are summarized in Table 6. Geochemical testing of these rocks indicate that the majority of the exposed wall rock formations have positive net neutralization potentials (NNP) and therefore are expected to produce neutral to alkaline solutions upon weathering. Only rocks from the Beartooth and Colorado Formations have slightly negative NNP values. As shown in Figure 15, the Beartooth Formation is a narrow strip exposed at an elevation between about 2040 and 2070 meters (6,700 and 6,800 feet) along the west side of the pit, while the Colorado Formation lies above about 2073 meters (6,800 feet). Both formations are above the predicted steady-state water level of the pit lake (for both the non-expanded and expanded scenarios). Table 6 includes a "Marble Alteration" formation that comprises a significant portion of the wall rock, but for which no ABA data are available. It is assumed that it has the same percent sulfide and NNP as the average of the two marble samples analyzed from the Lake Valley Formation and the one marble sample analyzed from the Fusselman & Montoya Formation.

HCT were performed on six samples of wall rock from the Abo & Syrena (WRC-02), Colorado (WRC-01), Fusselman & Montoya (WRC-04), Hanover-Fierro (WRC-06), and Lake Valley (WRC-03 high sulfide, WRC-05 low sulfide) Formations. In addition, an HCT was performed on the oxidized leach cap from the Colorado Formation at Hanover Mountain (WRC-09). Splits of these samples were also used in the ABA. Table 7 summarizes several chemical parameters that relate to the acid generation potential and the pH of water that contacts these samples. HCT were not performed on Beartooth, Continental Breccia, El Paso limestone, Marble alteration, Oswaldo and Percha Formations. Figures 17 and 18 show that the El Paso Formation comprises a very small portion of the wall rock, whereas the Marble alteration, and Oswaldo and Percha limestones comprise a large portion of the wall rock that will be submerged. Therefore the untested formations were grouped with one of the rock types for which HCT were performed based on similarities in rock type, sulfide concentration, NNP and whether they occur at nearly the same elevation. The final groupings are summarized in Table 8. All rocks within a group are assumed to have the same weathering characteristics as the rock that was tested in the HCT. Based on these HCT groupings, the amount of each group exposed as a function of elevation was determined for the non-expanded (Figure 19) and expanded pit configurations (Figure 20).

3.2.3 Derivation of Weathering Rates

In this section, the weathering and solute release rates for geologic materials are determined from the laboratory HCT. Disparities between laboratory conditions and field conditions are identified (e.g., particle sizes and wetting/drying cycles) and procedures for converting laboratory values to field values are developed.

HCT data are commonly evaluated on a basis of mass of solute per mass of rock per time. Interpreting the weathering rate based on the mass of rock fails to account for the fundamental role that surface area plays in chemical weathering. Numerous studies of sulfide oxidation, as well as rock weathering, have shown that reactions occur at exposed surfaces (e.g., Holdren and Berner, 1979) and derive rates based on the surface area of the material (e.g., Nicholson and Scharer, 1994; Davis and Ritchie, 1986; Rimstidt et. al, 1994). Interpreting the results on a per-unit-area basis is particularly important when results from laboratory tests that use crushed rocks are applied to more monolithic pit wall rock. Therefore, the HCT data are evaluated in the more geochemically relevant units of milligrams solute per square meter of rock (mg/m^2).

The McClelland modification to the standard HCT test procedure was used for conducting the HCT tests (SMI, 1999). The McClelland procedure differs from the standard procedure (ASTM D5744-07) in that the material is crushed to pass a 2 millimeter (mm) sieve instead of a 6.3 mm sieve. Crushing wall rock to < 2 mm increases the external surface area, increases the internal surface area through cracks, and exposes fresh surfaces for weathering, all of which can increase the measured weathering rate. To account for at least the increased external surface area, the specific surface area (i.e., m^2 of surface/m^3 of volume) of rock particles used in the HCT was estimated assuming that the average shape and size of particles was a cube 1.5 mm on each side. This assumed shape is intended to account for the angularity of freshly crushed rock, and the wide range of particle sizes that pass through a 2 mm sieve. This gives a specific surface area of 4,000 m^2/m^3 (1,200 ft^2/ft^3). The minimum specific surface area would be for a 2 mm sphere, which is 1,500 m^2/m^3 (460 ft^2/ft^3). Given the mass of rock used in the HCT (1.20 kg, 2.6 lb), and an assumed density of 2,500 kg/m^3 (156 lb/ft^3) which is between a medium density limestone (2,160 to 2,560 kg/m^3 and marble 2,590 to 2,780 kg/m^3), the total surface area exposed for oxidation in the tests is calculated as 1.92 m^2 (20.7 ft^2).

Because sulfide oxidation produces sulfate and acidity, and drives other weathering reactions, sulfate was selected as the primary solute to track. Figure 21 shows the amount of sulfate released (mg sulfate/m^2 rock) each week during HCT tests for the five materials collected from the pit walls (WRC-01 [Colorado], WRC-02 [Abo/Syrena/Percha/Oswaldo limestone], WRC-03 [Lake Valley limestone], WRC-05 [Lake Valley limestone and marble], WRC-06 [Hanover-Fierro stock], while Figure 22 shows the amount of sulfate released for WRC-04 [Fusselman/Montoya/El Paso limestone and marble], and the Hanover Mountain leach cap (WRC-09). These plots show that the mass of sulfate released decreases with time (or with each wetting and drying cycle) during the HCT. The initially high release rate is due to a combination of dissolving old oxidation products that existed before the tests, rapid oxidation of fresh surfaces created by crushing, and inadequate time for insoluble reaction products to accumulate and foul the reacting surfaces. Had the HCT continued beyond 20 weeks, a continued decline in the release rate would be expected due to decreased availability of pyrite for oxidation.

It is assumed that all sulfate released during the HCT came from oxidation of sulfide during the test. This assumption is protective because any sulfate that is present before the tests started would be interpreted as active weathering. An assumption is called "protective" in this report if it leads to over-estimating the mass or concentration of a solute released to the environment. Active weathering is used to refer to weathering solutes produced during humidity cell testing rather than those produced before the testing. This over-estimate in weathering rate is evident in data from WRC-09 which released more sulfate in the first four weeks (430 mg) than could have been produced by oxidizing all the sulfide that was initially present in the sample (120 mg) into sulfate (360 mg).

The production and release of solutes from the wall rock that ultimately becomes submerged is viewed to occur in two stages. The first stage is the continuous weathering of the exposed wall rock that started in 1999 when the current configuration of the pit was finalized. During the weathering stage it is assumed that the soluble weathering products accumulate in the wall rock. This weathering stage continues until the rock is submerged by rising water levels. The second stage is release of solutes to the water. At the moment wall rock at a given elevation is submerged, all of the weathering solutes that have accumulated since 1999 are instantly dissolved and mixed into the pit water. Therefore, the approach used to quantify the mass of solute that has accumulated and can be released at any time, is to replot the HCT data as cumulative mass of sulfate released per area of rock (mg/m^2). This is shown in Figure 23. Because samples in the HCT were weathered for 20 weeks, and the wall rock will weather for decades, the plotted release rates from the HCT must be extrapolated into the future. It was assumed that the release rate observed during the last five weeks of the HCT continued indefinitely into the future. This is a protective assumption because the release rate will continue to decrease as the availability of sulfide decreases. Equations were fit to the last five weeks of HCT data as shown in Figure 24. Linear equations are used instead of polynomial or other forms because the trend during the latter period of the HCT is more accurately fit, and the linear function is more protective than other more probable forms (e.g., log functions). The slope of the equations represents the long-term areal mass weathering rate ($mg/m^2/year$), while the y-intercept accounts for the initially fast weathering and release.

3.2.4 Disparities Between Laboratory and Field Conditions

The methodology of the HCT has significant disparities with site conditions, so scaling factors are introduced to convert "accelerated weathering" results from laboratory HCTs to natural weathering rates in the field. The two most important disparities between HCT and field conditions relate to the particle size and the wetting/drying cycle. In addition, a correction factor is introduced to account for the difference between the composition of the test sample and the average composition of the formation. The method for converting the HCT data that takes into account the differences between laboratory and field conditions is described here. First, the approach for scaling the release rates for differences in wetting/drying cycles is described. Second, application of HCT release rates to intact wall rock is described. Finally, an approach is developed to estimate the amount of wall rock that is producing and releasing solutes.

Four of the HCT (WRC-01, WRC-02, WRC-05, and WRC-06) were performed on samples of the rock formation that had concentrations of sulfide that were lower than the average for that formation, while one was near the average (WRC-04), and one was higher (WRC-03). To determine whether this selection biased the results toward lower sulfate release rates, the mass of sulfate released was plotted against initial mass of sulfide (Figure 25). The figure shows a negligible correlation between initial sulfide mass and released sulfate mass, suggesting that this selection of samples did not bias the results to lower release rates. Regardless, the sulfate release rates were scaled to account for the difference in sulfide concentration between the samples used in the HCT and average concentration for that formation. This was done by multiplying the release rates by the average sulfide concentration divided by the HCT sulfide concentration. The values of these factors are 4 (WRC-01), 6.3 (WRC-02), 0.5 (WRC-03), 1.3 (WRC-04), 4 (WRC-05), 8 (WRC-06), and 1 (WRC-09).

3.2.5 Wetting/Drying Cycle

Because the weathering of pyrite requires water and oxygen, the cumulative solute release (as a function of time) derived from the HCT, depends on the number of wetting and drying cycles. The approach used here assumes that the weathering cycles created by wetting and drying in the HCT and the wetting and drying cycle created by precipitation in the field, are analogous. In the HCT the period of the wetting and drying cycle is one week. To account for the effect of the number of weathering cycles, the frequency of cycles in the HCT was scaled to the frequency of wetting and drying cycles in the field.

To adjust the HCT average rates to field rates, the precipitation record at the main tailings impoundment weather station from November 4, 2005 to July 14, 2007 was used to determine the wetting and drying cycle for the exposed wall rock. During this 605-day period of record (1.66 years), there were 129 events with precipitation being greater than or equal to 0.4 mm (0.01 inches). Figure 26 is a histogram of the frequency and intensity of precipitation events. All precipitation events of 0.4 mm (0.01 inches) or more were considered a wetting cycle that would drive weathering similar to the HCT cycle. At the weather station there was an average of 77 cycles per year (1.5/week), while the frequency in the HCT is 52/year (1.0/week). So the scaling factor for the frequency of wetting/drying cycles is 77/52 (1.5), and the solute release rates (the slope of the equations) from the HCTs are multiplied by 1.5. The scaling factor affects only the rate of release, so it is only multiplied times the slope of the equation, not to the intercept. Table 9 summarizes the solute release equations scaled to sulfide concentrations and the weathering cycle.

In order to estimate the release rates of solutes other than sulfate, the original intent was to use sulfate as the pilot solute, and to relate the release rates of other solutes to the release rate of sulfate in the HCT. This approach was based on the association of many metals with sulfides and their release upon sulfide oxidation, and the association of other elements with carbonates that would be released by dissolution of carbonates due to the acidity produced by sulfide oxidation. At weeks 5, 10, 15 and 20, the HCT leachates were analyzed for a more complete list of solutes (i.e., Ag, Al, Ba, Be, Cr, Cd, Co, Cu, Fe, Mn, Mo, Hg, Ni, P, Se, Tl, and Zn). No significant positive correlation was found between the concentration of sulfate and the concentration of metals in any of the HCT tests. Further, the metal concentrations showed no consistent temporal trend through the HCT such as was observed for sulfate. Therefore, the release rates of metals were assumed to be constant and equal to their average concentration in the leachate. For the more common matrix solutes (e.g., Na, K, Mg, Cl and HCO_3) an evaluation similar to the metals was performed, which also showed no consistent, predictable relationship. Table 10 summarizes the average concentrations for metals and matrix solutes for each of the HCT, along with their areal release rates.

3.2.6 Particle Size and Specific Surface Area

Although there is a significant difference in particle size between the HCT and intact wall rock, rubble and backfill, no scaling factor is needed. Rather, because an areal release rate is used, the amount of solute released from the rock is calculated by simply multiplying the HCT areal release rate by the specific surface area of the wall rock that is available for weathering. The model recognizes two types of exposed surfaces on the rocks; one is the visible external surface of the wall rock, and the other is the non-visible internal surface of the rock that is created by voids and cracks.

The external visible surface area of the wall rock is estimated assuming that, except for flat areas like haul roads, the walls have a slope of 35 degrees. The model performs calculations based on changes in elevation, so for each elevation change (ΔH) along the shoreline length (L), the external surface area of the wall rock is:

$$ExternalArea = \frac{L \cdot \Delta H}{\sin 35^{\circ}} = 1.74 \cdot L \cdot \Delta H$$

The externally visible area of wall rock is only a small percentage of the surface area of the rock that is available for weathering; the majority is in the form of internal cracks.

The surface area of these internal cracks can be quantified with the specific surface area, which is the surface area of cracks per unit volume of rock. The specific surface of wall rock was estimated based on porosity and geometry of pores. In wall rock, the porosity occurs primarily as cracks or fractures that are relatively narrow and locally planar. The porosity of bedrock aquifer materials adjacent to the wall rock used in the ground water model is 1 percent. However, the process of blasting to create the pit increases the porosity. Siskind and Fumanti (1974) found that the porosity adjacent to a blast face, like a pit wall, increased by 0.2 to 5.8 percent. The porosity increase stopped at a distance of about 1.1 m (3.8 ft) from the blast face, and averaged 3 percent. Assuming the porosity (φ) is 0.03 v/v, and planar cracks that are 1 mm (0.001 m, 0.0033 ft) wide, the specific surface area of wall rock can be calculated as:

$$SA_{wallrock} = \frac{2 \times \varphi}{0.001} = \frac{60m^2}{m^3}$$

The factor of 2 occurs because each crack has two exposed surfaces. All types of wall rock are assigned the same specific surface area. This specific surface area is equivalent to infinite, planar, parallel cracks spaced at 3.3 cm (1.3 in) intervals, or an average particle size of about 10 cm (4 in).

For rubble and back fill materials, it is assumed that the particles have no internal cracks, because they broke into smaller particles along cracks. Assuming that rubble has an average particle size of 3 cm (1.2 in) and is cubic, the specific surface area of rubble is about 200 m^2/m^3 (61 ft^2/ft^3). Because the backfill is material that was produced by blasting, the specific surface area of backfill is assumed to be the same as wall rock that has been fractured by blasting (i.e., 60 m^2/m^3, 18 ft^2/ft^3).

3.2.7 Weathering/Oxidation Zone in Wall Rock

Because chemical weathering requires water, and pyrite oxidation requires water and oxygen, the distance into the wall rock that weathering can occur (the weathering/oxidation zone) depends on the supply of both. Making the simplifying and protective assumption that oxygen delivery into the rock is unlimited, the thickness of the weathering/oxidation zone depends on the delivery of water into the wall rock. Precipitation is the only source of water into the wall rock, so, the thickness of the oxidation zone is essentially the distance that precipitation infiltrates into the wall rock. Figure 27 is a conceptual model for the oxidation zone in the wall rock. Precipitation infiltrates via cracks in the wall rock, and causes the oxidation reaction perpendicular to the plane of the crack. Where there is no water, there is no reaction.

The amount of precipitation, along with the porosity of the wall rock, was used to estimate how far into the wall rock precipitation infiltrates, and therefore the oxidation zone thickness associated with that precipitation event. It was assumed that no evaporation occurs, and that all precipitation from an event immediately infiltrates into the voids of the wall rock. This is protective because it assumes that the smallest events (0.25 mm, 0.01 in) produce infiltration even though they are much smaller than the daily evaporation rate (6 mm, 0.24 in). Due to the high rate of evaporation, it is assumed that following initial infiltration, the wall rock would dry before the next precipitation event, thereby completing the wetting/drying cycle, analogous to HCTs. Based on the porosity (φ) of the rock (0.03 v/v) and the depth of precipitation ($D_{precipitation}$) for each event, the vertical depth of water infiltration (vertical depth of infiltration/weathering, $V_{weathering}$) into the rock can be estimated from:

$$V_{weathering} = \frac{D_{precipitation}}{\varphi}$$

Essentially, the estimated depth of infiltration is about 30-times the depth of precipitation. Instead of accounting for the weathering caused by each precipitation event individually, the model uses an average vertical depth, which is calculated by summing the infiltration depths for all precipitation events and dividing by the total number of events. The average vertical depth of infiltration for a precipitation event is 19 cm (7.5 in). At each time-step in the model, the solutes produced by weathering are washed in by the increasing water level. Because the water table is essentially horizontal at the pit wall, those solutes are washed in from a thin horizontal annulus of wall rock around the pit.

Therefore, the vertical infiltration (weathering) depth is converted to a horizontal weathering distance. The average horizontal distance into the wall can be calculated from the slope of the pit walls (35°) and the vertical weathering depth. This is illustrated in Figure 27. The horizontal weathering zone thickness is about 1.4 times (the sin of 35°) the vertical thickness. Table 11 summarizes the horizontal and vertical depths of the oxidation zone in the rock for the range of precipitation events in the 2005-2007 data. The average event results in a horizontal oxidation zone of about 27 cm (11 in).

This approach ignores some processes that can increase or decrease the depth that precipitation infiltrates into the wall rock. It does not account for unsaturated flow which would increase the distance. On the other hand, it does not include evaporative losses before infiltration. If the local evaporation rate of 0.6 cm/day (0.24 inches/day) were subtracted from each rain event, it would decrease the number of events that produce infiltration from 129 to 29. This decrease would be partially balanced by an increase in average depth of infiltration to 50 cm (70 cm horizontal), and essentially the same amount of oxidation would be predicted. It is assumed that the opposing effects of unsaturated flow and evaporation nominally cancel each other.

3.2.8 Volume of Oxidizing Rubble

In addition to intact wall rock, there is rubble on the haul road, benches, and other relatively level surfaces that will weather and release solutes. The rubble is handled separately from the intact wall rock because it is not uniformly distributed and it has a higher specific surface area. The area of haul roads on which rubble can accumulate was determined at three-meter (10-foot) elevation increments. Figure 28 shows the area of "level" surfaces as a function of elevation for the non-expanded and expanded pits. It is assumed that rubble has accumulated on these surfaces to a depth of 0.20 m, that it is gravel-sized and has a specific surface area of 200 m^2/m^3 (61 ft^2/ft^3).

Rather than attempting to develop a model for how deeply precipitation penetrates rubble and causes oxidation, it is assumed that the whole depth of rubble is subject to oxidation caused by cyclic precipitation. This is a simplifying and protective assumption because the average vertical infiltration of precipitation will be less than in wall rock (19 cm, 7.5 in) due to the greater porosity.

3.2.9 Volume of Oxidizing Backfill

It is assumed that all of the backfill material above the water level at any given time is subject to oxidation because the humidity of the air in the pore spaces will likely be high. Consumption of oxygen from the air in the voids of the non-submerged rock, will likely result in anoxic conditions in the air immediately above the water table in the pit, and also in the pore water of the rock after submergence. Therefore, it is assumed that there is no oxidation of rocks following submersion. For this scenario, the worst-case assumption was applied, which is that the pit is backfilled first and then the water starts to fill the voids from the bottom up. In this case, the duration of oxidation prior to submersion is maximized.

3.2.10 Weathering of Submerged Rock

It is assumed that oxidation of the wall rock and rubble will continue following submersion because the water will have a dissolved oxygen (DO) concentration close to saturation. However, weathering is assumed to continue only at the visibly exposed surfaces and not the surfaces of internal cracks and voids that create porosity. This is because the ground water entering the pit through these cracks is from a deep aquifer and the DO concentration in deep aquifers is essentially zero. In addition, DO from the lake water will not penetrate into voids significantly for two reasons. First, advection of ground water into the pit will prevent significant diffusion of DO into pores. Second, following inundation precipitates and fine-grained material will be redistributed from the walls to relatively flat surfaces where they will tend to clog pore spaces. Numerous studies have shown that in diffusion-dominated sediments, dissolved oxygen rarely penetrates more than 2 mm (e.g., Gunderson and Jorgensen, 1990). To simplify the model, the protective assumption was made that the oxidation rate of pyrite on the wall rock is not decreased by the lower temperature in the hypolimnion (4-10° C, 38-50° F). So, the same release rate is used for all water depths.

4.0 COMPONENT MODELS

4.1 Dynamic Systems Model (DSM)

The Continental Pit Lake DSM was developed using information from the existing pit lake water balance model (SMI, 1999) and additional available site-specific hydrogeologic and climatic data. The DSM model was constructed using the computer code GOLDSIM (2004). This DSM utilizes graphical elements to represent the relationship between the various mass balance components and is similar in a general way to constructing a mass balance model using a spreadsheet. One advantage of using a DSM is that it provides an efficient method for evaluating changes in water and chemical mass balances over time. In the context of pit lake modeling, the DSM includes time-dependent representations of the relevant hydrologic processes and their associated chemistries. The DSM approach also allows for integration of the pit lake model into the larger site-wide water management representation. Combining the water chemistry of various inflows (Tables 4, 5 and 6) and the volumetric inputs of each water source, along with the mass of solutes released from geologic materials, the DSM was used to calculate the bulk chemical composition of the pit water at each time step.

The main components of the DSM include: (1) "stocks", which represent storage reservoirs, in this case, the pit lake, (2) data elements such as daily precipitation, daily evaporation, stage-area relationships, etc., (3) "expressions", which contain the logic for routing water and chemical mass through the model, and (4) the master clock, which tracks time.

The DSM utilizes the standard equation for time-dependent water mass balances:

$$\Delta V = (Q_{in} - Q_{out}) * \Delta t$$

where,

ΔV = the change in water volume storage (L^3)

Δt = the change in time (T)

Q_{in} = the volumetric water flow rate into the stock (L^3/T)

Q_{out} = the volumetric water flow rate out of the stock (L^3/T)

By analogy, the standard equation for time-dependent chemical mass balances is the following:

$$\Delta M = (L_{in} - L_{out}) * \Delta t$$

where:

ΔM = the change in chemical mass storage (M)

Δt = the change in time (T)

L_{in} = the chemical mass loading rate into the stock (M/T)

L_{out} = the chemical mass loading rate out of the stock. (M/T)

At the beginning of the simulation ($t = 0$), it is assumed that the pit is dry and the initial water and chemical mass storage is zero.

4.1.1 Calculation Sequence

At each time step the DSM proceeds through a series of calculations to determine the new water level, water volume and mass of solutes in the water. Once the calculations are performed, the model advances one more time step into the future and repeats the sequence of calculations. The sequence of calculations is as follows:

1. At each time step (Δt) the DSM uses the water inputs and outputs to calculate the incremental change in water storage. Based on the stage volume relationships, (Figures 7 and 10) the DSM computes an associated elevation for each time step.

2. The increment in water level (Δh) is calculated by dividing the volume of water by the surface area of the lake

3. The masses of solute associated with the inflows of ground water, precipitation and runoff are calculated from their volumes and concentrations.

4. The mass of solute from each type of geological material (wall rock and rubble) that was submerged in this time step is then calculated. First, the volume of each type of wall rock submerged in the time step is calculated by multiplying the following: (a) shoreline length (Ls), (b) increment in water level (Δh), (c) thickness of the weathering zone ($V_{weathering}$) and (d) percent of each rock group (R%) at that level. The volume of oxidized pit wall that is leached by inflowing groundwater is shown in Figure 29.

   $$Vr = Ls*\Delta h * V_{weathering} *R\%$$

   Second, the mass (M) of solute released by each rock type is calculated by determining the cumulative areal solute mass (Ma) using the equations in Table 10, and multiplying this by the volume of submerged rock (Vr), and the specific surface area of the wall rock ($SA_{wallrock}$).

   $$M = Ma*Vr* SA_{wallrock}$$

5. The mass of sulfate released from each type of wall rock that has been submerged since the pit started filling is then calculated by multiplying the total area of the rock below the current water level by the slope of the cumulative solute release equation (Table 9) and the time step (Δt).

6. The new masses associated with water inflows and weathering are added to that accumulated in previous time steps, and a new concentration is calculated.

4.2 Chemical Equilibrium Model

The equilibrium geochemical model PHREEQC Version 2.13.2 (Parkhurst and Appelo, 1999) was applied to the unequilibrated solute concentrations calculated by the DSM at various stages of filling and after hydrologic steady-state. The PHREEQC model utilizes the MINTEQ database that was developed by the U.S. Environmental Protection Agency (Allison and others, 1991). The MINTEQ database was used because it represents an extensive compilation of thermodynamic data adequate for addressing a broad range of geochemical conditions. PHREEQC calculates activity coefficients for ion pairs and complexes using the Davies Equation (Langmuir, 1997) and therefore is valid to ionic strengths of approximately 0.5 molal. The PHREEQC input files are provided in Attachment 4.

Prior to using PHREEQC to calculate equilibrium chemical conditions using unequilibrated solute concentrations from the DSM, several runs were made to determine whether seasonal differences in water temperature and of partial pressure of CO_2 (pCO_2), caused by stratification, need to be included. A test was made to determine whether seasonal changes in temperature and stratification have a significant effect on the equilibrated chemistries of the epilimnion and hypolimnion in summer and winter. For the summer, the epilimnion was simulated with an assumed temperature of 22°C and a pCO_2 of $10^{-3.5}$ atmospheres (atm), while the hypolimnion was simulated with a temperature of 4°C and a pCO_2 of $10^{-2.5}$ atm. The higher value for pCO_2 in the hypolimnion is due to a combination of biological activity and increased pCO_2 in ground water, and is consistent with observations of pCO_2 in the hypolimnion of existing pit lakes with near-neutral pH (Eary, 1999). The pO_2 was given a value of $10^{-0.78}$ atm, which corrects the sea level value ($10^{-0.68}$ atm) for the elevation of the pit. This pO_2 is equivalent to a dissolved oxygen concentration of between 8 and 10 mg/L, depending on temperature. Winter conditions were simulated using a temperature of 4°C, a pO_2 of $10^{-0.78}$ atm, and a pCO_2 of $10^{-3.5}$ atm. Thirteen solid phases were allowed to precipitate to equilibrium if they became oversaturated (Table 12). For calcite ($CaCO_3$[s]), at a slight degree of oversaturation (saturation index = 0.5) was allowed. Sorption of trace elements by hydrous ferric oxide (HFO) was also included. The amount of HFO was equal to the amount of iron released from geologic materials by weathering (Section 3.2.5). The physical properties for HFO were taken from Dzombak and Morel (1990), and the area available for adsorption was allowed to change dynamically as a function of the mass of HFO predicted to precipitate. Due to the potential for coating of the carbonate rocks on the pit walls with iron oxides and other precipitates, water was not allowed to equilibrate with the wall rock. The unequilibrated concentrations for the non-expanded pit after 5 and 200 years were used for the test. Table 13 summarizes the concentrations of solutes for the unequilibrated solution, and the equilibrated epilimnion, hypolimnion and winter conditions. The results show that simulation of the three pairs of temperature and pCO_2 conditions yield essentially the same result, and that it does not affect the predicted concentrations significantly.

Based on these results, all subsequent geochemical equilibrium calculations used a fixed set of conditions (a temperature of 15°C, pCO_2 of $10^{-3.5}$ atm, and pO_2 of $10^{-0.78}$ atm), the same allowed solid phases, and sorption to HFO.

5.0 RESULTS

The DSM was used to predict both the water level and unequilibrated solute concentrations for the three closure scenarios (the non-expanded pit option, the expanded pit option, and the partial backfill with evapotranspiration option). After equilibrating the DSM-projected solute concentrations with PHREEQC, the concentrations of solutes in the pit were compared to NMED water quality standards.

5.1 Non-expanded Pit

Figures 30 through 33 show the water level, shoreline length, water surface area, and lake volume as a function of time while the non-expanded pit fills. The time frame for predictions was set at 300 years because the pit approaches hydraulic steady state during the first 200 years and results from the next 100 years were used to evaluate the components of the water balance after the pit reaches steady state. Table 14 summarizes the inputs and outputs of water at several points in time while the pit is filling. The results indicate that water level in the non-expanded pit lake is expected to reach the steady-state elevation of 2009.3 meters (6,592.5 feet) approximately 250 years after closure, and that the final water depth will be approximately 95.3 meters (313 feet).

The cumulative mass of sulfate entering the pit from each source is shown in Figure 34. The results show that in the first 250 years the mass of sulfate that enters the pit is approximately 12,300,000 kilograms (kg). The sources of sulfate in decreasing order of importance are ground water (11,600,000 kg, 94%), runoff (514,000 kg, 4%), weathering of wall rock and rubble before and after submergence (82,000 kg, 1%), and precipitation (19,000 kg, 0.1%). The mass of sulfate released by wall rock and rubble in the first 250 years is dominated by the Lake Valley (41,000 kg) and Fusselman-Montoya (30,000 kg), with lesser amounts from the Abo-Syrena Formations (6,200 kg) and Hanover-Fierro Formations (4,500 kg).

Figure 35 shows the temporal pattern in loading rate of sulfate to the pit for sources other than precipitation. The initial decrease in sulfate loading from ground water is due to decreasing ground water inflow rate. After about 100 years, when the water level is about 95% of its steady-state value, the total loading rate of sulfate to the pit becomes fairly constant at about 46,000 kg/yr. The sources and loading rates of sulfate are, ground water (44,500 kg/yr, 95%), runoff (2,050 kg/yr, 4%), wall rock (100-200 kg/yr, 0.2-0.4%) and precipitation (90 kg/yr, 0.1%). Figure 36, details the temporal pattern in sulfate release from wall rock and rubble, with the highest rates occurring early due to rapid filling (1,500 kg/yr), and low rates later on (100 kg/yr) when weathering rates are reduced due to submergence.

Table 15 summarizes the unequilibrated concentrations of solutes predicted for the non-expanded pit after 10, 50, 100, 200 and 300 years. Figure 37 shows the unequilibrated concentration of sulfate in the water as a function of time, along with the equilibrated concentrations at several points in time during filling. About 250 years after the pit starts filling, the rate of increase in concentration of sulfate is predicted to decrease due to precipitation of gypsum. The equilibrated concentrations of all solutes after 10, 50, 100, 200 and 300 years of filling, along with NMED surface water quality standards, are shown in Table 16. The model predicts that water quality standards for livestock watering are exceeded for selenium after 200 years.

5.2 Expanded Pit

Figures 38 through 41 show the water level, shoreline length, surface area, and lake volume as a function of time while the expanded pit fills. Table 17 summarizes the inputs and outputs of water at several points in time while the pit is filling. The results indicate that the water level in the expanded pit lake will reach the steady-state elevation of 1974.7 m (6,479 ft) approximately 300 years after closure, and that the final depth will be approximately 146m (479 ft).

Figure 42 shows the cumulative mass of sulfate entering the pit from each source for 300 years. The results show that in the first 300 years, while the water level approaches steady-state, the total mass of sulfate that enters the pit is approximately 16,800,000 kg. The sources of sulfate in order of decreasing importance are ground water (15,700,000 kg, 93%), runoff (962,000 kg, 5%), weathering of wall rock and rubble (173,000 kg, 01%), and precipitation (25,000 kg, 0.1%). The mass of sulfate released by wall rock and rubble in the first 300 years was dominated by the Lake Valley Formation (82,000 kg), with lesser amounts from Fusselman-Montoya (68,000 kg), Abo-Syrena (14,000 kg), and Hanover-Fierro (9,000 kg).

Figure 43 shows the loading rate of sulfate to the pit for the first 300 years for sources other than precipitation. The loading rate of sulfate from ground water decreases gradually over the first 100 years due to decreasing inflow rate. After 300 years, the total loading rate of sulfate to the pit becomes fairly constant at about 52,000 kg/yr. The rates of sulfate inputs from ground water (50,000 kg/yr, 94%), runoff (3,200 kg/yr, 6%), and precipitation (100 kg/yr, 0.2%) are essentially the same as the first 200 years. Weathering of the geologic material decreases significantly from an initial rate of about 15,000 kg/yr to less than 200 kg/yr after 300 years. Figure 44 details the temporal pattern of sulfate loading from wall rock.

Table 18 summarizes the unequilibrated concentrations of solutes predicted for the non-expanded pit after 10, 50, 100, 200 and 300 years. Figure 45 shows the unequilibrated concentration of sulfate in the water as a function of time, along with the equilibrated concentrations at several points in time during filling. The equilibrated concentrations of all solutes after 10, 50, 100, 200 and 300 years of filling, along with NMED surface water quality standards are shown in Table 19. The model predicts that water quality standards for livestock watering are exceeded for selenium at 10 years and then at 300 years.

5.3 Partially Backfilled with Evapotranspiration

For this scenario, backfill material is added to an elevation that results in balancing the total water inflows (i.e., the sum of ground water, runoff and precipitation) with the loss from evapotranspiration (i.e., the product of the surface area of the backfill and evapotranspiration rate). The surface area of the backfill as a function of elevation was determined from Figure 9, and the ET rate was assumed to be 160 cm/yr (64 in/yr). Ground water inflow as a function of elevation was calculated by interpolating values from Figure 13. Runoff from the pit walls above the backfill and infiltration into the backfill from precipitation, were calculated assuming 15 percent infiltration (SMI, 1999) of the annual precipitation (46.5 cm/yr, 18.3 in/yr) that lands on the total area of the pit watershed 370,000 m^2 (3,980,000 ft^2). The elevation at which evapotranspiration equals total inflows was determined graphically from Figure 46. The cross-over occurs at a water table elevation of 1,986.8 m (6,518.7 ft), and a water depth in the pit of 72.7 meters (238.5 feet). To allow for the ground water level to rebound during the low ET months and prevent flooding from killing the plants, an additional 1.5 meters (5 ft) of material is included for the root zone, which brings the backfill elevation to 1,988.3 m (6,523.6 ft). It is assumed that the top 1.5 meters of material for the root zone is from a source other than the oxidized leach cap and will not release a significant amount of solute to the water. The effective porosity of the backfill is assumed to be 25 percent.

Figure 47 shows the predicted water elevation in the backfilled pit as the pit fills with water. The water level rises to about a quarter of its final depth within two years, and about half its final depth in 5 years. The plot shows that when the water rises to the elevation of the root zone and evapotranspiration starts, the water table fluctuates about 0.4 meters (1.3 feet) seasonally due to evapotranspiration. At steady-state, the water depth is about 20 meters (66 feet) lower than in the non-expanded pit without backfill. The relatively fast filling rate, compared to the open pit, is due to backfill occupying 75% of the pit volume and postponement of any water loss (evapotranspiration) until the water level reaches the root zone.

Figure 48 shows the cumulative mass of sulfate that enters the pit from each source for the first 100 years. When the water table reaches the root zone in 20 years, the results show that the sources of sulfate in decreasing order of importance are weathering of backfill (3,800,000 kg, 77%), ground water (1,050,000 kg, 21%), runoff (87,000 kg, 1.8%) and weathering of wall rock (11,000 kg, 0.2%). After 20 years the rate of sulfate input from backfill drops to zero, while others remain fairly constant (ground water (46,900 kg/yr), and runoff (2,800 kg/yr)). At 100 years the sources of accumulated sulfate in decreasing order of importance are ground water (4,700,000 kg, 53%), weathering of backfill (3,800,000 kg, 43%), runoff (430,000 kg, 4.8%) and weathering of wall rock (11,000 kg, 0.1%).

Unlike Scenarios 1 and 2, in which the pit is open and the water column is well-mixed, the concentrations of solutes in the pore water of the backfill cannot be calculated by simply dividing the mass of solute by the volume of water. This is because the solutes are not introduced to the water uniformly, and vertical and horizontal mixing of water and solutes in porous media like backfill is much slower than in open water. In this scenario the rising water table is continuously dissolving weathering products from the backfill and carrying them upward in the direction of water flow. Mixing of pore water and solutes in the backfilled pit will be similar to mixing in ground water which is caused by mechanical dispersion as the pore water takes irregular paths through the backfill. This relatively low rate of dispersion can maintain large concentration gradients.

A separate model was developed in order to predict the concentration gradients that would develop in the pore water. First, it is assumed that the rising water table in the pit is horizontal due to the hydraulic properties of the backfill. As the water level in the pit rises, it is assumed that all of the soluble weathering products of the backfill at a given level dissolve as soon as the water reaches that level, and that no precipitates are allowed to form. So, as the water level rises, more solutes are dissolved into the upper portion of the saturated zone and are transported upward with the moving water. The result is that higher concentrations of solutes will develop in the upper layer of water, while the concentrations in pore water near the bottom of the pit will be essentially the same as the inflowing ground water. The resulting concentration gradient that develops between these layers depends on how well dispersion mixes the high concentrations downward with respect to the water table.

5.3.1 Vertical Concentration Gradient Model

To estimate the vertical concentration gradient in the pore water of the backfill, a numerical model was developed. A numerical rather than analytical approach is used because the temporal pattern of solute loading at the upper boundary is not a simple function, and an analytical solution does not exist. The model simulates a vertical column of backfill and water, by including the loading of sulfate at the upper water surface, and vertical dispersion of solutes within the column. To simplify the mathematical representation, the model uses the upward moving water table as a fixed spatial reference (a Lagrangian frame of reference). Only the upper portion of the column is modeled because dispersion will only slowly transport solutes downward relative to the water table, and the concentrations of solutes in the root zone are of interest. The column of backfill and water was segmented into 30 cells, and cell thickness (z) was changed depending on the time it takes for the water table to rise through the backfill and the thickness of the backfill. Figure 49 shows how the physical system and processes are represented. A mass balance equation is written for each cell. For Cell 1 at the top of the water table, the mass balance equation is:

$$M_{1,t} = M_{1,t-1} + L - D * A * P * \left(\frac{C_{1,t-1} - C_{2,t-1}}{z} \right)$$

where,

$M_{1,t}$ = mass in Cell 1 at time t,

$M_{1,t-1}$ = mass in Cell 1 at previous time step,

L = loading rate of sulfate ($mg/m^2/y$),

D = dispersion coefficient (m^2/y),

A = area of cell (m^2),

P = porosity,

$C_{1,t-1}$ = sulfate concentration in Cell 1 at previous time step,

$C_{2,t-1}$ = sulfate concentration in Cell 2 at previous time step, and

z = characteristic mixing length, cell thickness (m).

For all other cells, the only inputs and outputs of mass are through dispersive mixing with overlying and underlying cells, so the mass balance equation is:

$$M_{n,t} = M_{n,t-1} - D * A * P * \left(\frac{C_{n,t-1} - C_{n-1,t-1}}{z} \right) - D * A * P * \left(\frac{C_{n,t-1} - C_{n+1,t-1}}{z} \right)$$

where,

$M_{n,t}$ = mass in cell n at time t,

$M_{n,t-1}$ = mass in cell n at previous time step

$C_{n,t-1}$ = sulfate concentration in Cell n at previous time step

$C_{n-1,t-1}$ = sulfate concentration in Cell (n-1) at previous time step

$C_{n+1,t-1}$ = sulfate concentration in Cell (n+1) at previous time step

Dispersion coefficients (D) for ground water are commonly reported as longitudinal (along the direction of flow) and transverse. Because mixing along the direction of flow is of interest (in this case vertical), the longitudinal dispersion coefficient is the relevant parameter. The dispersion coefficient used in the model (3 m^2/year) is based on values at similar low velocities (1.7 to 9 m/yr, 6 to 30 ft/yr) and porosity (0.25) (Crane and Gardner, 1961). Dispersion coefficients are positively correlated with water velocities, and although D will decrease slightly as the water level rises in the backfill, a fixed value is used.

The loading rate (L) of sulfate to the top of the rising water table depends on how long the backfill weathered prior to being submerged, and the amount of backfill inundated per unit time. These are both determined from the rate at which the pit fills, shown in Figure 47. The weathering rate and release of sulfate (W) from the backfill material (Hanover Mountain oxidized leach cap) was determined from humidity cell test WRC-09, and is given in Table 9. At any time during filling, the plan-view areal loading rate (L) for sulfate to the water table (mg/m^2/year) is the product of the backfill weathering (W), and the upward velocity of the water table (V), corrected for porosity (φ) and specific surface area (SSA) of the backfill.

$$L(t) = W(t) * V(t) * (1 - \varphi) * SSA$$

where

t = time since water started filling pit (years)

$W(t) = 171 \times t + 332$ (mg/m^2)

$V(t)$ = water table migration velocity (m/yr)

φ = porosity (0.25)

SSA = specific surface area of backfill (60 m^2/m^3)

The water table migration velocity is merely the slope of the water elevation in Figure 47, which is nominally,

$$V(t) = 9.00e^{(-0.1079t)} \quad \text{(m/yr)}$$

Figure 50 shows the estimated loading rate of sulfate from the backfill to the water table as a function of time, starting from the deepest point in the pit (1,914.1 m, 6,280 ft) and ending at the bottom of the root zone (1,986.8 m, 6,518.7 ft). The early part of the curve shows the effect of a rapidly rising water level coupled to a short duration of weathering, while the later part shows the effect of a slowly rising water level coupled with a long duration of weathering. This curve was developed for the deepest part of the pit where there would be about 73 m (239 ft) of backfill through which the water rises and accumulates solutes. However, closer to the edge of the pit where the backfill thickness is less and the water level does not reach that elevation until later, only the later portion of the loading curve is applicable.

Initial and boundary conditions for the concentration of sulfate in the water column and ground water were set at 240 mg/L, which is the concentration in the surrounding ground water. To keep the model stable and avoid numerical dispersion, both the time-step and cell thickness were adjusted depending on the length of time being modeled and thickness of the backfill. The time-step in the model ranged from 0.001 to 0.01 years, while the cell thickness ranged from 0.05 to 0.1 m (2 to 4 inches), depending on the depth of the backfill.

The model was used to predict the concentration profiles of sulfate when the water table reached the bottom of the root zone and before any evapotranspiration occurs, which is about 20 years. The evolution of the sulfate concentration profile at the center of the pit was determined at four depths during filling (25% full, 19 m; 50% full, 37 m; 75% full, 56 m; and 100% full, 73 m). These concentration profiles are shown in Figure 51. The results indicate that 1) the highest concentrations would exist at the water table, 2) there would be strong vertical concentration gradients, and 3) that most of the solute mass will be contained in the upper several meters of the water column. The figure also shows that the vertical extent of dispersion increases with increasing time. The model predicts that when the water table reaches the root zone after 20 years of filling, the concentration of sulfate ranges from about 42,000 mg/L in the top 0.1 meters (0.3 feet), to about 300 mg/L at a depth of 2.0 meters (6.6 feet), and over 90 percent of the solute mass released by backfill is contained in the top 1 meter (3.3 feet) of water.

Because the thickness of backfill that the rising water contacts varies with location in the pit, the amount of solute in the water at the end of 20 years also varies with location. Therefore, the concentration profiles that would occur at four different points in the pit having different backfill thicknesses were also calculated. Representative backfill thicknesses were determined by dividing the total area of the pit (150,000 m^2) into four equal areas (37,000 m^2) and determining from Figure 9, the depth/elevation intervals that correspond to 25 percent increases in area. The depth intervals from the top down are 0 to 6 meters, 6 to 21 meters, 21 to 29 meters, and 29 to 73 meters. These values mean that 25 percent of the pit area has an average depth of 3 meters, 25 percent of the pit area has an average depth of 13.5 meters, 25 percent of the pit area has an average depth of 25 meters, and 25 percent of the pit area has an average depth of 51 meters. Figure 52 shows the sulfate concentration profiles that are predicted to develop at the end of filling for different backfill thicknesses. The results indicate that solute concentrations would decrease from the middle of the pit towards the edge of the pit. Because transverse dispersion is typically an order of magnitude less than longitudinal dispersion, there would be little horizontal mixing in the pit and these concentration gradients would diminish slowly. With time, the continuous loading of solutes from runoff around the perimeter of the pit would decrease the horizontal concentration gradients. Beyond 20 years, evapotranspiration, which is greater than infiltration, would remove water from the root zone and increase solute concentrations at the water table.

The unequilibrated concentrations of solutes predicted with the dispersion model have ionic strengths (greater than 1 molal) that are above those for which activity coefficients can be calculated with PHREEQC. Therefore, equilibrated concentrations are not calculated. Because there is little horizontal mixing, the pore water will not effectively contact the wall rock which has calcite, the alkalinity of the pore water is expected to remain low and not precipitate calcite. Limited contact with wall rock would also keep most of the sulfate in solution due to relatively little calcium compared to sulfate.

5.4 Complete Backfill with Flow-through

This scenario was not run in its entirety. Results from Scenario 3, partial backfill with evapotranspiration, indicated a significant accumulation of solutes in the upper part of the water in the pit. Water balance calculations indicate that backfill would have to be added to an elevation of 2,100 meters (6,900 feet) for the ground water to flow through. Given this 250 meter depth of backfill, and the length of time to reach a steady state water level, sulfate concentrations would reach twice that estimated for Scenario 3. With flow through, this high sulfate (>100,000 mg/L) water would ultimately discharge to the regional ground water, which would clearly violate NMED regulations for sulfate in ground water (600 mg/L).

5.5 Sensitivity Analysis

The values of parameters and variables used in the model have different levels of uncertainty associated with them depending on the source and method of estimation. The values were derived from a combination of laboratory testing, estimates based on known relationships, and values based on relatively precise measurement techniques. The uncertainties in some of these values could span an order of magnitude. To determine how these uncertainties propagate through the model and affect the results, a range of values for each major variable were used in the model to determine how sensitive the model predictions are to the values.

After running the model for the nominal conditions discussed in Section 3, the dominant parameters and variables were identified and selected for testing in the sensitivity analysis. These test variables include both hydrologic processes and geologic material weathering processes. In the sensitivity analysis, the value of a particular variable is varied through a range that brackets the expected range of uncertainty, while all other variables are kept at their nominal values. This range includes the contributions of analytical uncertainty, natural variability, and uncertainties in assumptions. The results of these sensitivity tests are then compared with the results obtained using the nominal values to determine if the model has unexpectedly non-linear responses, and how strongly each variable controls model predictions.

As a terminal lake that takes decades to fill, the flow rate of ground water into, precipitation on and evaporation from the pit are the dominant hydrologic processes, and the sensitivity of the model to these variables are tested. The values for these variables are increased and decreased by 10% around their nominal values.

Nominal test conditions also showed that solute contribution from major sources were in the following order: ground water > runoff > wall rock >> precipitation. Because ground water is the primary source and there is limited data on ground water chemistry from the formations that will discharge to the pit, the sulfate concentration for ground water is tested. The values of release rate of solutes from the wall rock are not tested because the values used are protective over-estimates based on the data. Rather, parameters developed from the conceptual model of how much wall rock is available for oxidation (e.g., reaction zone thickness) are tested.

Several estimated variables about the physical characteristics of the geologic materials, such as specific surface area, crack size and porosity, can affect model predictions. For example, the assumed porosity affects the calculation for the specific surface area of the wall rock, as well as the depth to which precipitation infiltrates the rock. There are a number of variables that are not tested because their values are known relatively precisely. These variables include the physical dimensions of the pit, and the distributions of rock formations on the walls. Sensitivity analysis was only performed on the non-expanded pit (Scenario 1).

5.5.1 Ground Water Flow Rate

The ground water flow rate was increased and decreased by 10% above and below the nominal values (shown in Figure 13) for non-expanded pit. Changing the ground water flow rate resulted in a change of the predicted steady state elevation as shown on Figure 53. Increasing the ground water flow rate by 10% resulted in a change of the predicted steady state elevation from a nominal value of 2,008 m (6,588 ft) at the end of 200 years to 2,013 m (6,604ft). Decreasing the evaporation rate by 10% caused the predicted elevation of the pit lake to reach 2,002 m (6,568 ft) at the end of 200 years. Figure 54 shows the time trend for unequilibrated sulfate concentration for both sensitivity tests and under the nominal flow rate.

5.5.2 Precipitation

Precipitation was increased by 10% and decreased by 10% around the nominal value of 18.3 inches/year. The effect of changing the evaporation rate on the water elevation is shown on Figure 55. Increasing the precipitation rate by 10% resulted in a change of the predicted elevation of the pit lake from a nominal value of 2,008 m (6,588 ft) at the end of 200 years to 2,011 m (6,598 ft). Decreasing the evaporation rate by 10% caused the predicted elevation of the pit lake to reach 2,005 m (6,578 ft). Figure 56 shows the unequilibrated sulfate concentrations for these two tests along with the nominal scenario.

5.5.3 Evaporation

The evaporation rate was increased by 10% and decreased by 10% around the nominal value of 55.8 inches/year. The effect of changing the evaporation rate on the water elevation is shown on Figure 57. Increasing the evaporation rate by 10% resulted in a change to the predicted elevation of the pit lake from a nominal value of 2,008m (6,588ft) at the end of 200 years to 2,000m (6,562ft). Decreasing the evaporation rate by 10% caused the predicted elevation of the pit lake to reach 2,015 m (6,611 ft).

Figure 58 shows the effect of changing the evaporation rate on the unequilibrated sulfate concentrations in the pit lake. Increasing the evaporation rates results in a significantly higher concentration (25 to 30 percent) of sulfate after 100 years. The higher sulfate concentration is a consequence of increased evapoconcentration of solutes in the pit lake.

5.5.4 Ground Water Sulfate

The concentration of sulfate in ground water that flows into the pit was tested at the maximum value of 316 mg/L (measured on April 2007) in monitoring well MW-5. The maximum sulfate concentration measured at MW-5 is about 30% greater than the mean value of 242 mg/L, which is used as the nominal value in the pit lake chemistry model. As expected, the unequilibrated sulfate concentrations in the pit lake (shown on Figure

59) increase by about 30%. This shows that concentration of sulfate in the pit is linearly related to concentration of sulfate in the ground water.

5.5.5 Porosity

Initially, it was intended that the sensitivity of the model to porosity of the wall rock would be tested. However, an increase in porosity decreases the calculated depth that precipitation will infiltrate in an inversely proportional manner, thereby decreasing the thickness of the oxidation zone. At the same time, an increase in porosity increases the specific surface of the wall rock in a directly proportional manner. Because the area of wall rock exposed to weathering is in part the product of the specific surface area (m^2/m^3) and the oxidation zone thickness (m), the effects cancel, and porosity is not an important variable. Rather, the crack thickness is the important variable.

5.5.6 Crack Thickness

The thickness of cracks in the wall rock was decreased 50% (0.5 mm) and increased 100% (2mm) around the nominal value of 1 mm. For a given porosity, halving the crack thickness doubles the number of cracks and doubles the specific surface of the wall rock, which doubles the amount of solute released by the wall rock. Conversely, doubling crack thickness halves the number of cracks (and specific surface of the wall rock), which halves the amount of solute released by the wall rock. Figure 60 shows the long term trend for unequilibrated sulfate concentration in the pit lake for the two different crack thickness values (0.5 mm and 1 mm) along with the nominal value of 1mm. The sulfate concentration profiles are almost indistinguishable from each other because the change in crack thickness only changes the sulfate loading from the wall rock and rubble (Figure 61). Since wall rock loading only contributes about 1% of the overall chemical mass loading to the pit lake, the magnitude of change in wall rock loading is not large enough to affect the concentration profiles.

5.5.7 Dispersivity

For the backfill scenario, the dispersion coefficient used in the vertical concentration gradient model was varied from a nominal value of 3 m^2/yr to 1 m^2/yr and 10 m^2/yr. As shown on Figure 62, increasing the dipsersivity decreases the predicted sulfate concentrations in the root zone.

6.0 CONCLUSIONS

Results from the pit lake model will be used in Condition 32 of discharge permit DP-1403 to evaluate potential ground water and surface water impacts, and select abatement options and closure alternatives for ground water.

The water balance for the open pit scenarios confirms that for the ground water inflow rates predicted by the numerical ground water model (Telesto, 2008)~~,~~ and evaporation, the primary loss, the pit lake will be terminal~~,~~ with no outflows. Ground water is the primary water source, with precipitation on the surface of the lake and runoff from wall rock contributing less than 40 percent.

Results of simulations for the open pit scenarios (Scenarios 1 and 2) indicate that sulfate and calcium are the major dissolved constituents, and that the majority of the calcium and sulfate loading to the pit lake is from the ground water. Runoff from pit walls above the steady state water level is the second largest (after ground water) contributor of chemical mass to the pit, while weathering of geologic materials below the steady state water level contributes a negligible (relative to ground water) amount of sulfate to the pit. As should be expected, precipitation contributes a negligible amount of solute to the pit.

Because the pit forms a terminal lake with evaporation, the concentration of chemical mass in the pit lake will continue to increase with time. This is evident in the increase in predicted chemical concentrations of all constituents for various time periods shown in Tables 16 and 19. The increase in solute mass with time will be somewhat offset by precipitation of minerals from the water. Selenium is the only constituent that is predicted to be out of compliance with NMED surface water regulations within the 300 year simulation period. Surface water standards are chosen for comparison because the water in the pit lake would never become ground water. The concentrations of trace metals are predicted to comply with NMED regulations largely due to sorption onto ferrihydrite which will precipitate in the lake. The pit lake chemistry is expected to be characterized by a circum-neutral pH solution due to the predominance of limestone in the wall rock formations. The water will also have high total dissolved solids (predominantly calcium and sulfate).

Sensitivity analyses show that changes to evaporation rate and the concentration of sulfate in ground water have the potential to significantly alter the pit lake chemistry. Although the wall rock weathering component of the model shows significant sensitivity to the assumed thickness of cracks, this does not affect the predicted concentrations in the pit because the mass generated by weathering is much smaller than the mass carried in by ground water.

Unlike the open pit scenarios, when a steady-state water level is reached in the backfilled pit in 20 years, the majority of solutes originate from weathering of geologic materials, in this case the backfill. However, with continued ground water inflow, the mass of solutes contributed by ground water dominates other sources after 100 years. Results of simulating a backfilled pit with evapotranspiration, show that high concentrations of solutes will develop within the pore water of the backfill. This is in part due to the material assumed to be used for backfill (Hanover Mountain oxidized leach cap, belonging to the Colorado Formation). The majority of solute mass will be focused in the top 1 meter, just below the water table. Concentrations were compared to NMED ground water standards and predicted concentrations exceed standards for sulfate and most trace metals. No evaluation or selection was made of plants that will tolerate the predicted concentrations.

For the backfill scenario, the dispersion model showed significant sensitivity to the value of the dispersion coefficient, with a 3-fold higher value decreasing the concentrations predicted in the root zone by 40 percent, and a 3-fold lower value increasing the concentrations by 60 percent.

Overall, the predicted water chemistry for the non-expanded and expanded open pits yield fewer predicted exceedances of NMED surface water quality standards, than for the backfilled pit. Improvements in predicted backfilled pit water chemistry may be achieved by using a different source of backfill, or augmenting the leach cap material with a carbonate-rich material.

7.0 REFERENCES

Allison, J.D., D.S. Brown, and K.J. Novo-Gradac. 1991. MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Systems. Version 3.0 User's Manual. Environmental Research Laboratory, Office of Research and Development, U.S Environmental Protection Agency, Athens, Georgia. 106 pp.

ASTM International. 2002. Standard Test Method for Accelerated Weathering of Solid Materials Using a Modified Humidity Cell. ASTM Method D 5744-96. ASTM International, West Conshohocken, PA.

Brutsaert, W. 1982 Evaporation Into the Atmosphere: Theory, History, and Applications. Published by Springer.

Crane, F.E., and G.H.F. Gardner. 1961. Measurements of transverse dispersion in granular media. Chem. Eng. Data, 6, pp 283-287.

Daly, C., R.P. Neilson, and D.L. Phillips. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. Journal of Applied Meteorology. 33: 140-158.

Davis, G.B. and A.I.M. Ritchie. 1987. A Model Of Oxidation In Pyritic Mine Wastes: Part 3, Import of Particle Size Distribution. Appl. Math. Modelling. 11: 417-422.

Davis, G.B. and A.I.M. Ritchie. 1986. A Model of Oxidation in Pyritic Mine Wastes: Part 1, Equations and Approximate Solutions. Appl. Math. Modelling. 10: 314-322.

Davis, G.B., G. Doherty, and A.I.M. Ritchie. 1986. A Model of Oxidation in Pyritic Mine Wastes: Part 2, Comparison of Numerical and Approximate Solutions. Appl. Math. Modelling. 10: 323-329.

Dzombak, D.A. and F.M.M. Morel. 1990. Surface Complexation Modeling, Hydrous Ferric Oxide. John Wiley & Sons, New York.

Eary, L.E. 1999. Geochemical and Equilibrium Trends in Mine Pit Lakes. Appl. Geochem. 14: 963-987.

GoldSim. 2004. GoldSim Dynamic Systems Model. GoldSim Technology Group LLC; 22516 SE 64th Place, Suite 110. Issaquah, Washington, 98027-5379.

Gundersen, J.K. and B.B. Jorgensen. 1990. Microstructure of diffusive boundary layer and the oxygen uptake of the sea floor. Nature, 345: 604-607.

Hillesland, L.L., W.T. Worthington, and R.B. Hawkin. 1994. General Geology of the Continental Mine, Grant County, NM. Cobre Mining Company, Inc.

Holdren G.R., Jr., Berner R.A. Mechanism of feldspar weathering: I. Experimental studies. Geochim. Cosmochim. Acta 1979;43:1161-1171

Jones, W.R., R.M. Hernon, and S.L. Moore. 1967. General Geology of Santa Rita Quadrangle, Grant County, New Mexico.

Kohler, M.A., Nordenson, T.J., and Fox, W.E., 1955 Evaporation from Pan and Lakes US Weather Bureau Research Paper 38. Washington, DC.

Langmuir, D. 1997. Aqueous Environmental Chemistry. Prentice Hall, New Jersey. 600 pp.

Nicholson, R.V., and J.M. Scharer 1994. Laboratory Studies of Pyrrhotite Oxidation Kinetics. Environmental Chemistry of Sulfide Oxidation. ACS Ser. 550, pp14-30.

Parkhurst, D.L. and C.A.J. Appelo. 1999. User's Guide to PHREEQC (Version 2.0). A Computer Program for Speciation, Reaction-Path, Advective-Transport, and Inverse Geochemical Calculations. Water-Resources Investigations Report 99-4259. U.S. Geological Survey, Lakewood, Colorado.

Rimstidt, J.D., Chermak, J.A., and Gagen, P.M. 1994. Rates of reaction of galena, sphalerite,chalcopyrite and arsenopyrite with Fe(III) in acidic solutions in: Alpers,C.N and Blowes,D.W.(eds) Environmental Geochmistry of Sulfide oxidation. American Chemical Society Symposium series 550, Washington, 2-13

Siskind, D.E., and R.R. Fumanti. 1974. Blast-produced fractures in Lithonia granite.U.S. Department of Interior. Twin Cities Mining Research Center. Minneapoplis,MN.

SMI. 1999. Baseline Characterization of the Hydrology, Geology, and Geochemistry of the Proposed Continental Mine Expansion Project, Cobre Mining Company, Inc. Prepared for Cobre Mining Company, Inc. (Hurley, NM) by Shepherd Miller Inc., (Fort Collins, CO).

Telesto Solutions, Inc. 2005. Workplan to Predict Continental Pit Lake Water Quality Following Mine Closure. Prepared for Cobre Mining Company (Hurley, NM) by Telesto Solutions, Inc. (Fort Collins, CO). December.

Telesto Solutions, Inc. 2008. Supplemental Ground Water Study Continental Mine Expansion, Grant County, New Mexico. Prepared for Cobre Mining Company c/o Freeport McMoRan Copper.